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Improving Integrated Surface and Groundwater Resources Management in a Vulnerable and Changing World

Edited by

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Improving Integrated Surface and Groundwater Resources Management in a Vulnerable and Changing World

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Publ. 321 (*2008*) ISBN 978-1-901502-59-6, 214 + x pp; £48.00

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Outcome of a symposium organised jointly by the IAHS International Commission on Groundwater and the International Association for the Physical Sciences of the Oceans, IAPSO, in 2007, the research presented here forms an important contribution to the literature.

Publ. 312 (*2007*) ISBN 978-1-901502-04-6; 344 + x pp; £64.00

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edited by Ioulia Tchiguirinskaia, Siegfried Demuth & Pierre Hubert

A joint IAHS / UNESCO-IHP publication – proceedings of the 9th Kovacs Colloquium

A review of the practice and realities of undertaking research for river basin management (how to involve the public as stakeholders, building trust with decision-makers, the research funding situation), the tools we have available (hydrological models, how good are they, how can we reduce uncertainties and explain them to policy makers), their application and the current situation regarding water monitoring and management in El Salvador, India, Romania, Russia and South Africa. The authors' main conclusions and recommendations are summarized in a final section which proposes issues for future consideration in hydrological research and management.

Publ. 323 (*2008*) ISBN 978-1-901502-69-5, 154 pp; £40.00

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edited by Xiaohong Chen, Yongqin David Chen, Jun Xia & Hailun Zhang

Many regions in Asia are experiencing unprecedented rapid development causing pressures on the environment and management of natural resources. An insight to the innovative work there:

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- Hydrological modelling
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- Optimal allocation of water resources
- Resources/watershed management
- Water environment and aquatic ecosystems

Publ. 319 (*2008*) ISBN 978-1-901502-44-2 422 pp; £72.00

Bringing Groundwater Quality Research to the Watershed Scale

edited by Neil R. Thomson

Sustainable economic growth requires plentiful supplies of high-quality water. Pressure on resources globally has forced the international groundwater science, engineering and regulatory community to recognize current limitations of research and management. Integrated and technically feasible approaches tackling local issues and watershed-scale concerns concurrently are required. This volume, an outcome of GQ2004 (Waterloo, Canada) deals with: Global and national perspectives; Contaminant input processes; Site characterization; Management and decision making; Natural attenuation processes and applications; *In situ* remediation; and Flow and transport modelling at various scales.

Publ. 297 (*2005*) ISBN 978-1-901502-18-3; 576 + xiv pp; £85.00

Wastewater Re-use and Groundwater Quality

edited by Joop Steenvoorden & Theodore Endreny

Re-use of treated wastewaters is important for conserving regional water resources. However, questions arise when this solution is envisaged, such as: Which pre-treatment is necessary? What are the pathogenic risks? What is the long-term environmental sustainability? These, and issues such as centralized *vs* de-centralized systems were discussed at a symposium jointly organized by IAHS, IAH and the UNESCO Division of Water Sciences.

Publ. 285 (*2004*) ISBN 978-1-901502-52-7; 110 + vii pp; £29.30

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Preface

Many parts of the world are extremely vulnerable environments with declining potable water resources and an increasing risk of extreme events due to population growth, intensification of agriculture and urbanisation, and limited development opportunities. With the increasing difficulties of meeting human demands on water resource quantity and quality, new concepts in water management need to be explored, with a move away from centralised command and control approaches to more participatory multi-stakeholder approaches that have the potential to be more flexible and responsive. New concepts, such as Integrated Water Resources Management (IWRM) and Adaptive Management (AM) are being put into practice, but their scientific basis has not been fully explored.

During the joint Convention of the International Association of Hydrological Sciences (IAHS) and the International Association of Hydrogeologists (IAH), 6–12 September 2009, in Hyderabad, India, a symposium was held entitled *Improving Integrated Surface and Groundwater Resources Management in a Vulnerable and Changing World.* The Symposium was organised by the IAHS International Commission on Water Resources Systems (ICWRS), together with the IAHS International Commissions on Water Quality (ICWQ), Remote Sensing (ICRS) and the International Association of Hydrogeologists (IAH). The broad coverage and the multi-facetted nature of the subject area are reflected in the large number of contributions to the symposium drawn from a range of disciplines. Out of the contributions, 50 papers were selected for this volume.

Given the integrated nature of water resources management it has been very difficult to organise the papers into groups. An attempt has been made, in order to assist the reader in more quickly finding the papers of interest. However, in most cases, more than one subject is dealt with and the section heading under which a paper has been listed relates to its main emphasis within integrated water resources management rather than to the subject of the paper.

The volume starts with the keynote presentation of the symposium, which deals with managing aquifers to sustain irrigation with examples from Australia, India and the Philippines. The first section of the volume is on water resources availability where the focus, in the main, is on assessing the water budget of catchments and aquifers, and managing their quantitative aspects. This includes two case studies on artificial recharge. Water for food has been singled out as an individual section because of its importance at the global scale. The emphasis is on the effect of crop production on water resources and suitable methods of managing water for irrigation purposes. The next section takes a closer look at the water quality of both surface and ground waters, including saltwater intrusion problems. Floods and droughts are dealt with in the next section. Change assessment and management is a particularly timely issue. The main subjects in this section deal with the effects of water works on the streamflow and groundwater regime, changes in erosion and land cover, and climate change. A number of papers in this section propose adaptive management strategies. The following section deals with the methodological aspects of monitoring and optimisation. The monitoring studies make use of satellite data. The optimisation studies focus on the mathematical aspects of integrated water resources management. The final section on integrating water resources management presents the papers that include water demand, water allocation and policies, in addition to the other subjects dealt with in this volume.

The editors gratefully acknowledge the assistance of a large number of reviewers in bringing together this volume. Many thanks to Cate Gardner from IAHS Press for her professional approach and all the help with the processing of the manuscripts.

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(Proc. of JS.3 at the Joint IAHS & IAH Convention, Hyderabad, India, September 2009). IAHS Publ. 330, 2009.

Keynote Paper

Managing aquifer recharge and discharge to sustain irrigation livelihoods under water scarcity and climate change

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Abstract Decreasing mean annual rainfall, and the increasing rainfall intensity, temperature and evaporation, forecast for semi-arid parts of the world where water supplies are already stressed will require storage capacity to be increased or more stable resources to be harnessed to maintain security of water supplies at current levels. Managed aquifer recharge (MAR) to enhance below-ground storage of water is considered a positive contribution to stabilising drinking water supplies in cities subject to climate change. However, this paper shows that in rural irrigation areas where groundwater levels are already dropping due to an imbalance between extraction and natural recharge, unless favourable conditions permit sufficient recharge enhancement, MAR will need to be supplemented by discharge management to be successful in sustaining irrigation supplies. In fractured rock aquifers with low storage capacity, the symptoms of excessive demand are accelerated. In some cases MAR may give false hope where the benefits only accrue to the wealthiest landholders with deepest wells, or landholders closest to recharge facilities. This paper contains theoretical examples and case studies from Australia and India to illustrate a spectrum of approaches involving different contributions of recharge enhancement and discharge management to reduce groundwater deficits. A model for farmer-led groundwater demand management in the Philippines is anticipated to be effective in constraining consumption and preventing coastal saline intrusion in northern Luzon where aquifers are at an early stage of development. Similarly, models are proposed to reduce demand on aquifers that are already showing advanced symptoms of stress, while equitably supporting livelihoods at their maximum sustainable value.

Key words hydrogeology; groundwater; recharge; water supply; sustainability, irrigation

INTRODUCTION

Historically, there have been typically three stages of groundwater development for irrigation supplies in most countries. When the first irrigation wells are installed in an area, natural replenishment on an annual cycle can compensate for storage loss and lateral aquifer discharge decline has no observable effect on other groundwater users, on surface water flows or levels, or on groundwater-dependent ecosystems. The much higher production resulting from irrigation leads to a growing number of farmers installing wells and commencing irrigation, and eventually the level of development is in hydrological balance with the recharge. This second stage can occur very quickly in areas where current rainfall is less than recent or palaeoclimatic conditions. Symptoms of the second stage are evident in dryer than average years when depletion of stream flow, drying of groundwater dependent wetlands, declining inter-annual groundwater levels and possibly reduced well yields may occur. In wetter than average years the equilibrium is restored. If these symptoms are ignored and further groundwater development occurs, the system reaches a third stage where even in average years declines in storage, ecosystem health and yields are observed and wet years no longer restore hydrological equilibrium. Other indicators are declining groundwater extraction and agricultural production and incomes, noticeable interference between wells, drillers being called to deepen wells, and in some cases increasing groundwater salinity, particularly in coastal areas (Beernaerts, 2006).

At this point, if nature is left to take its course, the irrigated area and agricultural production will decline to no more than at stage two. In reality production may be reduced further, especially if groundwater has become more saline and soils more sodic. Pumping costs will be higher due to lower groundwater levels so farmers' margins will be reduced. The "tragedy of the commons" (Burke & Moench, 2001) is that the majority of small farmers who are unable to deepen their wells will have to abandon irrigation, while a few of their richer neighbours will utilise what is left of the resource. An alternative stage four is to consider whether there are opportunities to harvest other water resources either directly, or via the aquifer. Such sources may include river flows that exceed environmental flow requirements and commitments to downstream users, and new sources of water such as urban stormwater and reclaimed water. Several examples are given below.

CLIMATE CHANGE AND ADAPTIVE STRATEGIES

The climate change threat to groundwater (and dependent systems) in many regions of the world is reduced availability of groundwater, due to reduced groundwater recharge, increased demand on groundwater (due to reduced surface water supplies and increased evaporation) or groundwater contamination. A forthcoming World Bank report (SKM, 2009) reviews the state of the science of the effects of climate change on groundwater and proposes a broad range of adaptation strategies, with a focus on the developing world. Even though recharge worldwide is expected to slightly increase, there will be significant decreases in recharge in many countries. In most regions this decrease is predicted to be far more than the decrease in rainfall.

Adaptation strategies focus around five main themes: managing groundwater recharge, protection of groundwater quality, managing groundwater discharge, management of groundwater storage and managing demand for groundwater. In the first theme, the use of managed aquifer recharge (in its many and varied forms, e.g. Dillon, 2005) is strongly promoted, but not at the expense of environmental flows. It is noted that the economic feasibility of MAR varies widely, but as water availability becomes even more critical with climate change, then the economics will be improved. However, effective long-term adaptation to climate change and hydrological variability requires measures which protect or enhance groundwater recharge and manage water demand.

Adaptation to climate change can not be separated from actions to improve management and governance of water reserves (e.g. education and training, information resources, research and development, governance and institutions). In many cases, adaptations to reduce the vulnerability of groundwater dependent systems to climatic pressures are the same as those required to address non-climatic pressures, such as over-allocation or overuse of groundwater. MAR has an important role here.

IRRIGATION WITH MAR AND MANAGING DISCHARGE – CONCEPTS

Managing recharge alone without managing discharge has been applied as a successful strategy only where alternative source water supplies and capabilities to recharge are larger than groundwater deficits. Figure 1 shows that recharge enhancement can potentially be ineffective or inequitable in managing over-exploited aquifers unless done at an adequate scale, or combined with groundwater demand management. Effective MAR may also need surface water demand management.

Implementation of demand management alone is extremely difficult unless the irrigation community understands the need for this and is engaged in decision making. This is best facilitated, as will be seen later for a case study in the Philippines, at Stage 1 of development of the groundwater resource. At that stage irrigators have not committed capital to irrigation systems that are unsustainable, the concept of sharing of a public-good groundwater resource is understood, leading to a stewardship conceptualisation of rights of access to groundwater that can be embodied in groundwater management plans. Another advantage of early intervention is that groundwater dependent ecosystems can be protected. These are the first to be impacted by groundwater exploitation, well before groundwater users observe declines in yields. Community recognition of such impacts so that extraction is constrained to preserve ecosystems is highly desirable. The

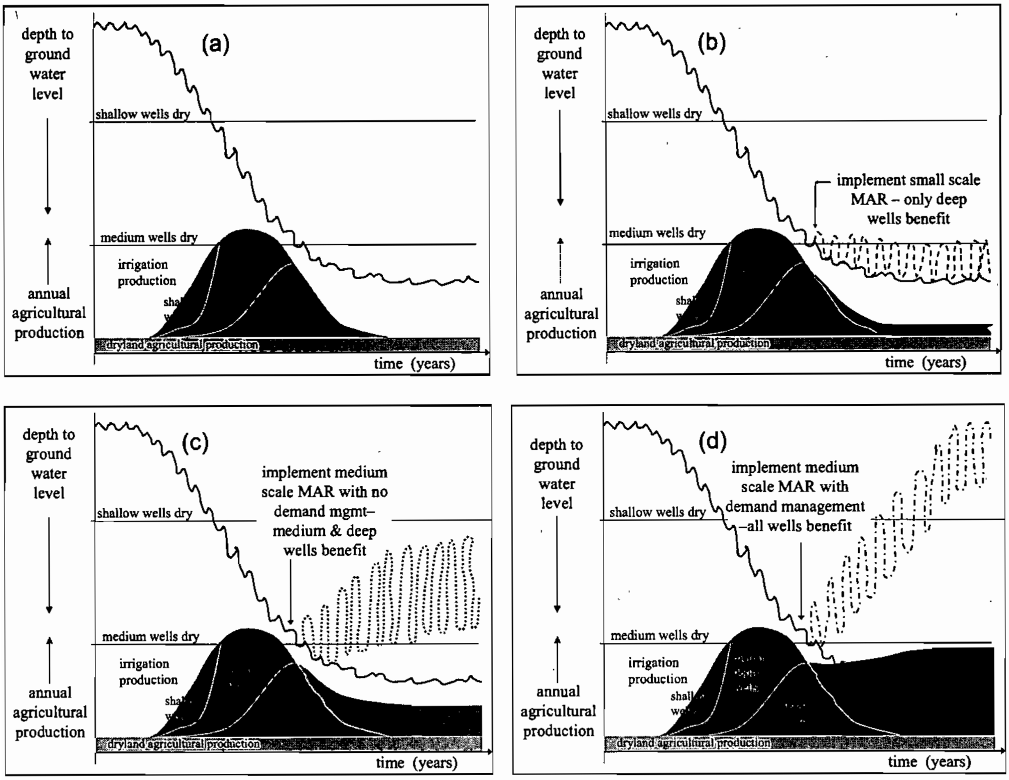

Fig. 1 Groundwater level and agricultural production history in an over-exploited aquifer. For this conceptual example, over a 30-year time scale, dryland agriculture gives way to irrigation that lowers water tables, and production from shallow, medium and deep wells successively declines until the water has been mined, forcing a return to dryland farming. Interventions have different degrees of success: (a) with no intervention the resource is depleted or salinised and production from irrigation ceases; (b) with small-scale MAR only deep wells benefit and small irrigated production continues; (c) with medium-scale MAR deep and medium wells benefit and larger production continues; and (d) with medium-scale MAR and demand management all wells benefit and substantial production is sustained.

complication of variable rainfall and recharge in a changing climate which, regardless of groundwater extraction, can impact on the health and transition of ecosystems, provides context for objectives for protection of groundwater-dependent ecosystems. For example, over time, wetlands may dry out as a result of climate change in an area, and so it is an impossible imposition on local irrigators to manage extraction so as not to impact on the wetlands.

Managed aquifer recharge can help communities to invest in demand management because it enables the minimum reduction in demand to reach a groundwater balance (assuming a surface water allocation to recharge is possible). Where the marginal cost to a community of implementing recharge enhancement projects may be smaller than the costs of foregone production, it is rational to form a collective management approach to support recharge operations such as groundwater users associations (e.g. Water Boards of Burdekin Delta, Australia; see next section). Recognising the public good aspects of groundwater, tools such as groundwater pricing have been used to pay for recharge operations, monitoring of groundwater use and levels, informing on methods to improve irrigation efficiency, and supporting investigations to determine sustainable yields and impacts on groundwater-dependent ecosystems. In many locations where groundwater irrigation increases the value of agricultural production by a factor of 5 to 50 times over that of non-irrigated land, this does give opportunities for support of investments to conserve, sustain and enhance the groundwater resource while increasing farmer incomes.

The likely effectiveness of managed aquifer recharge to address groundwater over-exploitation may be determined approximately by conceptualising the irrigation demand and aquifer properties and levels as uniform and with the same spatial extent. Assuming that the storage (S) accessible to irrigation wells in the aquifer is homogenised over the irrigation area, then if extractive discharge (D) expressed as mm/year exceeds the rate of recharge R_n (natural recharge plus deep seepage), then the average number of years (T) before yield failure occurs will be given by:

$$T = S / (D - R_n) \quad (1)$$

If managed aquifer recharge is expressed as an effective rate R_m over the irrigation area, and groundwater discharge management methods reduce discharge by D_m then the intent is to attain an equilibrium (where $T_m \leq 0$). However, less effective efforts serve to prolong the average number of years ($T_m > 0$) before yield failure occurs, as given by equation (2):

$$T_m = S / ((D - D_m) - (R_n + R_m)) \quad (2)$$

where D = discharge (pumping for drinking, irrigation, industry) (mm/year); D_m = reduction in discharge due to demand management (increased irrigation efficiency, substitution of surface supplies or restrictions on groundwater use) (mm/year); R_n = natural rate of recharge and including irrigation deep seepage (mm/year); R_m = effective rate of managed aquifer recharge (mm/year); S = storage accessible before yield failure (mm) = accessible saturated thickness × porosity; T = years until yield failure if $D > R_n$ (no failure otherwise as not over-allocated); and T_m = years until yield failure if $(D - D_m) > (R_n + R_m)$ (no failure otherwise as management is effective in sustaining yield.)

The prolonging of irrigation in an over-allocated aquifer by managing aquifer recharge and discharge is therefore given by:

$$T_m / T = 1 / (1 - (R_m + D_m) / (D - R_n)) = 1 / (1 - r - d) \quad \text{for } r + d < 1 \quad (3)$$

where $(D - R_n)$ = annual deficit = rate of over-exploitation; $r = R_m / (D - R_n)$ = proportion of deficit addressed by recharge enhancement; and $d = D_m / (D - R_n)$ = proportion of deficit addressed by discharge reduction.

Hence, there are three management scenarios for restoring an over-allocated aquifer to hydrological equilibrium:

(1) where managed aquifer recharge alone is able to overcome the deficit (i.e. $r \geq 1$);
(2) where demand reduction alone is sufficient to restore hydrologic equilibrium (i.e. $d \geq 1$); and
(3) where the combination of managing aquifer recharge and discharge is able to restore hydrological equilibrium (i.e. $r + d \geq 1$).

Consider a simple worked example of an aquifer 20-m thick with an effective porosity of 0.1 giving an accessible storage, S of 2000 mm. If the excess of demand over natural recharge replenishment $(D - R_n)$ is 100 mm/year then, from equation (1), the expected irrigation lifetime of the aquifer, T, is 20 years. To sustain the system either 100 mm of recharge enhancement, 100 mm of discharge reduction, or a combination of recharge enhancement and recharge reduction totalling 100 mm is required. The extent to which irrigated production could be prolonged by combinations of recharge enhancement and discharge management for this example are shown in Table 1.

Table 1 Effectiveness of MAR in over-exploited aquifer example (years to yield failure, T_m).

	$d = D_m / (D - R_n)$					
$r = R_m/(D - R_n)$	0	0.1	0.2	0.5	0.8	1
0	20	22	25	40	100	∞
0.1	22	25	29	50	200	∞
0.2	25	29	33	67	∞	∞
0.5	40	50	67	∞	∞	∞
0.8	100	200	∞	∞	∞	∞
1	∞	∞	∞	∞	∞	∞

EXAMPLES OF EFFORT IN REVIVING OVER-ALLOCATED AQUIFERS – WITH SPECIAL REFERENCE TO MANAGED AQUIFER RECHARGE

Case studies in Australia

Burdekin Delta, Queensland Where sugar cane is irrigated with groundwater from a shallow unconfined sand aquifer in the coastal area of the Burdekin Delta of Queensland, it was recognised in the 1960s that groundwater levels were already below sea level and saline intrusion would be inevitable without intervention. Rather than restrict irrigation, farmers and government established Water Boards that constructed channels and recharge pits and pumped water from the Burdekin River when turbidity was low. The North and South Burdekin Water Boards have achieved up to 45 million m^3/year of recharge over more than 40 years and this has maintained groundwater levels and sustained irrigation (Charlesworth *et al.*, 2002; Marchant & Bristow 2007, Dillon *et al.,* 2009) (Table 2).

Angas-Bremer rivers, South Australia In the Angas-Bremer River area of South Australia, a large viticultural area was established based on groundwater irrigation from an unconfined to a confined limestone aquifer. As irrigation expanded groundwater salinity rose to more than 2000 mg/L in places. In 1970 one farmer began pumping water from the Bremer River (an ephemeral stream) into his well when stream salinity and turbidity were low. This halved the average salinity of water recovered from the well and by 1992 there were 30 wells recharging a total of 2.4 million m^3/year (Gerges *et al.*, 2002).. The systems were run by the farmers with supporting technical advice from the state government (Department of Mines and Energy). However, the volume that could be recharged was inadequate to sustain irrigation and groundwater levels and yields were in decline. The government intervened by providing a pipeline from the River Murray, a perennial stream, and this now contributes 90% of the irrigation water used in this area.

Northern Adelaide Plains, South Australia On the Northern Adelaide Plains, South Australia, irrigated horticulture has relied on confined limestone aquifers. Expansion of irrigation in the 1960s and 1970s ultimately produced a drawdown cone 70 m deep, centred on Virginia, which recovered to a smaller degree in successive winters, leading to dewatering of the overlying aquitard. The government's response was to licence groundwater extraction from 1976, penalising use in excess of licensed amounts, thereby reducing use from 25 to 18 million m^3/year. This did not receive popular support, compliance was less than ideal and administration was difficult. In the 1990s the government supported a plan to establish a water reclamation plant at Bolivar wastewater treatment plant that receives more than half of the city of Adelaide's sewage, and to reticulate the reclaimed water to the irrigation area (Bosher *et al.*, 1998) to support growers while reducing demand on the aquifer to a sustainable level. In addition, a trial was conducted to test the concept of recharging the aquifer directly with reclaimed water, so that effluent that discharged to the sea over winter could be harvested and stored in the aquifer to further expand irrigation (Dillon *et al.*, 2003). This reclaimed water aquifer storage and recovery (ASR) trial proved successful and, when the irrigation demand expands, this approach is ready for implementation. In addition, urban stormwater ASR has also taken place, primarily for landscape irrigation and industrial water supply in the urban fringes of the horticultural area. Government support for these ASR projects accounted for their net benefits on sustainable yield in the irrigation area (Table 2).

Case studies in India

Groundwater from hand-dug wells and boreholes is widely used to irrigate small farms in rural India. Farmers traditionally practiced rain-fed agriculture, but changed to groundwater with the advent of rural electrification to power submersible pumps, market integration and cheap credit, particularly in the 1980s and 1990s. This led to overexploitation and falling groundwater levels (exacerbated by deeper drilling and periods of drought), and irrigated agriculture became unsustainable. By the mid-1990s, agricultural production was in decline in some areas. Managed aquifer recharge was seen as a resolution to this dilemma and resulted in the proliferation of check dams and other recharge structures constructed to capture monsoon rainfall in the depleted basement aquifers that have low porosity and storage. The efficacy and impacts of this approach require data for assessment (e.g. Dillon, 1983) and have been tested since in many studies, including a detailed three-year study (2002–2005) at selected research sites (Gale *et al.*, 2006) including the two following sites.

Satlasana, Gujarat The site at Satlasana in Gujarat state, India, in the foothills of the Aravalli Hills, is underlain by shallow weathered and fractured granitic rocks, overlain along the valley floors by 15–20 m of alluvium washed down from the hillsides. Annually, 690 mm of rain falls for a total of 30–35 days between late June and the end of September. Unrestricted abstraction from 85 boreholes and numerous wells in an area of about 20 km^2 supports agriculture for a population of about 4200, but agricultural production decreases in drought periods when irrigated areas decline in response to lack of groundwater. Subsequently, construction of four major check dams combined with myriad other rainwater harvesting interventions has resulted in the capture of an estimated additional 3 to 13% recharge, although this is only equivalent to 1% of rainfall. (Average annual rainfall for the period monitored, 440 mm, was well below mean rainfall.) The high permeability of the overlying alluvial sediment ensures rapid infiltration of flood waters in only a few days. Before recharge interventions, the percentage irrigated area dropped from about 80% to 30% during drought periods, but with recharge interventions the irrigated areas were maintained. These figures relate to areas (totalling 28 ha) in the immediate vicinity of the recharge structures as the nature of the aquifers limits the impacts to a few hundred metres.

Kodangipalayam, Tamil Nadu A comparative study of the effectiveness of MAR was carried out near Coimbatore in Tamil Nadu state, India. The Kodangipalayam micro-watershed (5 km^2) has a similar population of about 5700, and is underlain by shallow weathered crystalline hard rocks which have relatively low storage capacity. The average rainfall of 530 mm arrives in a dual-season monsoon in June–September and October–December, but during the study period, 728 mm/year were recorded. Groundwater is abstracted from farm boreholes and hand-dug wells and the area and type of crops grown is determined by the amount of recharge received during the monsoon. Where check dams and other impoundments capture and recharge additional water, farming is more sustainable. The impact of the structures is restricted to a few hundred metres due to the low transmissivity of the aquifer but, during the period monitored, an additional volume of water was recharged equivalent to 23% of natural recharge for the area (1.4% of rainfall). Transport of silt reduces the infiltration rate of water captured in check dams and extends the period of additional recharge. Over the preceding 10 years, the proportion of irrigated land reduced from 67% to 41% and, except in pockets benefiting from the effects of the recharge structures, less water-intensive fodder crops are grown. That is, groundwater availability continues to reduce in areas other than in the immediate vicinity of recharge structures.

Maheshwaram, Andhra Pradesh Natural recharge in the Maheshwaram watershed of India, a 53 km^2 area situated on granitic terrain, is supplemented by 13 tank structures distributed across the watershed. Water balance studies performed by the Indo-French Centre for Groundwater Research (Dewandel *et al.*, 2007) show that, in a normal monsoon year, the 190 mm of groundwater irrigation is largely offset by recharge of 170 mm comprising natural (diffuse and localized) recharge and irrigation return flows of similar magnitude. The annual average net groundwater balance deficit is estimated to be 20 mm, which is three times greater than the contribution from MAR structures (Table 2). Under the current pumping regime, in an aquifer with a low storativity and shallow total depth, it is estimated that half the pumping wells would run dry within three years with accompanying serious socio-economic consequences. Sustainability could be achieved by combining the construction of new rain-harvesting structures with changes in cropping pattern and improvements in irrigation techniques.

Table 2 demonstrates that groundwater management is effective at some sites in addressing groundwater deficits. Two types of groundwater irrigation area are shown to be amenable to augmentation in the event of over-exploitation of aquifers. Those on the fringes of urban areas have the benefit of a nearby source of storm water and/or treated sewage which may be recycled, either directly or via aquifers, through managed aquifer recharge (e.g. Northern Adelaide Plains).

Table 2 Case studies of groundwater irrigation systems that have mostly been overallocated and actions taken to support ongoing irrigation, including the role of managed aquifer recharge and discharge reduction.

Area	Rain-fall (mm/ year)	Groundwater irrigation (mm/year) [Area (ha)]	Implied natural recharge* (mm/year)	Deficit (mm/ year)	Deficit as % of irrigation	% Deficit met by MAR	% Deficit met by demand manage-ment	MAR opportunities and solutions	Reference
Burdekin Delta, Queensland, Australia	1000	570 [42 000]	345	225	39%	100%	0%	Recharge via pits 75 mm, and in-channel seepage 150 mm	Marchant & Bristow (2007)
Angas-Bremer area, SA, Australia	380	260 [6 800]	22	238	92%	15%	85%	30 recharge wells 35 mm. Balance of irrig demand is now met by a new pipeline	Muller (2002); Gerges *et al.* (2002)
Northern Adelaide Plains, SA, Australia	440	400 [4 500]	140	260	65%	33%	>100%	4 GL stormwater ASR (89 mm) plus future reclaimed water ASR. 22 GL recycled water pipeline enabled expansion of irrigation	Northern Adelaide and Barossa Catchment Water Management Board (2000)
Satlasana, Gujarat, India	440	500 [28]	30–60	450	90%	60%	0%	Check dams, field bunds Impact on small local area only	Gale *et al.* (2006)
Kodangipalayam, Tamil Nadu, India	728	500 [500]	41–47	460	92%	2%	0%	Check dams. Impact averaged over catchment but benefits only local to dams	Gale *et al.* (2006)
Maheshwaram, Andhra Pradesh India	750	190 [5 300]	170	20	11%	33 %	0%	Percolation tanks	Dewandel *et al.* (2007)
Burgos, Ilocos Norte, Philippines	2000	320 [43]	340	0	0%	0%	0%	Groundwater extraction to be managed by farmers with govt technical support. Irrigation of dry season garlic with good mulching.	Contreras & Dillon (2008)

*This includes deep seepage under irrigation.

In these areas, if the urban water supply is secure, then the effects of climate change can be buffered by these alternative sources. Depleted aquifers may provide the most efficient storage for recycled water in places where evaporation rates are high. The second category is where there are nearby surface water systems with surplus capacity to supply water while still allowing for environmental needs. Here conjunctive supplies of surface water and groundwater are possible (e.g. Angas-Bremer) or recharge can be enhanced (e.g. Burdekin). For groundwater irrigation locations without the benefit of urban areas or perennial streams, such as in Indian examples and on the coastal plains of northern Luzon in the Philippines, alternatives are needed to manage the effects of climate change and an increasing demand for groundwater. Management strategies may differ depending on the stage of groundwater development.

FARMER-LED GROUNDWATER MANAGEMENT IN ILOCOS NORTE, PHILIPPINES

An ACIAR project entitled "Enhancing agricultural production in the Philippines by sustainable use of shallow groundwater" (Contreras & Dillon, 2008) has focused on two pilot sites within the neighbouring municipalities of Pasuquin and Burgos, Ilocos Norte in the northwestern tip of Luzon, Philippines. The project investigations indicated that in both pilot sites the use of shallow groundwater is generally still at a sustainable level although farmers raised their lack of awareness as to what is happening to the resource. The farmers revealed that sometimes there was noticeable competition in groundwater extraction, particularly during the peak of the dry season, but such problems can be addressed through coordinating pumping schedules. However, with the continued utilization of shallow groundwater for agricultural production there is a need to ensure that future increase in groundwater use will still be sustainable. The challenge is to increase production by the use of groundwater without encountering these constraints.

The project has been successful in demonstrating to the local policy/decision makers and farmers the value of groundwater in agricultural production and in giving a preliminary indication of the constraints on the expansion of groundwater use. Realizing the current and future scenarios of groundwater utilization, they agreed that there is a need to introduce local policies that will be anchored on national policies to protect and manage our groundwater resource. A "Covenant of Support" to pursue this effort was promulgated by the farmers and local government leaders when the results of the project were presented to them and they were consulted on what should be done. This is a significant project milestone and supports one of the project objectives (i.e. to implement appropriate management strategies at the pilot site in cooperation with local government units, to enhance sustainable production). However, this could only be fully realized if the farmers themselves directly participate in the assessment and management of the resource. As such, they should be empowered and developed as a critical sector to influence local policies toward sustainable groundwater management. This could be achieved by pursuing the concept of *farmer-managed groundwater systems* to continue the momentum gained by the ACIAR project. The concept (started in Andhra Pradesh, India; Mani, 2008; FAO, 2008) intends to increase the level of awareness and understanding of farmers about groundwater and its occurrence, cropping pattern development, and other technological concepts leading to more sustainable management of the groundwater resource. Due to the local costs of managed aquifer recharge this is only a contingency option in the event that climate change reduces recharge or if initial estimates exceed actual sustainable yield.

The concept of *farmer-managed groundwater systems* was introduced in late 2007 through farmer workshops. The local government unit (LGU) officials, academic representatives, and farmers in the two areas, Pasuquin and Burgos, warmly received the concept. Capitalizing on the momentum gained by the project, taking the successful experience of establishing farmer-managed groundwater systems in India, and with the commitment of support by the LGUs and farmers, the current work aims to ensure the sustainable use of shallow groundwater for increased agricultural production through the empowerment of farmers and creation of an enabling environment on groundwater management in the pilot sites.

This aims to:

- enhance the capacity and skills of farming communities in understanding the groundwater systems and collecting basic field data;
- develop and implement crop plans and related management strategies through the involvement and engagement of farmers; and
- communicate and promote results to facilitate broader adoption and strengthen advocacy on the concept of *farmer-managed groundwater systems* (FMGWS) in the Philippines.

As a strategy of implementation, the concept of FMGWS was introduced through the establishment of a Farmer Water School (FWS) in the pilot sites (i.e. also following the Indian model). A training of trainers (TOT) was held in November 2008 to develop the agricultural technicians in Ilocos Norte as potential trainers who will empower farmers and groundwater users with the required knowledge and skills to protect and manage the resource (Fig. 2). Training modules that suit the Philippine setting and culture were prepared. In broader terms, these modules consist of: (1) the human dimension of groundwater management; (2) basic concepts on groundwater, its occurrence and sustainable use; and (3) crop planning and formulating of appropriate groundwater management (e.g. effective MAR and protection of coastal aquifers from pollution). With these modules, several other sub-modules were developed that include the facilitation and presentation skills to enable them to conduct the FWS in an effective and interesting way for farmers. Modules are translated into the local dialect so that farmers can easily understand each topic.

The training of trainers (TOT) is concerned with imparting practical knowledge and skills in measuring recharge and extraction to estimate groundwater balance which will serve as a basis for making informed decisions on crop planning and formulating groundwater management strategies. Subsequently, such knowledge and skills will be conveyed to farmers through the FWS. The TOT also emphasized the need for collective and unified actions and decisions to achieve effective and sustainable groundwater management. A highlight of the TOT was the field activities wherein the trainees had the experience of actual measurement of recharge and discharge (Fig. 2), among others.

Efforts of empowering the farmers have started as orientation and briefing of 50 participants of FWS in January 2009, and a schedule of twice-monthly FWS sessions for farmers. Each session consists of a topic on hydro-ecosystem analysis, a special topic (e.g. knowing weather and climate), and group dynamics. With the great interest of farmers during the initial session, it is expected that the knowledge they will gain will be put into action to achieve an effective *farmer-managed groundwater system* and contribute in developing a strategy to manage the groundwater resource in the Philippines.

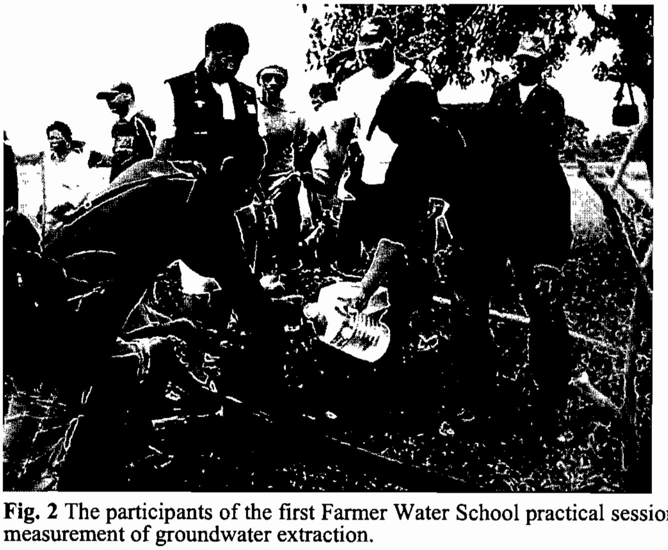

Fig. 2 The participants of the first Farmer Water School practical session on operating the pump for the measurement of groundwater extraction.

CHARACTERISTICS OF GROUNDWATER MANAGEMENT STRATEGIES IN RELATION TO STRESS

Unlike Ilocos Norte in the Philippines, which is at Stage 1 of groundwater resource development, in many parts of Australia, India and elsewhere, groundwater is already over-allocated and competition for a scarce resource makes collegiate stewardship approaches to groundwater management very difficult to implement. The water resources planning framework proposed for MAR in Australia (Ward & Dillon, 2009) follows an existing system of entitlements, allocations and use conditions for surface water and groundwater. A regulator for each catchment or groundwater system assigns an *entitlement* to users of water in that system after allowing first for an environmental entitlement. In dry years there may be less water available than can meet each shareholder's entitlement, and the regulator determines the *allocation* within that year or period, normally as a specified percentage of each entitlement. This ensures that each user shares uniformly in the restrictions imposed by dry years or years when groundwater storages are low. Allocations vary from year to year, and even from month to month in some river systems, whereas groundwater allocations are often set over longer time scales, typically five years. Finally, *conditions of use* may also be imposed, for example to foster increased irrigation efficiency, or to give priority to say drinking water supplies over other uses when allocations are very low. An additional ingredient in these water allocation arrangements is the ability for trading of entitlements and allocations among existing and new water users within the catchment or groundwater system. This means that one user can sell part or all of their perennial entitlement or their current year's allocation to another user. This assists the use of water for its highest-valued uses within a catchment or basin.

There are four discrete components for any MAR project which warrant separate entitlement, allocation and use conditions to allow effective management of one or many MAR projects within a catchment or aquifer: (1) water capture and harvesting; (2) recharge; (3) recovery; and (4) use. Table 3 shows a robust separation of water rights for discrete elements of a MAR system as a possible policy framework for addressing MAR on a sustainable basis. For comprehensive details, see Ward & Dillon (2009), or for an overview, see Dillon *et al.* (2009). For groundwater systems without managed aquifer recharge only the recovery and use columns are required.

For the first MAR projects in any area it is not essential to have clearly defined water allocation policies in relation to MAR. This parallels the first stage of groundwater development, where there is no need to restrict wells or extraction so long as the resource can meet demand without adverse outcomes for users or the environment. Ultimately, however, the time may come

Table 3 A proposed policy framework based on robust separation of water rights for discrete elements of a MAR system.

Governance instrument	MAR component: Water capture and harvesting	Recharge	Recovery	Use
Entitlement	Unit share in river, stormwater or effluent consumptive pool (i.e. excess to environmental requirements)	Unit share of aquifer's finite storage capacity	(Tradable) extraction share a function of managed recharge	
Periodic allocation	Periodic (usually annual) allocation rules based on a water plan. Potential for additional stormwater or effluent offsets	Annual right to raise the water table subject to ambient rainfall and total abstraction	Extraction volume contingent on ambient conditions, natural recharge and spatial constraints	
Obligations and conditions	Third party rights of access to infrastructure for use of stormwater and sewage	Requirement not to interfere with entitlements of other water users and water bankers	Existing licence may need to be converted to compatible entitlement to extract (unit share)	Water use licence subject to regional obligations and conditions for use

when management intervention in a system is needed to prevent problems, and wise resource managers will have prepared policies and plans in consultation with groundwater users to address this eventuality.

CONCLUSIONS

Relatively few sites had sufficient information on groundwater use or on groundwater declines and specific yields to enable firm estimates of groundwater deficits. Where such data were available and MAR was used to reduce deficits, its contribution ranged from 2% to 100% of deficit. That is for sites where high infiltration rates could be achieved and there was an abundant source of water (e.g. Burdekin) the deficit could be met by MAR alone without relying on demand management. In hard-rock catchments with limited ability to capture and store water, MAR could only address a small fraction of the deficit. As indicated in Fig. 1 and Table 1, in such systems extraction at current rates cannot be sustained, and continued irrigation will depend on accessing alternative sources of water for direct irrigation or for MAR. At some sites, such sources of water have been found in the form of recycled water from a nearby city, and irrigation has expanded beyond that which could have been sustained by the aquifer alone.

A promising model of farmer-led groundwater management is being tested in the Philippines to avoid over-extraction of water through careful stewardship of the resource in combination with water-efficient farming practices. Where it is too late to apply this method, such as in already over-allocated aquifers, a holistic framework for management of recharge and discharge in a market trading framework has been proposed in Australia, but is yet to be tested. Success with management will depend on both technical and social criteria, and the capability to enhance recharge and reduce discharge and their relative costs. Effective management will require measurement of groundwater storage and measurement, or at least indicators, of groundwater use. Access to good groundwater technical investigations skills and data will be valuable.

MAR can be a vehicle to assist in implementing economic and other policy instruments where demand needs to be reduced. It may require new regulatory issues to be addressed in a holistic way, e.g. ownership of stored water. Application of MAR opens the way for more sustainable and economic conjunctive use of surface water and groundwater resources, but, as stated above, its ability to augment the resource will vary among catchments. It is not a panacea for water shortage, and every situation demands adaptation to suit local natural resources and societal needs. IAH is working with UNESCO under the International Hydrological Programme to inform and educate on recharge enhancement so that all new projects will be sustainable and that expectations of MAR will be closely related to its potential capabilities within the catchment.

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1 Water Resources Availability

Simulation of the proportion and interaction of surface and groundwater resources

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Abstract The present article deals with the simulation of rainfall–runoff using the SAC-SMA and BROOK models. It has been helpful for this purpose to follow the proportions of simulated runoff components, i.e. of streamflow (direct runoff, interflow and baseflow), and the available examples illustrate the significant contribution of simulation. The length of the evaluated time series is significant as it permits us to follow the influence of the vegetation development, which affects evapotranspiration demand. The influence of the diverse hydrogeological conditions in the area – various crystalline and sedimentary bedrocks – is important. Comparison of water levels monitored in boreholes and/or of spring discharge fluctuations, with the variability of simulated water storage in the basin, may facilitate the identification of the parameters for baseflow simulation.

Key words rainfall–runoff simulation; runoff components; evapotranspiration demand; groundwater components; Czech Republic

INTRODUCTION

Evaluation of interactions of surface water and groundwater storage in streamflow is a relatively complex process because in many cases there are several reasons for the appearance of changes in this relationship. Besides natural influences, artificial events may also contribute to this phenomenon, especially in more recent periods, and complicate the evaluation, i.e. determination of which tool is most appropriate for the evaluation in the given conditions.

Simulation of the rainfall–runoff process has been extensively used in our experiments, and provides the possibility to discover the frequent changes of flows in the long time series of relatively small streams. The change often appears to be due to withdrawals of groundwater for drinking water needs and the evaluation is used to ascertain the alteration in the water regime. The simulation also supports the efforts to follow the effects of vegetation development, which influences evapotranspiration demands and consequently the uptake of groundwater, as recently reported (Buchtele & Tesar, 2008).

A further natural reason for long-term fluctuation in water resources could be the variation of the solar radiation on the scale of tens of years, which affects both evapotranspiration and precipitation fluctuations. This may be significant considering that recent research indicates the similarity of that phenomenon with sun spot frequencies (Beer, 2005). The distinct consequences of the oscillations appearing in atmospheric processes, as usually reported in connection with El Niño or Atlantic Multidecadal Oscillation (Wikipedia, 2009), could be more obvious in water resources, the processes of which have further significant accumulation effects.

METHODOLOGY

For simulation, daily time series have been used. In most cases the series are 30–40 years long. For the Czech part of the Labe River basin, a series a bit longer than 100 years was available for the modelling of rainfall–runoff processes. These are long data sets in which it is expected that some trends in runoff could appear. The weekly measurements of the groundwater were used: the water level at the boreholes and the runoff of the springs. These series have been compared with the simulation of groundwater storage using a rainfall–runoff model in two basins, also having series 40 years long. In the Svratka River basin, near to Banin, groundwater level observations are available for about 100 years.

For simulation of the rainfall–runoff process, the conceptual model SAC-SMA (Burnash, 1995) has been mostly used. It has been calibrated for the basins using the period that is probably without artificial changes, as presented in Table 1, which includes some simulation outputs.

The simulation using the SAC-SMA model provides as outputs the contribution of groundwater storage to the whole flow in the stream. It may facilitate the judgement of the interaction between the surface water and sub-surface water. This model generates, besides surface- and interflow components, three other components of the groundwater volume: Lower Zone Tension Water Content (LZTWC), Lower Zone Free Supplemental Content (LZFSC) and Lower Zone Free Primary Content (LZFPC).

For some components the simulations were also prepared using the physically-based BROOK model (Federer, 1993). Amongst the outputs of this model is evapotranspiration, presented as the process which consists of several components: transpiration (TRAN), rain interception (IRVP), snow interception (ISVP), snow evaporation (SNVP) and soil evaporation (SLVP). It may be useful as the subsurface water is the source for the evapotranspiration.

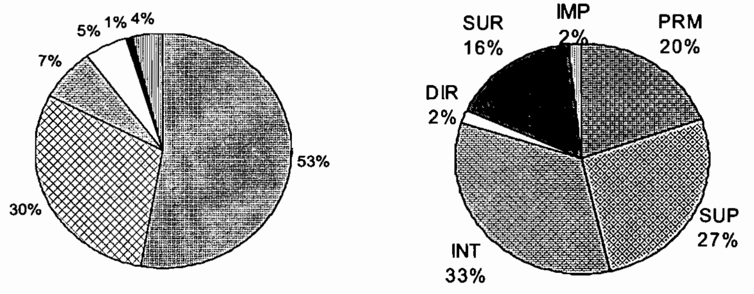

Fig. 1 Proportion of runoff components in the Pilar basin with sedimentary base (on the left) and in the Raztoka basin with the flysh base (on the right).

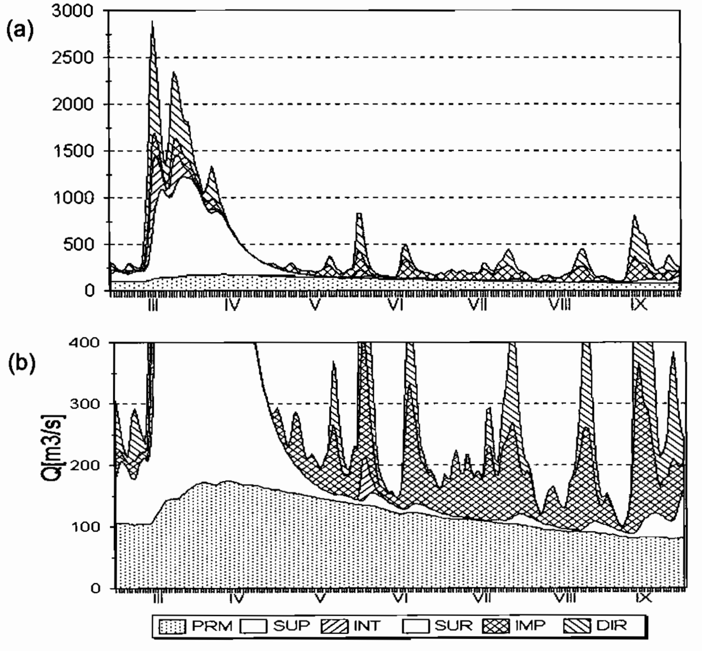

Fig. 2 (a) Simulated runoff components of the Czech part of the Labe River basin in 1940 – whole range of the flood flows ($m^3 s^{-1}$). (b) Simulated runoff components of the Czech part of Labe River basin in 1940 – bottom part of the hydrograph: PRM – primary base flow; SUP – seasonal base flow; INT – interflow; SUR – surface flow; IMP – flow from impervious areas; DIR – direct flow.

RESULTS

Runoff components

The simulation of the rainfall–runoff process provides as the outputs the runoff components, which may enable the envisaged judgement. Figure 1 provides the results of the SAC-SMA model and explains the runoff components of two basins with the diverse hydrogeological conditions, which illustrates the significant portion of groundwater in the runoff. It is significant not only in the Pilar basin (932 km^2) with large sedimentary bedrock areas, but also in the hilly Raztoka basin (2.1 km^2) with flysh bedrocks. Two baseflow components should be recognised, and are the outputs of zones in the SAC-SMA model: LZFPC and LZFSC, i.e. primary baseflow (PRM) and supplementary baseflow (SUP), which is frequently seasonal, associated with groundwater storage.

The continuance of the runoff components during the period of one large spring flood and the following low flows in the summer, are presented in Fig. 2. This example is from the Czech part of Labe River basin (51 103 km^2), in which both hilly regions with the crystalline bedrock and Cretaceous areas are situated. Figure 2(a) shows the whole range of the flood flows, while Fig. 2(b) provides more detail of the bottom part of the hydrograph. Both figures show that the baseflow is an important part of the discharge and other results confirm that this is valid even from hilly regions and, of course, during rainless periods. In this whole context it could be considered as one favourable fact that the proportion of groundwater volumes to other forms of runoff is characteristic of most basins in this region.

The variability in the long-term time series

Further evidence of the variability of groundwater storage is shown in Fig. 3, where two long time series are presented. The upper graph represents the simulated water volume in the LZFPC (Lower Zone Free Primary Content) of the SAC-SMA model, which was implemented for the Czech

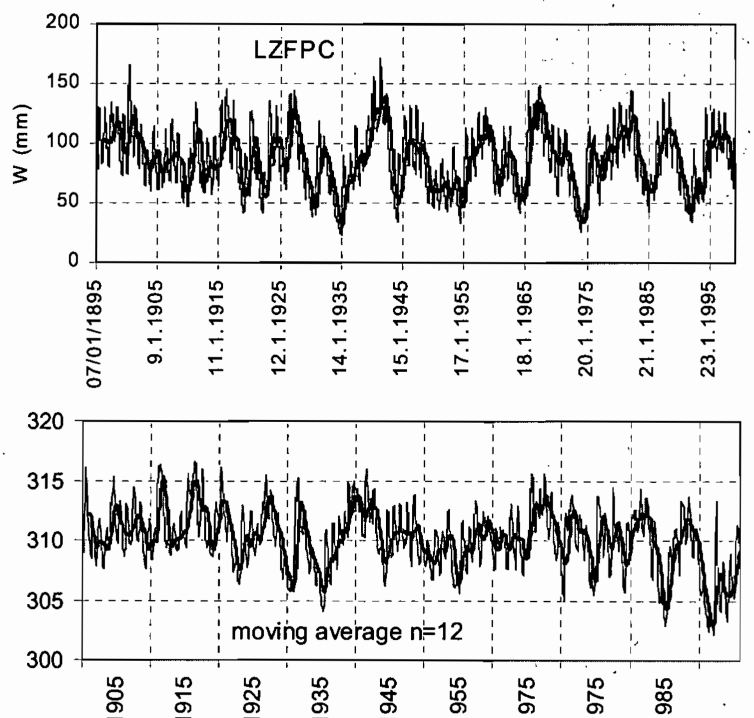

Fig. 3 Comparison of simulated groundwater storage in the zone LZFPC in the Labe River basin (upper part) and observed water level of the borehole in the Banin area (bottom part).

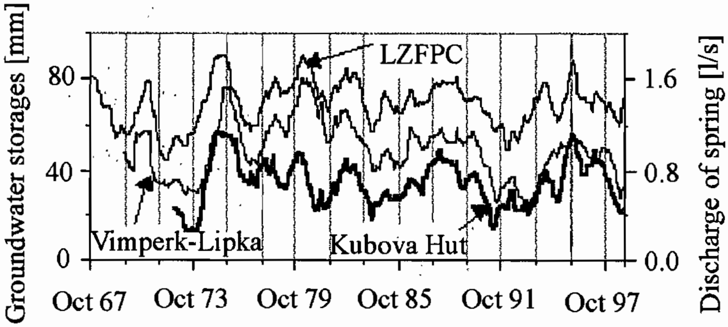

Fig. 4 Simulated groundwater storage and outflow of two springs in the Vltava basin.

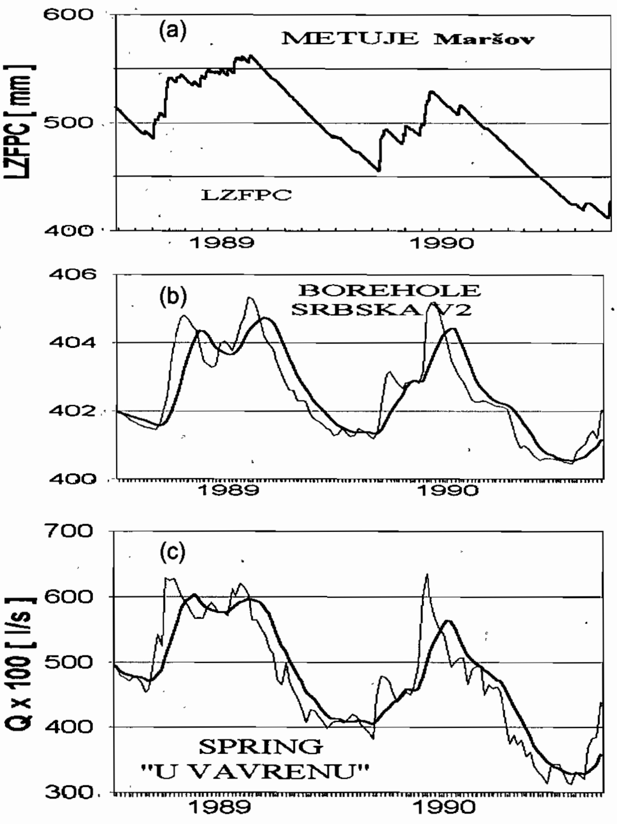

Fig. 5 (a) Simulated groundwater storage LZFPC, (b) observed water levels in borehole, and (c) outflow from the spring "U Vavrenu" in Metuje River catchment.

Labe River. The bottom graph presents the water level of the borehole located in one spring region of the Cretaceous area, Banin, situated outside the Labe River basin, but not far from its border. Similar fluctuation seems to confirm partly the existence of the visible fluctuation and partly the credibility of each of these quantities.

The simulated groundwater volumes in LZFPC and the outflows of two springs are also presented in Fig. 4 to show the situation in the upper part of the Vltava River basin (176 km^2) with crystalline bedrocks. This example shows that even in such conditions the groundwater storage exhibits a significant part of water balance, and also with some oscillation.

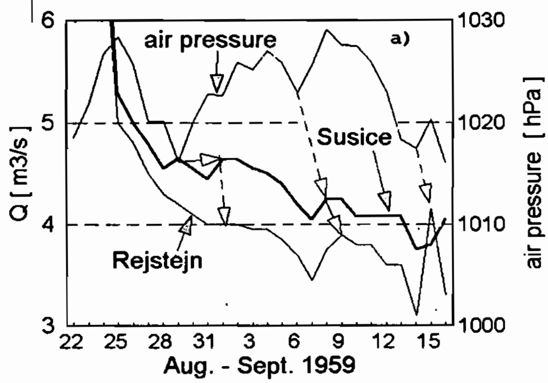

Fig. 6 Discharge increases in the Otava River at two closure profiles, Rejstejn and Susice, during one rainless period due to the decreases of air pressure.

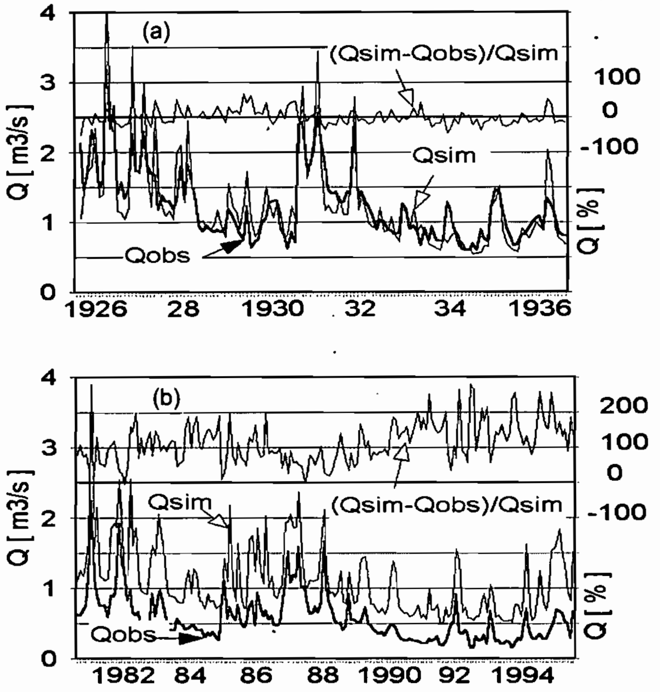

Fig. 7 The discharge decrease of Svratka River due to uptake of groundwater in the Banin spring region.

The shifts in the parameters of the model

The preceding presentation of simulations seems to confirm that tools such as rainfall–runoff modelling are useful for the evaluation of the groundwater influence on the flows in the streams. Figure 5 provides a comparison of simulated groundwater storage, with the observed water levels in the boreholes and the outflows of one spring in the Metuje River (94 km^2), which shows that this seasonal scale for the results shows reasonable coincidence. It offers the possibility for improved assessment of the relevant model parameters.

The identification of the parameters in the model could be meaningful in other circumstances. Figure 6 shows that the decrease of the flows may be interrupted during the rainless period by a decrease of air pressure, and that may affect the parameter estimation for the baseflow simulation. Figure 6 shows results from two hilly Otava River basins: first the closure profile Rejstejn (335 km^2) and second the closing profile Susice (536 km^2).

Streamflow changes by the uptake of groundwater and by evapotranspiration

Conspicuous change in the streamflows occurs frequently due to the uptake of groundwater for drinking water. Figure 7 presents the situation at the Svratka River (251 km^2) near the borehole Banin displayed in Fig. 3, in which the long-term tendency can be noticed. The marked decrease of river flows is apparent in Fig. 7, which presents firstly the period before the uptakes, used for the calibration of the SAC-SMA model, and secondly, the time interval with the affected flows, both presented in monthly series.

The groundwater components presented in Fig. 8 illustrate, in the upper part of the Labe River, the possibility of diversity, which may appear in the regimes of the extraordinary deep and long hydrological deficits and agricultural droughts. They are represented by the deficits in LZFPC in the first case and by the minimum in the LZTWC in the second case. The simulated course of groundwater storage plotted at bottom part of Fig. 8 presents results for the afforested Liz basin (0.99 km^2). It presents groundwater as the source of water for the evapotranspiration and it illustrates the possible abrupt changes of water storage.

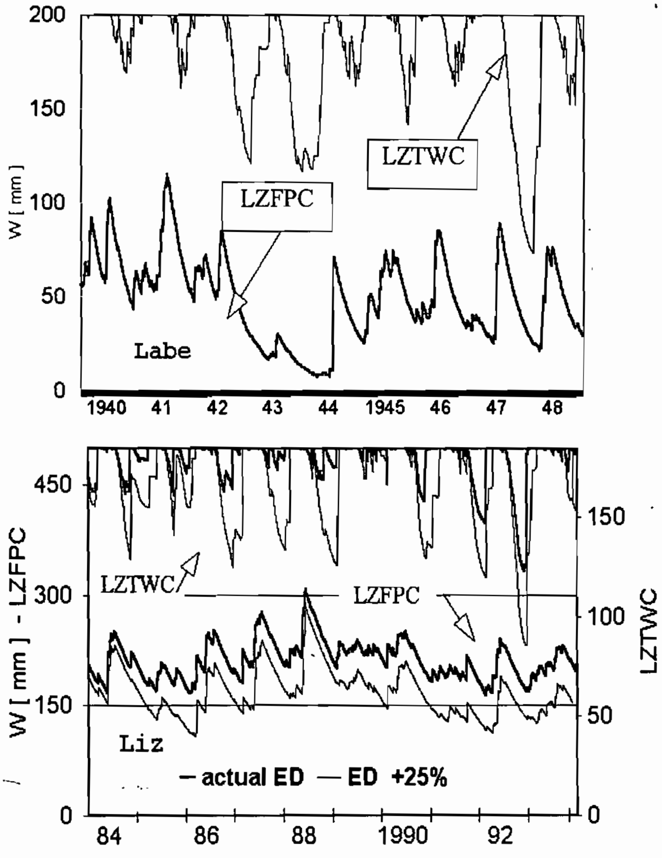

Fig. 8 Simulated tension and free soil water in the Labe basin with hydrological and agrometeorological droughts (upper part) and in the Liz basin during actual evapotranspiration and for its increase by 25%.

CONCLUSIONS

The simulation of the rainfall–runoff process has been helpful in the intention to follow the proportions of simulated runoff components; i.e. of streamflow (direct runoff, interflow and baseflow) and also for the intention to assess the interaction of surface water and groundwater in diverse natural conditions. The interactions are usually influenced by natural variability of several components of the water balance.

As the foregoing simulation also supports the efforts to follow the effects of vegetation development, another natural cause for long-term fluctuation in water resources could be the changeable solar radiation on the scale of tens of years, which might affect both evapotranspiration and precipitation variability.

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Groundwater development in Fergana Valley: the adaptation strategy for changed water management in Syrdarya basin

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Abstract During the last decade, the competition for water between the hydropower-oriented upstream and irrigated agriculture-centred downstream in the Syrdarya River basin, Central Asia, has significantly increased. Since 1993, 2–3 km^3 of winter flows from hydropower generation in the upstream have flowed annually into the saline depression of Arnasai located in the midstream. This results in much less water being available for irrigation during summer. Groundwater development modelling conducted for one of the Fergana Valley's aquifers suggests that temporary storage of winter flows in the aquifer – "water banking" – could be an effective adaptive strategy to optimize water management in the basin. The study concludes that a shift from canal to groundwater irrigation, combined with winter-flow banking can effectively reduce the upstream–downstream pressures and ensure improved water supply for downstream water uses during summer time.

Key words groundwater development; artificial recharge; river basin management; Syrdarya River, Central Asia

INTRODUCTION

Population growth and increasing demand for food and energy affects the priorities of the water use in upper and lower parts of river basins. In the upper parts, often rich with energy potential, winter water releases are needed for hydropower generation. Water availability is limited in summer in middle reaches with considerable areas of land suitable for irrigation. In the lower reaches, affected by multiple upstream impacts, water for environment purposes is often in pressing need. The growth of pressure on water resources complicates the trade-offs between management strategies. Hence, identifying the consequences of implementing these strategies and adaptation to changes in water management become very important.

The Syrdarya is a typical example of a river basin with contradictory water use requirements between its upper and lower parts. The natural flow of the river with maximum discharges in summer was favourable for irrigated agriculture practiced over 1.20×10^6 ha (CARE WEIS, 2008a). The ratio of summer to winter flow at the inflow to the Fergana Valley node under natural conditions was in the range of 2–5. After the construction of in-channel reservoirs in the 1970s, the river flow was fully subscribed to irrigation. The area of irrigated land increased from 1.2×10^6 to 3.36×10^6 ha, of which over 1.0×10^6 ha were in the middle reaches (CARE WEIS, 2008a). The ratio of summer to winter flow was still around 2–3. Since winter 1992–1993, due to increased demand for power, the operation of the upstream Toktogul Reservoir shifted from irrigation to hydropower generation mode. The ratio of summer to winter flow then dropped below 1.

The river inflow to the Fergana Valley in winter has reached 8.2–8.5 km^3 against 4.5–6.0 km^3 before 1992. The midstream reservoirs, the Kairakum and the Chardara, are full to the beginning of winter and have no free capacity to accumulate the excessive hydropower releases from the upstream Toktogul Reservoir. Moreover, the water in the downstream river channel has ice cover in winter. Mustafaev *et al.* (2006) stated that, under such conditions, winter flow of 2.0–3.0 km^3 was discharged into the saline depression called Arnasai, which at present has a surface area of over 3000 km^2 and a volume over 40 km^3. It was hypothesized that groundwater development for irrigation by banking (temporarily storing) the winter flow in the aquifers of the Fergana Valley could represent a feasible adaptive strategy to the new water management in the basin. The objective of this paper is two-fold: (a) to determine the potential of the Fergana Valley to regulate the winter flow of the Syrdarya River; and (b) to simulate groundwater development for irrigation and banking the winter flow in one of the 18 aquifers of the Fergana Valley. The Sokh aquifer having a high potential for development and water banking was selected as a pilot study area for groundwater modelling.

STUDY AREA

The Syrdarya River is formed in the Fergana Valley by the confluence of Naryn and Karadarya. It has a long-term average annual flow of 26.3 km^3 at the valley outlet, of which 55% is contributed by the Naryn River. Fergana Valley – a massive depression in the Tien Shan Mountains – is composed by sediments from the Naryn, Karadarya and the Syrdarya rivers. Secondary geological formations have accumulated due to the smaller rivers that flow into the valley from the mountains, which contribute to the coarse strata on the alluvial fans and now bring snowmelt mudflows into the valley. Sokh aquifer, formed in the fan of one of the small rivers, has a high potential for recharge (Fig. 1).

At the head of the fan of the Sokh River, 120-m thick shingle deposits with inclusions of boulders and sand-gravel filler occur. The same deposits form the water bearing strata in the central part of the fan. In the lower part, the shingle deposits are replaced by sandy loam and loam with inclusions of pebble and gravel. The lower horizon, represented by friable formations of alluvial deposits, is about 120 m thick. Mirzaev (1974) provides a detailed hydrogeological description of the study area.

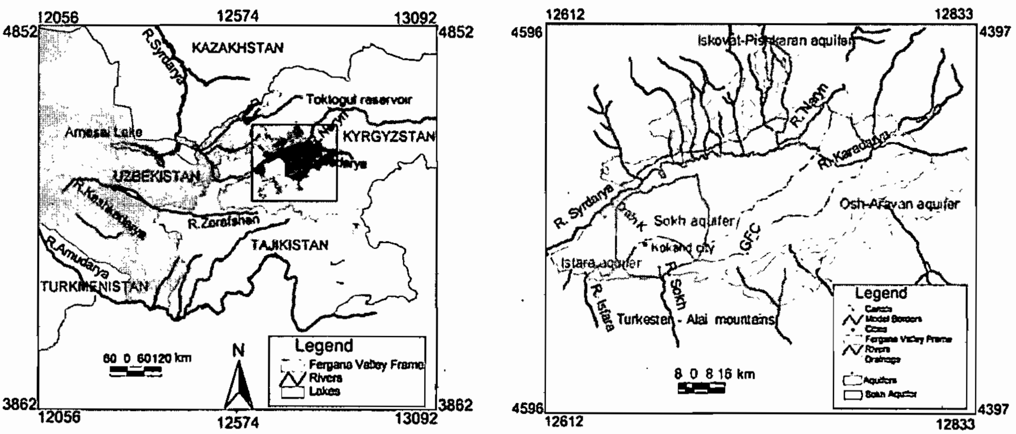

Fig. 1 The aquifers of the Fergana Valley.

METHODOLOGY

The three methodologies used in this paper are as follows:

(a) Historical analysis of the Naryn River seasonal flow within the Fergana Valley.
(b) Syrdarya River channel water budgeting within the Fergana Valley.
(c) Modelling of the Sokh aquifer development.

Data from the Hydro-meteorological service of Uzbekistan and the Basin Water Organization (BWO) "Syrdarya" were used for the river flow seasonal analysis and the water budgeting (CARE WEIS, 2008b). The Sokh aquifer model was compiled using Visual MODFLOW. Mavlonov *et al.* (2006) provided the groundwater data obtained from the hydrogeological survey carried out by the Institute of Hydrogeology and Engineering Geology in the study area. This data set was used for calibration of the model.

The surface of the model is approximately 2751 km^2. The model has grid spacing of 150 m and consists of 335 rows and 365 columns. A period of 5 years with a time step of 30 days was used. The three geological strata are represented as five layers in the model. Groundwater in layer 2 is unconfined–confined; in layers 3, 4 and 5 it is confined. The model is bounded on the north by general head conditions governing drainage outflow; the western and eastern boundaries are zero-flow ones lying at the edges of the aquifer. The upstream condition is a fixed flow boundary, representing underground inflow. The Great Fergana Canal (GFC) inside the model area is simulated as a river. Natural recharge, occurring directly as infiltration from precipitation, and seepage from streams of the Sokh River were included in the model. Another form of recharge is excess irrigation. Six recharge zones were defined based on soil types.

Two complementary water banking alternatives were examined: (a) banking of winter flow of the Sokh River was modelled in the natural recharge zone starting from the first simulation year; and (b) banking of winter flow of the Naryn River in the discharge zone from the third simulation year, after water table drawdown to 10–20 m deep. Four different scenarios were considered:

1. pumping by existing wells only;
2. additional irrigation wells in the transition and discharge zones;
3. as in scenario 2, but additional water transportation through the GFC in winter;
4. as in scenario 3, but structures incorporating in the head part injecting of 8.7 m^3/s in winter.

Scenario 1 was used to calibrate and verify the model. Summer withdrawal under Scenario 1 was 7.5 m^3/s and under scenarios 2 to 4, 38.0 m^3/s. The simulation results for the current scenario were evaluated based on data cited in Mavlonov *et al.* (2006) and obtained from the hydrogeological survey carried out by the Institute of Hydrogeology and Engineering Geology in 1978–1979. Further details of the model compilation and calibration are given in Gracheva *et al.* (2009).

RESULTS AND DISCUSSION

Flow-regulation function of the Fergana Valley

The Syrdarya River flow-regulation function of the Fergana Valley can be assessed by the analysis of seasonal flow of the river at the inflow and outflow nodes from the Fergana Valley (Fig. 2).

Under the irrigation mode of operation of the Toktogul Reservoir, the ratio of summer to winter was estimated at 1.27 at inflow and 0.86 at the outflow node (Fig. 2, left). Under the hydropower generation mode of operation of the Toktogul Reservoir, this ratio has reduced to 0.77 at the inflow and 0.40 at the outflow node (Fig. 2, right). This suggests that the summer flow within the valley is reducing while the winter flow is increasing. Under the new water management conditions, water withdrawals from the river, occurring mainly in summer, vary from 241 to 299 m^3/s in summer and from 85 to 138 m^3/s in winter in high and low water years, respectively. At the

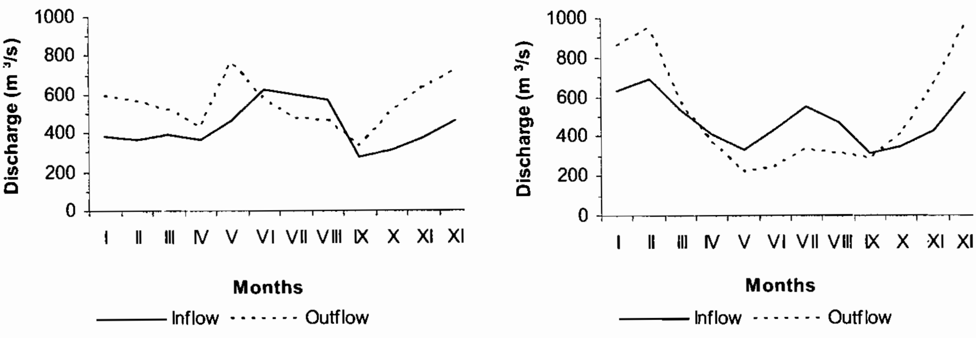

Fig. 2 Syrdarya River flow at the inflow and the outflow nodes of Fergana Valley under irrigation (1992 – left) and hydropower generation (2001 – right) modes of operation of the Toktogul Reservoir.

Table 1 Water budget of the Syrdarya River channel within the Fergana Valley for high and low water years.

Item	1995 (high water year) Winter (m^3/s)	Summer (m^3/s)	2001 (low water year) Winter (m^3/s)	Summer (m^3/s)
Inflow				
Surface inflow	526	404	538	416
Lateral	107	93	120	101
Drainage	156	119	185	111
Subsurface inflow	25	25	20	20
Total Inflow	813	641	863	648
Outflow				
Intake	85	241	138	299
Surface outflow	735	337	738	295
Total outflow	820	579	876	594
Balance	−6	62	−13	54
Error, %	−1	11	−2	9

same time, return flow to the river channel is in the range of 111–119 m^3/s in the summer and 156–185 m^3/s in the winter season. As a result, the return flow constitutes 40% of the river flow in winter at the outlet node in Fergana Valley. Water budgets of the Syrdarya River channel within the Fergana Valley for high (1995) and low (2001) water years reflect the flow-regulation function of the Fergana Valley (Table 1).

The data given in Table 1 suggest that, in a high water year, the winter inflow to the Fergana Valley amounted to 57% of the annual, while the summer inflow formed the remaining 43%. The water withdrawal for irrigation occurred mainly in summer, while 56% of the annual return flow to the river channel occured in winter. As a result, 69% of the river flow at the outlet of Fergana Valley occurred in winter and only 31% in summer. The same happened in the low water year, when 56% of the river inflow occurred in winter and 44% in the summer. Over 60% of the water withdrawals took place in summer, while 60% of the annual return flow to the river channel occured in the winter. As a result, 71% of the river outflow from the Fergana Valley happened in winter and only 29% in summer. This indicates that the share of winter flow in the annual flow is increasing within Fergana Valley from 56–57% to 69–71%, while the share of summer flow is reducing from 43–44% to 29–31%.

The data presented prove further transformation of the river flow within the Fergana Valley reduces the summer flow by 1.0–1.9 km^3 and increases the winter flow of Syrdarya River by 3.1–3.3 km^3. The summer evapotranspiration and the regulating capacities of the aquifers of Fergana Valley are contributing to the transformation of part of the summer withdrawals to the winter return flow.

Therefore, there are two complimentary strategies to adapt to new water management in Syrdarya River basin: (a) storing winter flow of the Naryn River in the Fergana Valley, most likely in underground aquifers, and/or (b) reducing winter return flow to the river channel. Both strategies require groundwater development and lowering of the shallow water table. We tested these two complementary strategies through: (a) modelling of the water banking at a sub-basin level; (b) assessing the groundwater irrigation potential of the Fergana Valley. Groundwater development for irrigation and water banking was simulated for the Sokh aquifer, one of the 18 aquifers of Fergana Valley.

MODELLING WATER BANKING IN THE SOKH AQUIFER

Simulations suggest that increasing the groundwater summer abstraction from 7.5 to 35.4 m^3/s creates free capacity of 10–20 m thick in the GFC zone, and hence increases the leakage (Fig. 3).

Figure 3 shows that leakage from GFC under Scenario 1 was 2.3 m^3/s in summer and 0.6 m^3/s in winter. Increasing the groundwater summer abstraction from 7.5 to 35.4 m^3/s in Scenario 2

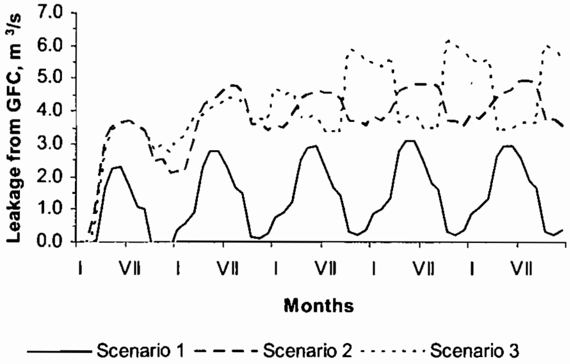

Fig. 3 Leakage from GFC under the different scenarios of groundwater development.

caused leakage from the canal at a rate of 4.6 m^3/s in summer and 3.0 m^3/s in winter. Water transportation through the canal in winter under Scenario 3 resulted in 5.9 m^3/s leakage from the canal. Therefore, increasing the groundwater abstraction in summer led to lowering of the water table and increased the leakage from the canal from 0.6 to 5.9 m^3/s in winter.

Under Scenario 4, 8.7 m^3/s of winter flow of Sokh River was added to the groundwater by artificial recharge in the head part of the fan. In total, 14.5 m^3/s of surface flow was recharged into the aquifer in winter. Winter return flow from the study area is reduced from 15.7 to 8.5 m^3/s, i.e. almost twice. Demand for summer withdrawals from Naryn River is reduced by 43.6 m^3/s.

GROUNDWATER IRRIGATION POTENTIAL IN FERGANA VALLEY

The results of modelling of the Sokh aquifer development prove that groundwater can supply 50% of crop water requirements. The groundwater irrigation in the Sokh aquifer spread area will contribute to reduction of the Syrdarya River winter flow at the outlet of the Fergana Valley by 0.112 km^3 (and to an increase of the summer flow by 0.678 km^3). Karimov *et al.* (2008) applied zoning of the Fergana Valley and showed that other aquifers in the Fergana Valley have similar potential. Zoning of the irrigated land based on the conductance of underground deposits and other parameters proved there is a high potential for groundwater development and conjunctive use for irrigation in Fergana Valley. The studies indicated that groundwater irrigation could be practiced on 290 000 ha in Fergana Valley (32% of the total) and conjunctive use on 243 000 ha (27%), which will cause a decline of the groundwater recharge by 1.7 km^3. This deficit of groundwater recharge can be compensated by banking the winter flow of the Naryn River and the small rivers of Fergana Valley. The free capacity of aquifers within the Fergana Valley amounts to 3.1 km^3, while winter flow of the small rivers averages 1.0 km^3. The deficit of the winter flow can be compensated for from the Naryn River. This amount is estimated to be 0.7 km^3. The winter return flow to the river channel will reduce by 0.8 km^3 due to the reduced groundwater recharge. Further reduction of the return flow from the system can be achieved by adoption of water saving technologies.

CONCLUSIONS

Groundwater development for summer irrigation and storage of the winter flow of rivers in aquifers can be a feasible practice for river basins with vulnerable water resources. This strategy allows reduction of the pressure on river basins in extreme situations emerging due to increased water demands and exposed water resources. The studies in the Fergana Valley prove that a shift from canal irrigation to groundwater and conjunctive use, combined with banking of winter river flow, can reduce this excessive flow, originating from hydropower generation, by 50% and ensure improved water supply to downstream water uses.

The groundwater irrigation may cause a drawdown of the water table in the Fergana Valley due to the reduction of the leakage from the canal system. This can be prevented by banking the winter flow of the small rivers and the Naryn River in the aquifers of the Fergana Valley. Further studies are required to examine the economically and environmentally optimal combination of summer extractions and winter banking for each of the aquifers of the Fergana Valley. This strategy would contribute to sustainable merging of the upstream hydropower and the downstream irrigation and environmental needs.

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Tank management in Andhra Pradesh, India: percolation *versus* irrigation

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Abstract Hard-rock aquifers located in semi-arid climatic conditions are especially prone to over-exploitation because of limited storage and recharge. In Andhra Pradesh (southern India), such geological and climatic settings prevail, and rapid increase in groundwater abstraction for irrigation has led to aquifer over-exploitation in many districts. As a response to over-exploitation, Central and State governments have launched watershed development programmes aiming at augmenting aquifer recharge using different man-made structures such as percolation tanks, check dams and defunct dug wells. The objective of the present study is to determine an accurate water budget for one experimental tank and then to simulate different tank water management practices with the scope to optimise water use. The selected tank is located in Gajwel watershed, 60 km north of Hyderabad, India, and has a maximum storage capacity of about 105 000 m^3. During one recession period, the water budget is calculated under present site conditions (percolation tank) and another water budget is simulated for the case of use of tank water for irrigation. Conjunctive use (i.e. optimised use of tank water complemented, if required, by groundwater) proves to be beneficial for the overall water budget as evaporative loss is minimised.

Key words tanks; hard-rock aquifer; semi-arid climate; water management; conjunctive use; India

INTRODUCTION

As a consequence of the "green revolution", many aquifers have become intensively exploited over large areas of India. Some of these aquifers are now considered as over-exploited, as groundwater draft exceeds recharge. Over-exploitation is a widespread concern in southern India where semi-arid climate and hard-rock aquifers prevail. A possible management solution, encouraged by Government agencies over the recent years, is to intervene on the supply side of the groundwater balance, i.e. by augmenting recharge artificially. For instance, the State of Andhra Pradesh set the objective to increase aquifer recharge from 9% of total rainfall under natural conditions to 15% by the year 2020 (Government of Andhra Pradesh, 2003).

Artificial recharge structures encompass percolation tanks, check dams, injection wells and defunct dug wells. In many places, a high density of tanks exists as they were the traditional water source for many centuries. Only in recent years were they renamed "percolation tanks". Several studies have looked at the hydraulic functioning of these tanks and their contribution to recharge augmentation (Muralidharan *et al.*, 1995; Selvarajan *et al.*, 1995; Sukhija *et al.*, 1997; Gore *et al.*, 1998; Chary & Subbarao, 2003; Sudarshan, 2003; Machiwal *et al.*, 2004; Sharda *et al.*, 2006). The efficiency (i.e. volume percolated compared to total volume stored) of these artificial recharge structures ranges from 30% (Sukhija *et al.*, 1997) to 57% (Mehta & Jain, 1997).

This present study focuses on a percolation tank located in Gajwel watershed (83 km^2, Medak district, Andhra Pradesh, Fig. 1) where a very detailed field investigation has been carried out to quantify the tank water balance (Perrin *et al.*, 2008; Massuel *et al.*, 2008a). On this basis, a simple tank water balance model has been designed in order to examine the impact of tank management strategies (e.g. enhanced percolation, conjunctive use) on the water balance. This approach may help to evaluate the adequacy of artificial recharge programmes carried out by Government agencies.

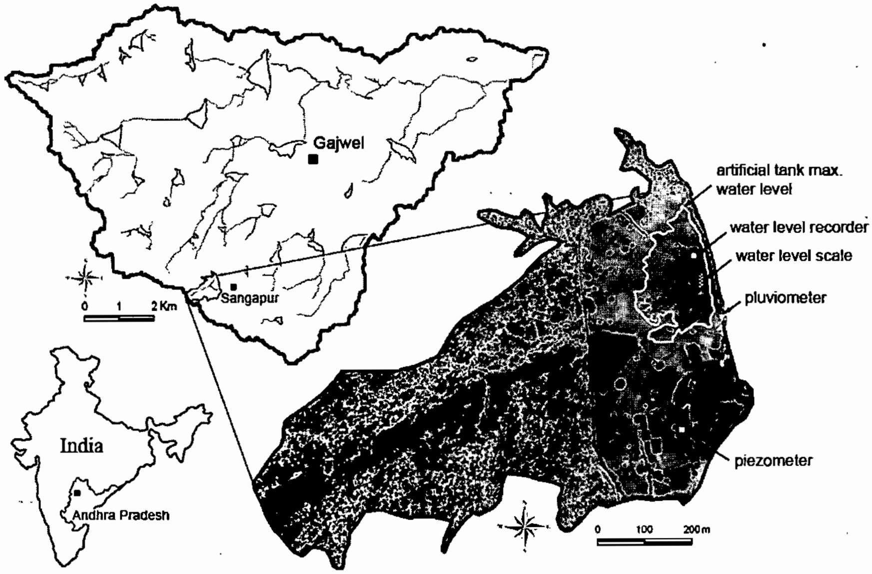

Fig. 1 Location of the study area and monitoring scheme. Artificial tank maximum water level refers to the maximum level reached in 2007 (h = 214 cm).

TANK WATER BALANCE MODEL

During tank recession periods (i.e. when no rainfall occurs), the tank water volume variation is only due to evaporation (E), percolation (Pe) and livestock consumption (L). Therefore, under present site conditions the tank water balance is simply:

$$\Delta V_{(t)} = Pe_{(t)} + E_{(t)} + L_{(t)} \tag{1}$$

where ΔV is the tank volume depletion, L is averaging 3.7 m^3/d (Massuel *et al.*, 2008a) and E is given by:

$$E = A \times (E_{pan} + ET) \tag{2}$$

where A is the tank area and E_{pan} is the evaporation measured in the class A evaporation pan at a farm less than 2 km away from the tank, and ET is the estimated evapotranspiration of the vegetation growing in the tank based on the crop requirement calculation (Perrin *et al.*, 2008).

The tank area was contoured regularly during the depletion period with a GPS so as to obtain altimetric contour lines which have been used to determine the precise tank-bottom topography. Based on the recorded tank water level and tank-bottom topography, empirical relationships between tank water level, tank volume and tank surface area are established (Fig. 2). Applying equations (1) and (2), Pe is calculated on a daily basis using the daily water level variation and E_{pan} measurements of the recession period 2007–2008. Volume of percolated water *versus* tank water level is plotted on Fig. 2. The percolation rate is then approximated (Pe^*) with a step function following the three different observed phases: (i) a constant percolation rate greater than 140 cm; (ii) a steep linear decrease in percolation between 140 cm and 130 cm; and (iii) a moderate linear decrease in percolation below 130 cm (siltation).

In order to simulate the management scenarios of the tank at time t, a virtual tank water uptake (e.g. for irrigation) term ($U_{(t)}$) was added in the balance equation which becomes:

$$\Delta V_{(t)} = Pe^*_{(t)} + E_{(t)} + L_{(t)} + U_{(t)} \tag{3}$$

Different values of $U_{(t)}$ are applied over different periods of time and the resulting volume variation ($\Delta V_{(t)}$) of the tank is calculated on a daily basis.

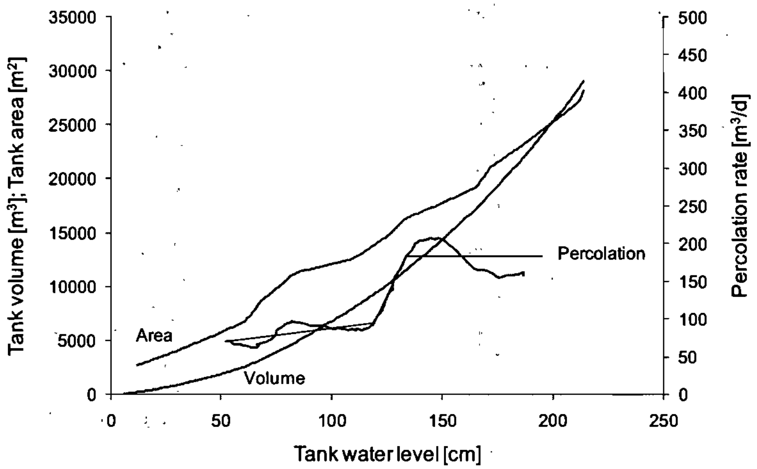

Fig. 2 The empirical relationships between tank water level and tank surface and volume are shown by the grey curves labelled "Area" and "Volume"; Also shown are the calculated percolation rate for the recession period 2007–2008 (grey curve) and the percolation rate approximation model (black curve).

RESULTS

Over the recession period 2007–2008, the tank water level was monitored on a 15-min interval with a pressure data logger (see Perrin *et al.*, 2008). Daily pan evaporation measurements were averaged over 10 days in order to get representative evaporation values. Tank depletion ended on 11 February 2008 when first rainfall occurred. A study by Bernatowicz *et al.* (1976) showed that transpiration of emergent plants in surface water bodies contribute on average to an additional 20% loss as compared to evaporation alone. Similarly ET (equation (2)) is taken as $0.2 \times E_{pan}$.

The tank water storage after the last flood (in October) was 29 000 m^3. At the end of the depletion period, 1300 m^3 remained in the tank. According to the water balance, evaporation accounted for 44% (~12 100 m^3) of the total losses, while 56% (~15 700 m^3) disappeared by deep percolation (Fig. 3, Table 1). This result corresponds to the upper range of percolation efficiency found by previous studies (e.g. Mehta & Jain, 1997, found Pe = 57%).

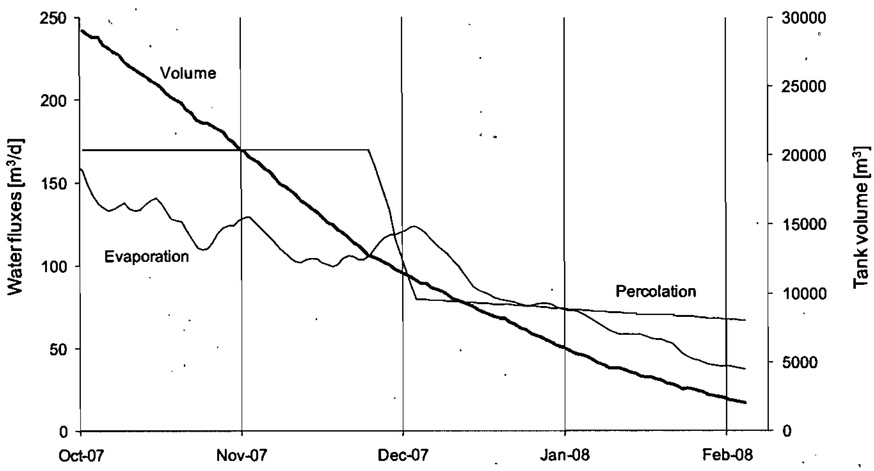

Fig. 3 Daily evolution of tank volume, evaporation fluxes and percolation fluxes for the monitored period.

Table 1 Comparison of total fluxes (m^3) in the case of no tank management (present status) and optimized management (daily uptake = 300 m^3).

	Present conditions	Optimized management scenario
Initial stored volume	29 070	29 070
Total evaporated volume	12 086	4 912
Total percolated volume	15 666	7 048
Total uptake volume	-	17 100
Final stored volume	1 318	10
Groundwater benefit	-	8 483

Optimised tank management with respect to the overall water balance means reducing evaporation loss; this can be achieved either by increasing percolation efficiency (e.g. desilting tank bottom, injection wells), or by direct use of surface water (e.g. conjunctive use, irrigation with tank water instead of groundwater). The first option may not be applicable in the case of shallow piezometric levels because the tank water level is hydraulically connected to the groundwater level (no unsaturated zone below the tank). In this situation, which prevails in our case study, the percolation rate is mainly controlled by the aquifer transmissivity. The second option is applied to the tank. In this case, the optimal tank management can be quantified in terms of groundwater benefit (B_{gw}),defined as the total tank water uptake (i.e. water that does not need to be pumped from groundwater) minus the shortfall of percolated water due to the shortened tank recession period; this can be expressed as:

$$B_{gw} = \sum_{t=1}^{n} U_{(t)} - \sum_{t=1}^{n} \left(Pe_{(t)} - Pe^{*}_{(t)}\right) \tag{4}$$

Various daily water tank uptakes, ranging between 50 and 1000 m^3/day, are applied to the tank model. Towards higher daily uptakes, total evaporation is minimised and groundwater benefit maximised (Fig. 4). However, the water uptake duration will decrease (as the tank gets depleted more rapidly). If tank water is available for too short a period, this may not be convenient for irrigation as water availability will be limited to only part of the crop growth calendar. Therefore, a trade-off between the higher groundwater benefits and tank water availability has to be determined. From Fig. 4, it appears that a good compromise may be found in the range of 300–400 m^3 daily uptake, which allows about 50 to 57 days of irrigation (the paddy irrigation period is 100 to 140 days).

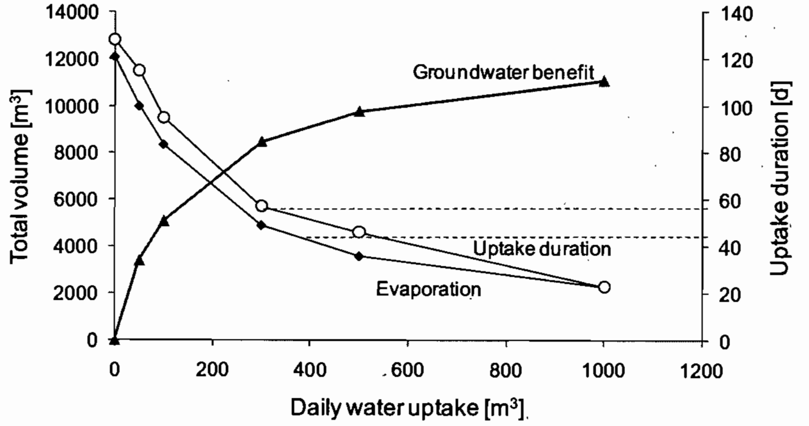

Fig. 4 Total evaporation, groundwater benefit and uptake duration for daily water uptake ranging from 0 to 1000 m^3. Daily uptakes within the shaded area represent a good compromise between maximal groundwater benefit and sufficient uptake duration.

The selected optimized tank management scenario consists of a daily tank water uptake of 300 m^3/d from 7 October until 2 December (57 days) when the tank is running dry (Fig. 5). This daily uptake may be used for augmenting aquifer recharge or for irrigation. In the case of paddy irrigation, which represents by far the main irrigated crop in the region, this would represent a significant supplementary water source for the rainy season crop calendar between July and November. In the region, irrigation water applied to paddy amounts to 10 mm/d during the rainy season (Marechal *et al.*, 2006; Dewandel *et al.* 2008). About 3 ha of paddy fields could be irrigated by applying a conjunctive use scheme (presently 13 ha of paddy are cultivated in the tank surroundings and the entire irrigation is groundwater based).

The estimated groundwater benefit under this optimized tank management scenario is about 8500 m^3 (Table 1); this corresponds to a 60% reduction of E from the tank. This is equivalent to the daily pumping from 1.7 irrigation wells over the same period of 57 d (a survey on 14 irrigation wells in the tank surroundings showed an average daily abstraction of 89 m^3 per well (Massuel *et al.*, 2008b).

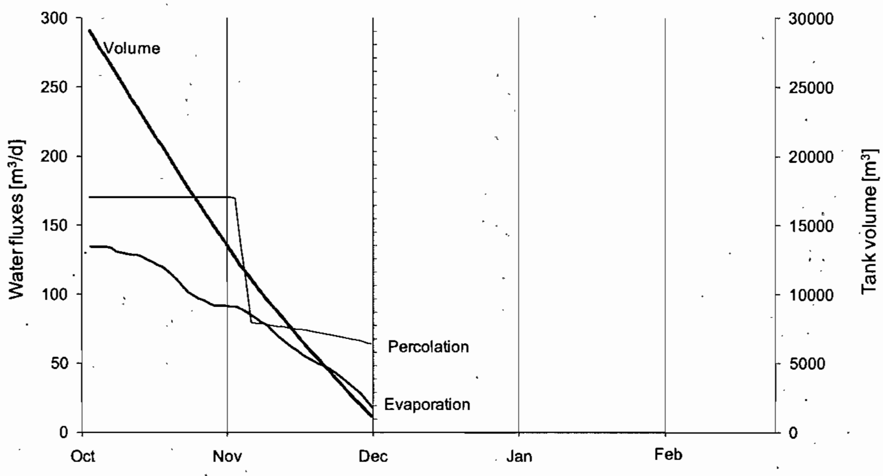

Fig. 5 Daily evolution of tank volume, water uptake, percolation and evaporation in the case of an optimized tank management scenario (daily uptake of 300 m^3/d, duration: 57 d).

DISCUSSION AND CONCLUSIONS

This simple tank water balance model shows that substantial irrigation can be carried out during the rainy season using tank water (e.g. conjunctive use for paddy irrigation) with a positive impact on the groundwater balance. Calculations have been made for an average monsoon year (total monsoon rainfall = 536 mm, maximum tank storage = 29 000 m^3); therefore greater benefits are to be expected during good monsoons (e.g. when the tank is at full capacity = 105 000 m^3).

Conjunctive use appears to be the most appropriate way to optimise tank management, as tank desilting or use of injection wells may fail as the groundwater cannot accommodate additional recharge in the vicinity of the tank. The construction of additional artificial recharge structures may also fail in watersheds such as Gajwel, where no surface outflow exits the watershed during a normal monsoon year meaning that all the rainfall/runoff is already captured by existing structures.

This optimised tank management is beneficial to the groundwater balance and equivalent to switching off two irrigation borewells over two months. However, other optimisation methods could be considered, such as managing borewell pumping rates close to the tank so as to increase tank percolation (i.e. by increasing hydraulic gradients).

Irrigation return flow does not appear in equation (4) since the water flux will remain the same whether from the surface or groundwater source. From a groundwater quality perspective, surface water irrigation return flow may, however, have a positive impact on groundwater mineralisation (dilution) or a harmful impact if anthropogenic contaminants are present in the tank water.

The usage of surface water is also beneficial from an energy stand point as virtually no pumping is required for tank water irrigation (gravity drainage towards paddy downstream).

The main difficulty of conjunctive use may be its social acceptance, as it means there is a community-based management of the water resource as opposed to the purely private management offered by irrigation wells.

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Modelling of the urban water cycle and the consequences of massive rain runoff infiltration for Copenhagen

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Abstract An integrated model is used to simulate the water cycle of the Copenhagen area (Denmark, 976 km^2) during the period 1850–2003. The water cycle is quantified in terms of root zone water balance, water supply, wastewater, rain runoff, groundwater flow, and streamflow. The simulation indicates that the historical increase in precipitation (20%) has led to a 36% net increase in recharge, although urbanization has prevented groundwater recharge due to impermeabilisation and negligible contributions from leaky pipe systems. A model scenario that attempts to infiltrate all rain runoff from the city shows that only 25% of present rain runoff can be infiltrated due to a shallow groundwater table. In the area, 51% of the rain runoff infiltration is simulated to find other pathways to the sewer pipes as groundwater infiltration. The baseflow of the depleted streams in the region will only increase by 18% if rain runoff is infiltrated.

Key words urban; water cycle; groundwater; rain runoff; wastewater; streamflow; modelling

INTRODUCTION

Integrated sustainable urban water management, including groundwater, is becoming increasingly important because of its significance to economic, social, political, and environmental issues. For such management, quantification of the urban water cycle is an essential task which is complicated by numerous possible flow interactions between the urban subsystems (see Lerner, 1986; Foster, 1990, for examples). Lerner (2002) therefore recommends the application of integrated model approaches, including a groundwater model, to minimize, or at least to better understand, the uncertainty associated with urban water cycle quantification. Moreover, Vázquez-Suñé *et al.* (2005) recommend the inclusion of historical changes in the urban water cycle in order to reduce uncertainty of the integrated model. These changes include land-use changes, evolution of indirect recharge sources from leaking pipe systems, groundwater abstraction by industries and water works, and development of urban underground structures.

Here, an integrated groundwater–surface water model is used to study the water cycle of the Copenhagen area (976 km^2, Fig. 1) for the period 1850–2003. The urban water cycle is quantified in terms of root zone water balance, water supply, wastewater, rain runoff, groundwater flow, and the interactions between these systems. By simulating the period 1850–2003, the complete history of groundwater abstraction and major city development in the region is covered.

The main objectives of this paper are to present the modelling principles, the simulated water balance for the period 1850–2003, and to simulate and analyse the hydrological consequences of large-scale rain runoff infiltration within the city.

URBAN HYDROLOGY MODEL

The urban hydrology model consists of three utilities used in sequence:

1. a root zone model,
2. a grid distribution tool, and
3. a groundwater flow model.

The major water flows simulated by the three model parts are explained in the following.

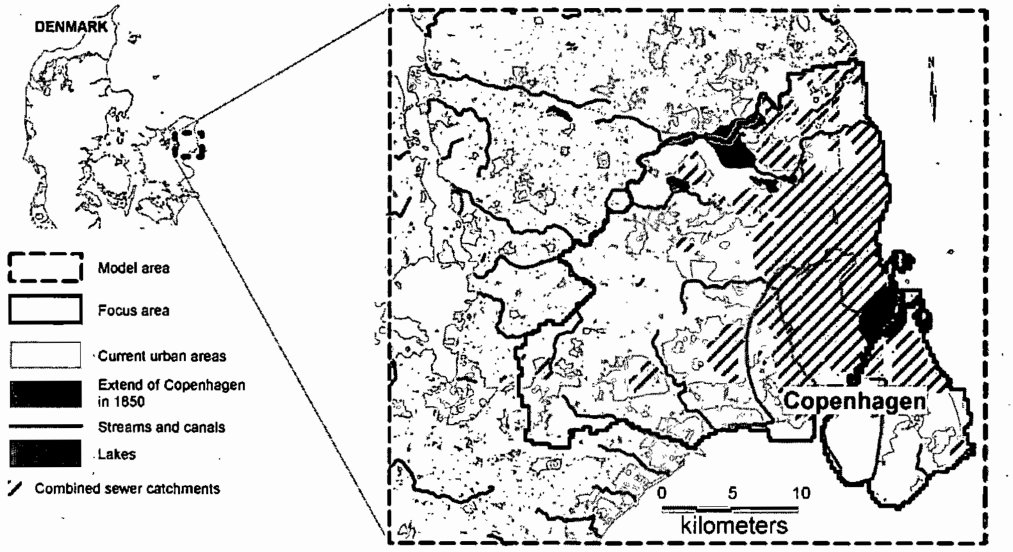

Fig. 1 Map of Copenhagen, Denmark, showing model area, streams and lakes, current urban areas, extent of Copenhagen 1850, and the area of focus.

Utility one: Root zone model

Utility one consists of the Daisy model (Hansen *et al.*, 1991), which simulates one-dimensional root zone water balances and crop production on a daily basis. This is done for relevant combinations of soil type, land uses and climate zone.

Utility two: Grid distribution tool

The grid distribution tool distributes percolation results from the root zone model to the finite difference grid of the groundwater flow model (Utility three). The distribution to each grid cell is done in accordance with the presence of soil types and land uses within each grid cell.

Rain runoff is calculated and distributed in accordance with precipitation and areas of impervious surface within each model cell. The historical evolution of impervious areas has been implemented on the basis of information from the literature and historical maps.

The tool also distributes leakage from water mains, gross wastewater, and irrigation excess to model cells. The amount of unaccounted water supply that becomes groundwater recharge was assessed, by model calibration, as 50%, whereas the remaining part is assumed to end up in sewers. The historical gross wastewater production within each municipality is estimated to be 94% of the total water supplied to the users (i.e. 94% of accounted water). For the remainder, 4% is assumed to be used for irrigation (evapotranspiration) and 2% becomes irrigation excess (artificial recharge).

Grid-based sewer type information is used to specify whether rain runoff should be added to stream discharge (for separated sewer systems) or contribute to the simulated inflow to sewage works (for combined sewer systems). Information about sewage work catchments is used to accumulate and aggregate rain runoff and wastewater production to the relevant positions (model cells) of outlets to streams or inlets to sewage works.

The direct leakage from rain drains to sewers in separated systems is explicitly represented by subtracting 10% of rain runoff in cells related to separate sewer systems, and adding this amount to the sewage inflow according to the specified sewer catchments.

The grid distribution tool writes the total groundwater recharge (the sum of recharge from the root zone, from water mains leakage, and from irrigation excess) as input to the groundwater flow model.

The tool also writes the historical distribution of drains and streams as input to the groundwater flow model. Foundation drains in urban areas are assumed to be located 1.2 m below surface, whereas buried tile drains in farmland are assumed to be located 0.8 m below surface. The historical change from agricultural to urban drainage is estimated from historical maps and implemented in the grid distribution tool by using zones associated with a specific temporal evolution of drainage depth. The historical impermeabilisation and piping of streams within Copenhagen are roughly represented by applying a significant decrease in stream bed conductance following city development. Finally, input to a new Sewer Package, described with Utility three, is written with the grid distribution tool.

All relevant grid-based water balance terms are saved to a Modflow-style binary output format (McDonald & Harbaugh, 1988). Afterwards, Zonebudget (Harbaugh, 1990) is used to accumulate rain runoff from separated sewer catchments, outflow from sewage works, and outflow from remediation works in specified zones (catchments). These accumulated flow contributions are added to the right segments and lake reaches in the relevant input files to the groundwater flow model (Utility three).

Utility three: Groundwater flow model

The groundwater flow model is based on the Modflow-2000 code (Harbaugh *et al.*, 2000) which has been modified for this work.

The groundwater model uses: the LPF-package (Harbaugh *et al.*, 2000) as the internal flow package; constant head cells to simulate interactions with the sea; the Recharge Package (McDonald & Harbaugh, 1988) to simulate input of recharge; the Well Package (McDonald & Harbaugh, 1988) to simulate groundwater abstraction; the SFR1-package (Prudic *et al.*, 2004) in conjunction with the LAK3-package (Merritt & Konikow, 2000) to simulate groundwater exchange with streams, lakes and wetlands; and the Drain Package (McDonald & Harbaugh, 1988) to simulate efflux to drains that contributes to either streamflow or sewage flow (the latter for city areas with combined sewer systems).

Exchange of water between aquifers and sewers is simulated by a specified-head style Sewer Package (developed in this project), which is very similar to the River Package (McDonald & Harbaugh, 1988). The calculated exchange rate depends on the pipe dimension, the groundwater level, the water level in the pipe, and a leakage factor.

A Selected Flow Integration Package (SFI-package) has been developed in order to accumulate, for example, drain flow within user specified zones (catchments) and add these flows within the Modflow iteration loop to specified stream segments or lake reaches.

The LPF-package is modified in order to overcome the problems with "dry" cells by applying the methodology described by Doherty (2001). The methodology ensures that no cell is ever declared as dry by allowing the head in a cell to fall below its base. In such situations an exponential decay function is used to significantly decrease transmissivity as the head in the cell falls deeper and deeper below its base. The methodology of Doherty (2001) is extended in this study to also include an exponential decay of actual vertical flow towards zero in a situation where a deeper cell is dewatered and the head in the adjacent upper cell falls below its base.

Furthermore, in order to roughly assess the hydrological consequences of massive rain runoff infiltration, a Head Dependent Recharge Package (HDPR-package) is developed. The package only enables rain runoff infiltration in a cell if the head in that cell is below a specified elevation (which could be the approximate depth of an infiltration device). However, in order to overcome instability problems caused by such a threshold, decaying recharge towards zero is allowed as the water table rises up to 0.5 m above the specified threshold (described as exponential decay).

A seven-layer Modflow-2000 model (Harbaugh *et al.*, 2000) has been set up for the investigation area in Fig. 1. Low-conductivity clayey till layers are represented by model layers one, three, and five, while sand layers are represented by model layers two and four. However, in areas with no observed till (where there are "windows" in the till layers) the corresponding model layer represents sandy sediments. Model layer six represents the uppermost 10 m of limestone which is usually found to have a high transmissivity, while model layer seven represents the deeper less-conductive part of the limestone which is assumed to be 30 m thick.

The horizontal model grid size varies between 125 m × 125 m and 250 m × 250 m. The 153 years of water balance and flow are simulated using a total of 4017 stress periods, each divided into 14 daily time steps.

Post processing: water balance extraction

Having used the above three modelling utilities, Zonebudget is used as a post processor to extract relevant water flows from the two Modflow-style binary output files written by the Grid-distribution tool and the groundwater model, respectively. Thereby water balance results can be compiled for arbitrary zones of the model area.

PARAMETERISATION AND CALIBRATION

A total of 15 model parameters were calibrated manually against 419 time series of observed hydraulic head, stream discharge observed at 11 stations, and measured inflow to three sewage works. The parameters include seven hydraulic conductivities; two specific yields; four leakage coefficients (one for streams, one for drains, and two for sewers); one multiplier used to adjust the initially-estimated impervious cover fractions; and one multiplier used to control the amount of unaccounted water that becomes groundwater recharge due to mains leakage.

During calibration, observed stream discharge was compared against the sum of groundwater contributions from drains, streams, and "urban water" from rain runoff, remediation works and sewage works. Observed inflow to the sewage works was compared against the sum of accumulated rain runoff from combined sewer catchments, wastewater production, small parts of unmeasured and measured water consumption, direct leakage from rain drains to sewers in separated sewer catchments, drain flow (contributing to sewage flow), and simulated net infiltration of groundwater into the sewer network.

RESULTS

Simulated water balance 1850–2003

The historical evolution of the water cycle for the greater Copenhagen area (focus area in Fig. 1) was simulated using the calibrated model.

Figure 2 shows the evolution of selected urban water flows in the greater Copenhagen area. Precipitation rates have increased steadily by about 125 mm/year (20%) from 1850 to 2003. Evapotranspiration from pervious areas has decreased by about 25 mm/year (5%) since the 1950s due to the growth of impervious areas and corresponding decrease of pervious areas. For the same reason, rain runoff has increased by 115 mm/year during the simulation period and is now a significant component of the urban water balance of the Copenhagen area. Despite the increasing proportion of precipitation that has turned into rain runoff, the increase in precipitation has been sufficient to also increase recharge of groundwater by 40 mm/year (36%).

Groundwater abstraction peaked in the 1960s and 1970s at 200 mm/year when the need for water supply was at its maximum. The massive groundwater abstraction taking place in the area has caused widespread streamflow depletion. Thus today the groundwater seepage into streams, which constitutes the baseflow, is only 35% of the seepage in the 1850s.

Simulated groundwater infiltration into sewers has risen to the current levels of 15 mm/year due to a rising water table and due to sewer ageing and consequent degradation. Exfiltration from sewers is not shown in Fig. 2 as this contribution was/is diminishing. The small exfiltration can be explained by the relatively shallow groundwater table in Copenhagen, which prevents leakage from the sewers. For other cities the exfiltration has often been estimated, or assumed, to be about 5% of sewage flow (see Vázquez-Suñé *et al.*, 2005, for an example).

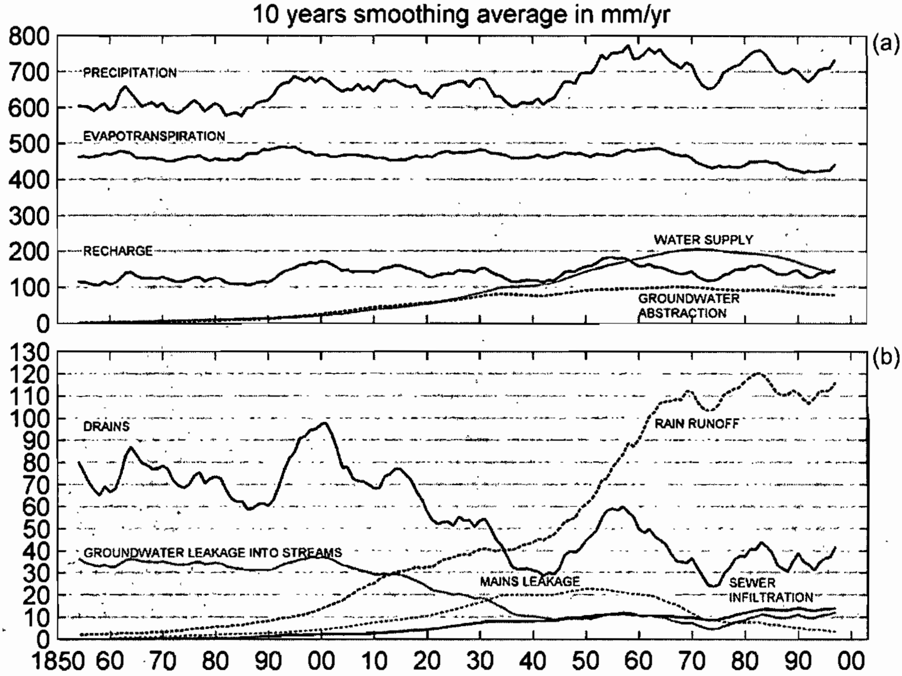

Fig. 2 Selected historical water flows for the focus area from 1850 to 2003.

In the 1940s and 1950s, leakage from water mains to groundwater, which is estimated as 50% of unaccounted water, peaked at about 20 mm/year. Since the 1960s and 1970s, intense pipe renovation has been carried out, so today's mains leakage corresponds to only 3 mm/year which is just 2% of the total pumping from water supplies. This is very small compared to the losses often reported for other cities. Carcia-Fresca (2004), for example, reports that losses of 20–25% are typical in the USA while the average in the UK is 16%.

In Copenhagen, rain runoff infiltration is considered as a strategy to overcome sewer overflows caused by insufficient sewer capacity during heavy rain storms. However, prior to the implementation of such a water management practise some key issues regarding the groundwater system have to be analysed, e.g. where, when, and how much rain runoff can be infiltrated given the groundwater levels, and which groundwater flows will consequently increase?

In order to help answer such questions, the model has been used to simulate the hydrological consequences of hypothetical massive rain runoff infiltration to the groundwater system by using the developed HDPR-package for all urban areas and for the entire simulation period 1850–2003. This is done purely to illustrate the long-term potential for and consequences of runoff infiltration.

This simulation is set up so that for each individual model cell all rain runoff generated within a time step is infiltrated to the groundwater system provided that the groundwater table is below a certain threshold depth (here 2 m below ground surface); otherwise, if the groundwater table exceeds the threshold, the rain runoff is directed to the sewer system (or to the rain drains in areas with a separate system). The simulated results of infiltrating rain runoff in the focus area are shown in Fig. 3 in terms of gains in sewer pipe leakage, drain flow, stream leakage, and flow into and out of the focus area. The following is observed.

Imagining that rain runoff had been infiltrated through the entire simulation for all urban areas, approximately 25% (28 mm/year) of present day actual rain runoff in the focus area can be infiltrated, while the remaining 75% of the runoff ends directly in sewers and rain drains. Besides this amount of rain runoff infiltration, the groundwater system in the focus area would receive

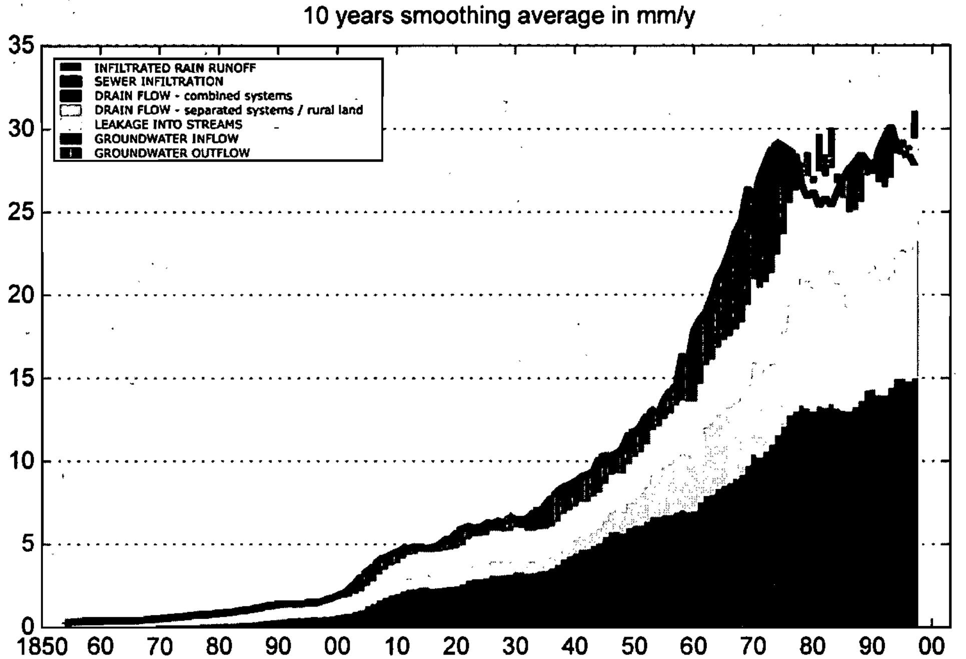

Fig. 3 Simulated changes of flows for the greater Copenhagen area caused by hypothetical rain runoff infiltration through the period from 1850 to 2003.

additionally 2 mm/year as horizontal flow which originates from rain runoff infiltration outside the focus area. Thereby the total input of rain runoff infiltration to the focus area is 30 mm/year.

Ten mm/year (33%) of the rain runoff infiltration will eventually leak into the sewer system in the focus area, 6 mm/year (18%) will leak into drains connected to combined sewers, 8 mm/year (28%) will leak into drains and end up in the streams, and 6 mm/year (21%) will leak into streams.

DISCUSSION AND CONCLUSIONS

Actual simulated water balance 1850–2003

The simulated water balance for Copenhagen shows that the historical impacts from urbanization and climate are massive and complex.

An interesting finding is that urbanization (impermeabilisation and negligible contributions from leaky pipe systems) seems to have lowered groundwater recharge in the Copenhagen area, which is contrary to the tendency found in other city studies (see Carcia-Fresca, 2004, for various city reports). However, for the Copenhagen case it should be noticed that recharge has increased by 40 mm (36%) from 1850 to 2003. Thus the increase in precipitation has more than counter-balanced the decrease in recharge caused by the development of impervious areas.

This clearly indicates that the current problems of groundwater shortages and streamflow depletion would have been far more severe if the climatic conditions (precipitation) had not changed (increased) significantly during the last 150 years.

Scenario of large-scale rain runoff infiltration

The hypothetical scenario of large-scale rain runoff infiltration indicates that if all rain runoff is directed to soak-aways and infiltration basins (which is not realistic in practise), the shallow ground-water table will only allow about 25% of current rain runoff to infiltrate to the deeper groundwater system. The remaining 75% of rain runoff would still have to be routed to sewers or streams.

This result indicates that it is unlikely that the current problems of combined sewer overflows to streams and canals within Copenhagen can be solved solely by infiltrating rain runoff to the groundwater system. Efficient reduction of overflows probably requires a greater reduction of rain runoff volumes. An interesting finding is that even if we could infiltrate 25% of rain runoff, 51% of the infiltration would find other (slower) pathways to the sewer pipes, either as indirect drain flow or as direct infiltration into the sewer pipes caused by the rising groundwater table. This finding is highly attributable to the layer of low permeable till covering the area, which hinders vertical deep percolation, whereas shallow horizontal subsurface runoff into drains or leaky sewers is enhanced.

For the same reason the contribution from rain runoff infiltration to stream baseflow is only 21%. The total contribution is simulated as only 2 mm/year, which correspond to 18% of current baseflow, or to 7% of the natural reference baseflow in the 1850s. Therefore, large-scale infiltration of rain runoff into the upper till layer does not seem to be a promising strategy to efficiently increase baseflow in the streams.

Although direct infiltration of rain runoff into the somewhat deeper upper sandy aquifer might be possible, significant reduction of groundwater abstraction seems to be the obvious way to increase stream baseflow within the region.

Evaluation of modelling concept

The developed concept is characterized by modelling groundwater flow and its related interactions with natural hydrological boundaries at a fairly advanced level. In contrast to this, modelling of interactions with urban subsystems such as leaky pipes is based on more simple principles. For example, stages in sewer pipes are specified and not modelled as dynamic variables calculated on the basis of routing.

A criticism could be that the modelling concept is too simple, and more advanced physically-based modelling concepts have indeed been developed (see Wolf *et al.*, 2006, for an example). In order to benefit from applying such advanced models very detailed and accurate input data are required. Such data will rarely (never) be available for an entire city such as Copenhagen. Moreover, it would not be computationally feasible to use such a detailed model for the purpose we have, to model the overall water cycle for an entire city for a long period of time for alternative water management practises. For our purpose, we find it more feasible to use the somewhat simplified approach developed here.

However, the detention storage of infiltration installations (e.g. soak-aways) should be included in the model in order to simulate rain runoff infiltration more realistically. We plan to do this by developing a soak-away package for Modflow.

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Assessment of managed aquifer recharge to improve the security of urban water supply in western NSW, Australia

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Abstract The city of Broken Hill in western New South Wales, Australia, obtains most of its water supply from the Darling River at Menindee. An allocation of water for the city is stored in Menindee lakes, and up to 90% of the allocated water is lost due to evaporation. A possible alternative to wasteful surface storage is to use managed aquifer recharge techniques to store water in the alluvial aquifers associated with the Darling River at Menindee. A project to investigate the feasibility of this concept has been underway since 2006. This project has included a drilling programme, surface and down-hole geophysics, pump testing, geochemical modelling and groundwater flow modelling.

Key words Australia; managed aquifer recharge; Menindee, Darling River; groundwater modelling

BACKGROUND

The City of Broken Hill, in western New South Wales, Australia, sources a substantial proportion of its water supply from the Darling River at Menindee, a supply that is maintained by storage in the Menindee lakes. At Menindee, as shown on Fig. 1, water is pumped from the pool behind Weir 32, and transferred to Broken Hill through a 120-km cast-iron pipeline. The town of Menindee is also supplied from the Darling at Menindee.

During early 2007, the water supply situation in Broken Hill and Menindee was becoming very serious. Following several years of drought, storage levels in both the local reservoirs were low, and the remaining water in storage at Menindee was of steadily declining quality. Without rainfall, the water would need treatment by desalination at some time in 2008. Beyond that, a continuing drought would have required water to be transported to Broken Hill by train, an extremely expensive operation. Although this did not eventually occur, the crisis focused attention on the fragility of the water supply arrangements; furthermore, the December 2007 rainfall has provided only a temporary reprieve.

Flows in the Darling River are highly variable, both seasonally and from year to year. To provide security of supply, it is necessary to allocate and store volumes of water that are large relative to annual consumption. Storage losses, including leakage and, particularly, evaporation, are very high at Menindee, amounting to up to 90% of the storage volume allocated for Broken Hill. Thus to ensure 18 months' supply for Broken Hill, 65 gigalitres (GL) must be stored in Menindee Lakes, of which 58.5 GL is lost. Overall average evaporation losses from the Menindee Lakes System amount to 393–426 GL/year. When the lakes are full, evaporation losses may be up to 700 GL/year.

One option that is being considered for providing short-term water supply is the exploitation of Quaternary and upper Tertiary alluvial aquifers in the vicinity of the Darling River at Menindee. It is also recognised that a possible option for a long-term sustainable supply is use of these aquifers in a managed aquifer recharge system. Previous studies have indicated that two approaches to managed recharge of these aquifers, using induced infiltration from the Darling River and borehole recharge, might be viable. In the Menindee area, aquifer storage would be particularly beneficial because water stored in aquifers cannot evaporate.

THE SITE

The investigations at Menindee have been focused on an area on the southeast side of the Darling River, between the Main Weir at the exit from Lake Wetherell, and Weir 32, 5 km downstream

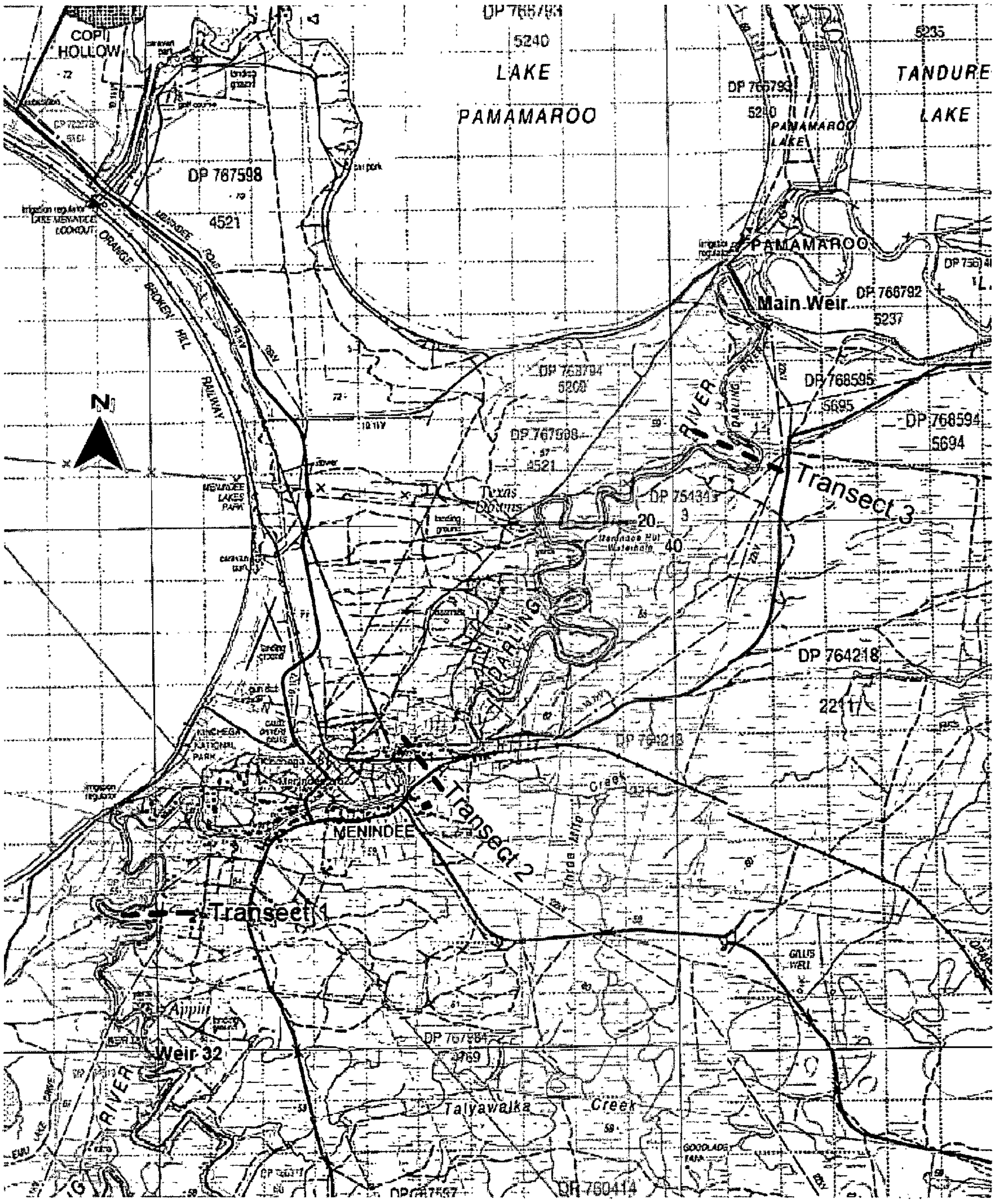

Fig. 1 Infrastructure and investigation locations.

from Menindee. This area is shown on Fig. 1. It is predominantly flood plain, and is a mix of low scrub, with denser vegetation and taller trees close to the river, and open areas. However, there are also substantial areas of irrigated farmland, particularly close to Menindee and between the town and Weir 32. The town itself is predominantly on the northeast side of the river.

Investigation drilling at Menindee was carried out on three transects (T1, T2 and T3), also shown on Fig. 1. A test-production well was later drilled on T2.

HYDROGEOLOGICAL INVESTIGATIONS

During 2007 and 2008, a series of investigations targeting the alluvial aquifers north and, predominantly, south of Menindee were carried out. These have included:

- collation of data from previous investigations carried out by NSW government agencies;
- investigation drilling using eight small-diameter boreholes drilled to depths between 80 and 220 m, located on three transects on the flood plain between the Main Weir and Weir 32;
- geological and geophysical logging, hydraulic conductivity measurement and hydrochemical sampling of these boreholes, some of which were completed as piezometers;
- construction and pump-testing of a test-production well and piezometer;
- a geophysical survey to allow the results of the drilling programme to be projected over a wider area extending 16 km northeast along the Darling River and 2.5 km southeast of the river;
- groundwater flow modelling (MODFLOW 2000); and
- geochemical modelling.

These investigations were located on Transect 2 (Fig. 1). The combined objectives of these investigations were to undertake a detailed assessment of a groundwater supply for the town of Menindee and to enable a preliminary feasibility assessment of a managed aquifer recharge (MAR) scheme.

THE CONCEPT

The idea of storing water underground in natural aquifers is not new. Managed aquifer recharge has been implemented on a large scale in many countries. Typically, water that is surplus to current requirements is introduced to the aquifer through infiltration basins or channels, or by injection through boreholes. Surface waters from lakes and rivers, urban stormwater and reclaimed sewage effluent have all been used as source waters for MAR schemes. Subsequently, and when required, the water is recovered from the aquifer by pumping from boreholes. The option of recovering water from the same borehole that was used for injection has a number of practical advantages and is known as aquifer storage and recovery (ASR).

Another approach that has been widely used in Europe is known as bank filtration or induced recharge. Where a river or lake is connected (in hydraulic continuity) with an aquifer, it is possible to induce flow from the surface water body to the aquifer by controlled pumping from the aquifer. This can provide both over-season storage and, by using the natural attenuation capacity of the aquifer, a beneficial influence on water quality.

Many MAR techniques are only applicable in areas where there is an upper, unconfined aquifer to which water can infiltrate from surface watercourses or engineered structures. Where aquifers are confined by low-permeability layers, as is the case for some of the aquifers at Menindee, only well (borehole) recharge methods can be used.

The crucial water quality parameter for borehole recharge system operation is turbidity, which must be low. Clearly, if the injected water is to be extracted for potable use, it must also meet chemical and bacteriological drinking water quality standards. Even when good quality water is used, injection wells tend to plug, due to the exfiltration of fine particulate matter on borehole walls and in the aquifer immediately surrounding the borehole. When injection rates fall due to plugging, boreholes need to be cleaned or redeveloped. The frequency of redevelopment required is highly dependent upon the quality of the injected water, and has a major impact upon the economics of borehole injection operations.

The distinguishing feature of ASR is that water is injected and recovered from the same borehole. As a consequence, the borehole becomes, to a large extent, self-cleaning, much reducing the need for redevelopment. A second beneficial consequence is that ASR using potable-quality water can often be carried out in an aquifer where the native groundwater is of poor quality, e.g. where salinity is high or the native groundwater has other undesirable chemical characteristics, such as high concentrations of heavy metals or other contaminants. ASR is achievable in such a situation because the injected water forms a "bubble" of high quality water around the well, and it is this water that is then recaptured by pumping. Of course there are constraints—aquifers are not homogeneous; mixing occurs at the margins; and recovery can never be 100% effective; nevertheless, the systems have been found to work well in many circumstances, with reasonable recovery ratios. Often, recovery ratio improves over a number of injection/pumping cycles (Pyne, 2005).

In general, the key criteria to be addressed in making an initial assessment of the feasibility of ASR are:

1. Is the formation sufficiently permeable to permit injection and recovery at reasonable (and reasonably economic) rates?
2. Is sufficient storage capacity available?
3. Is the aquifer geometry such as to provide adequate containment, and thus recovery of injected water?
4. Are the native groundwater and proposed recharge source water chemically compatible.

An MAR project proposed at Menindee would involve injecting fresh surface water derived from the pool behind Weir 32 on the Darling River into confined sand aquifers. The concept is to take the freshwater that flows down the river following rainfall in the upper catchment, and which can be kept in the existing weir-pool and in the Menindee lakes for many months before the quality deteriorates, and to inject this through boreholes to progressively build the storage of fresh water in the aquifers. Once in the ground, the water can be stored for many years without evaporative loss or deterioration, and it can then be recovered through the same boreholes (ASR wells).

Potential problems

Although the concept is relatively simple, experience has shown that many potential problems must be considered and, if necessary, overcome, if ASR schemes are to be technically and economically successful. Two issues that have been common problems elsewhere, and have been identified as potential concerns at Menindee, are the physical quality of the source water, and chemical reaction of the injected water with aquifer materials.

As indicated previously, ASR is sensitive to the suspended solids content of the source water. The water in the weir pool contains suspended clays carried down from the upper catchment and is often most turbid at the times when its salinity is lowest and therefore optimal for recharge. During the summer months the water may have a significant content of algae. Suspended particles, whether organic or inorganic in origin, can cause rapid clogging of the injection wells and must be removed by filtration prior to injection.

In some parts of the alluvial aquifers the aquifer grains are coated with (or cemented by) iron and manganese oxides or sulphides, which may include as impurities other heavy metals such as arsenic. The mobility of all of these metals is dependent upon the acidity and oxidation potential of the groundwater. Injecting surface water into the aquifer will result in changes to these parameters, and possibly mobilisation of metals from the aquifer matrix.

Enhancing the concept

It is essential that the source water for an ASR project have very low suspended solids content. This can be achieved by pre-filtering the water prior to recharge. Two alternatives have been considered: conventional water treatment using rapid sand filtration, and bank filtration.

Rapid sand filtration is a proven water treatment technology, but would require the installation of a number of filter units, probably on the south side of the river, close to the centre of the proposed well field. Land acquisition, power supply and sludge disposal are all likely to be potentially difficult, but solvable, problems.

Bank filtration involves pumping water from boreholes constructed in the shallow aquifer close to the river, utilising the riverbed and banks as natural filtration media. This technique has been widely used in Europe, particularly in Germany, and has been proven to retain its effectiveness for many years in appropriate geological conditions. Generally, at Menindee, the shallow Coonambigal Formation aquifer is discontinuous and not well suited to groundwater storage, but locally it has high permeability, forms the natural link between the river and the aquifer system, and should be suitable for bank filtration. One area that has been investigated and may be suitable is just to the south of Menindee, where a thin tongue of land is surrounded by the river on three sides (Fig. 1).

RESULTS OF INVESTIGATION

Drilling programme

The drilling and logging data for the three Menindee transects indicated that:

- At least two aquifer units—an upper (Plio-Pleistocene) aquifer and a lower (mid-Tertiary or Renmark age) aquifer—are present at Menindee.
- The Plio-Pleistocene aquifer includes at least 4 m of coarse sand and gravel that are assessed to be of high permeability, and a substantially greater thickness of fine sands and silty/clayey sands that are likely to have lower permeability but significant storage capacity. The potentiometric surface of this aquifer is about 10 m below the ground surface.
- The Plio-Pleistocene aquifer has reasonable lateral continuity.
- The groundwater contained in part (but not all) of this aquifer is of potable quality.
- The low salinity and chemical character of groundwater in several of the boreholes indicate that there is likely to be hydraulic continuity with water in the weir pool, which may be the primary source of recharge to the aquifer at this location.

The drilling and testing data thus indicated that the Plio-Pleistocene aquifer has sufficient thickness and continuity, and adequate hydraulic conductivity, to justify further testing to demonstrate suitability as a drought groundwater supply, and also to provide further data for the assessment of its suitability for long-term use as part of a managed recharge project. The data also showed that, in substantial parts of the aquifer, water quality is suitable for drought water supply. Whilst there are also areas of poorer quality, where the groundwater is not suitable for direct use, poor-quality water does not preclude the use of managed aquifer recharge approaches. One of the benefits of managed aquifer recharge is the long-term improvement of the quality of water in storage.

Pumping test

Analysis of the pumping test data derived the following hydraulic parameters:

- Transmissivity: 140 m^2/d
- Hydraulic conductivity: 15.6 m/d
- Storativity: 2.9×10^{-4}

Geophysics

A towed transient electromagnetic (TEM) array was used for this survey. The TEM survey successfully identified thick features of variable conductivity in layers to a depth of approximately 50 m, with inference of variability of deeper layers. The features appear to be relict palaeochannelrelated sands enhanced by recharge infiltration conduits, but no clearly continuous meandering features have been identified. Highly resistive sediment related to the present Darling River Channel does not extend far from that channel and therefore does not provide a large storage volume. Larger deeper aquifers are inferred to the northeast of the town.

Groundwater quality

The results of groundwater sampling and analysis during the 2007 programme show that salinity increases from west to east on Transect 1. Logging of T1:0701 indicated a predominantly sandy profile, and it is likely that there is at least partial hydraulic connection between the aquifer and the river. This well is screened in the upper part of the 30-m (Shepparton) aquifer. The water sampled from T1:0702 and T1:0703, screened across deeper horizons than T1:0701, is brackish.

On Transect 2, the only monitoring well sampled in 2007, T2:0703, had potable-quality water. The wells on T3 yielded groundwater of potable quality, apart from elevated arsenic in T3:0701.

The limited data available from earlier studies generally indicate that groundwater in the shallow aquifer is of variable quality and that salinity increases with distance from the river.

During the 2008 testing programme at the test site on Transect 2, the electrical conductivity and four other field physio-chemical parameters (dissolved oxygen, pH, temperature and redox potential (ORP), were regularly monitored. Samples for full laboratory analysis were also obtained during the test. In summary, the water produced during the pumping test can be described as a sodium-chloride dominant water of potable salinity (EC 1280 μS/cm, implying a total dissolved solids content of about 900 mg/L). The water is moderately reducing and of near-neutral pH.

Arsenic was measured in the groundwater at concentrations exceeding the guideline value for drinking water (Australian Drinking Water Guidelines, 2004). The concentrations reported were on a declining trend over the duration of the pumping test, and range from 0.03 to 0.02 mg/L compared to the Australian guideline value of 0.007 mg/L.

Manganese and iron were also reported at concentrations of 0.87 and 0.69 mg/L, respectively. These concentrations exceed the drinking water guidelines. The dissolved iron could be removed by aeration (possibly with some pH adjustment) and precipitation. It is likely that manganese and arsenic would co-precipitate with the iron. Thus these concentrations need not pose a constraint on the use of this groundwater for water supply purposes, as economic treatment is possible. However, the source of the arsenic, and groundwater geochemistry in general, requires further assessment if managed aquifer recharge is to be considered. Concentrations of all other analytes were within health guideline values for drinking water supply.

AQUIFER GEOCHEMISTRY

Heavy metals

The heavy metals/metalloids of interest are iron, manganese and arsenic. Each of these substances can exist in multiple oxidation states and form complexes with other species present. Their solubility therefore depends upon pH and redox potential, as well as the concentrations of other species.

The predominant sulphide is pyrite (FeS_2). Arsenic (valency state As −1) commonly substitutes for sulphur in a small proportion of this mineral, forming arsenian pyrite (FeAsS). The geochemical processes controlling arsenic mobility are complex. Hinkle & Polette (1999) describe these processes in detail, and the following discussion is largely derived from that source.

Two categories of processes largely control arsenic mobility in aquifers. These are adsorption and desorption reactions (involving the surfaces of minerals such as iron oxides and clays), and solid-phase precipitation and dissolution reactions. Arsenic adsorption and desorption reactions are influenced by changes in pH, occurrence of redox (reduction/oxidation) reactions, presence of competing anions, and solid-phase structural changes at the atomic level. Solid-phase precipitation and dissolution reactions are controlled by solution chemistry, including pH, redox state, and chemical composition. Geochemical conditions measured in the aquifer at Menindee straddle the arsenate/arsenite stability field boundary. This finding is significant because arsenite is less strongly adsorbed to iron oxide coating on aquifer grains than arsenate, and is soluble over a wider pH range. Thus relatively small changes in redox potential may have a significant effect on the dissolved-phase concentration of arsenic.

GEOCHEMICAL MODELLING

Geochemical modelling was carried out to further assess the potential for mobilisation of arsenic and other metals during ASR operations, and potential reactions between the aquifer water and injected surface water from the weir-pool. The results of geochemical modelling are summarised in Table 1. It can be seen that concentrations decrease or increase with mixing, depending on relative concentrations in the aquifer and river water. Concentrations of components that are strongly sorbed change more slowly than non-sorbed components. Arsenic shows almost no concentration change reflecting the high proportion of total mass that is in the sorbed phase.

Table 1 Geochemical modelling—primary case.

Primary fluid	Equilibrium phases	Mixing fluid	Mix	Observations
Native groundwater (Analysis A)	None	None	N/A	Primary arsenic species is arsenite (H_3AsO_3) (99%). Primary iron species are Fe^{2+} (50%), $FeHCO_3^+$ (45%) and $FeCO_3$ (5%). Primary manganese species are Mn^{2+} (68%), $MnHCO_3^+$ (24%) and $MnCO_3$ (8%). Solution is oversaturated with respect to goethite and slightly oversaturated with respect to siderite and rhodochrosite, undersaturated with respect to amorphous iron hydroxide.
Native groundwater (Analysis A)	$Fe\text{-}OH_{(ppt)}$ $Goethite_{(s)}$ $Siderite_{(s)}$ $Rhodochrosite_{(s)}$	None	N/A	Minor precipitation of rhodochrosite and significant dissolution of amorphous iron hydroxide. Small drop in pH and Eh.
Native groundwater (Analysis A)	$FeH_2AsO_{3(sorbed)}$ $FeH_2AsO_{4(sorbed)}$ $FeHAsO_{4(sorbed)}$ $FeOHAsO_{4(sorbed)}$ + other sorbed species	None	N/A	Mass of arsenic in adsorbed phase calculated as much greater than that dissolved phase, thus total mass in system is also much greater than that measured in the dissolved phase.
River (fresh)	none	None	N/A	Water in natural equilibrium with calcium and magnesium carbonate minerals.
Native groundwater	$Fe\text{-}OH_{(ppt)}$ $Goethite_{(s)}$ $Siderite_{(s)}$ $Rhodochrosite_{(s)}$ $FeH_2AsO_{3(sorbed)}$ $FeH_2AsO_{4(sorbed)}$ $FeHAsO_{4(sorbed)}$ $FeOHAsO_{4\,(sorbed)}$ + other sorbed species	River (fresh)	Progressive mixing of river water with aquifer water in ten stages, from 10:90 to 95:5	Progressive reduction in concentration of chloride from groundwater concentration to just above river concentration. Much slower change in concentration of other ions due to buffering effect of adsorbed phase. Little change in arsenic concentration. Some precipitation of goethite. Eh remains posed at close to groundwater value

PRELIMINARY FEASIBILITY ASSESSMENT

A preliminary assessment of the feasibility of a MAR scheme at Menindee has been completed. This feasibility assessment was based on the framework outlined in the May 2008 draft of the "*Australian Guidelines for Water Recycling – Managed Aquifer Recharge*" prepared by the

Environment Protection and Heritage Council, National Health and Medical Research Council and Natural Resource Management Ministerial Council. Although the focus of these guidelines is on MAR using recycled water, many aspects are relevant to a scheme based on the use of surface water.

The preliminary feasibility assessment and associated risk assessment have indicated that a MAR scheme at Menindee should be feasible, and that all assessed risks lie within an acceptable range. To prove the viability of the concept, further investigation and preliminary design work are required.

On the basis of the data acquired in this investigation, it is considered likely that an aquifer that is sufficiently permeable and extensive to be capable of functioning as an aquifer storage and recovery reservoir is present at Menindee. Filtration of the river water will be required prior to injection; methods of achieving this are being assessed. Whilst some potential geochemical constraints have been identified, the geochemical modelling that has been carried out to date indicates that these are manageable. Further geochemical and groundwater flow modelling is in progress.

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The evaluation of groundwater environmental restoration by artificial recharge in Pingtung Plain, Taiwan

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Abstract The Pingtung Plain, Taiwan, has abundant groundwater resources. The overall study objective focuses on groundwater restoration. In this study, MODFLOW was applied to evaluate the groundwater recharge dynamics between the recharge in the upstream reaches and pumping in the downstream reaches. Using the simulation model, the results show that storage of the groundwater increased by 683 266 m^3/year and 2 471 765 m^3/year and the groundwater table has risen 0.91 m and 1.21 m in Kaoping Lake and Wanlung Lake, respectively. From the view of storage change, the Wanlung Lake was found to be the most suitable area for constructing the artificial lake for artificial recharge. The variation of the groundwater storage of the aquifer is an important factor in artificial recharge. Managing artificial groundwater recharge can provide for shortages of good quality surface water during flood season, caused by higher turbidity, and conjunctive water use of surface and groundwater resources in the future.

Key words artificial recharge of groundwater; MODFLOW; simulation

INTRODUCTION

Pingtung Plain is located in southern Taiwan and extends approximately 1210 km^2. There is a very high annual rainfall in this area; however, the amount of rainfall received varies significantly over the course of the year. In terms of measurement data, 90% of the annual precipitation falls during the rainy season (i.e. May to September) while only 10% of the rainfall occurs during the dry season (October to April). This uneven distribution in the monthly rainfall poses a major problem to the planners involved in the protection and utilization of water resources in Pingtung Plain. As a result of the hypsographical variation, the water resources are not readily stored into aquifer, but tend to flow rapidly into the ocean. Consequently, the utilization rate of surface water resources is limited in Taiwan. However, the thriving aquaculture found along the southwestern coastal areas of Taiwan is dependent on availability of abundant freshwater. In most cases, this freshwater has been supplied by the intensive pumping of groundwater. This has led to a drop in the local groundwater level, and in severe cases has resulted in land subsidence and seawater intrusion.

The results of previous studies in artificial recharge of groundwater have indicated that the groundwater environment can be improved and restored (Bouwer, 1999; Ting *et al.*, 2002; CTCI, 2002; Yang, 2006). The overall study objective focuses on groundwater restoration. In this study, MODFLOW was applied to evaluate the groundwater recharge dynamics between the recharge in the upstream reaches and pumping in the downstream reaches.

STUDY AREA AND METHODOLOGY

Geographical location

Pingtung Plain is located in southern Taiwan. There are nine groundwater sub-regions from north to south, and these are mostly in the south (SRWRO, 2005). It mainly comprises the flat area of the catchments of the Kaoping, Tungkang and Linpeing rivers. The regional area covers in total 25 counties in Pingtung and Kaohsiung cities. This elongated plain covers an area of 1210 km^2, being 22 km wide in an east–west direction and 55 km along the north–south direction. The groundwater sub-region of Pingtung Plain is mainly in the east and close to the Central Mountain (Fig. 1).

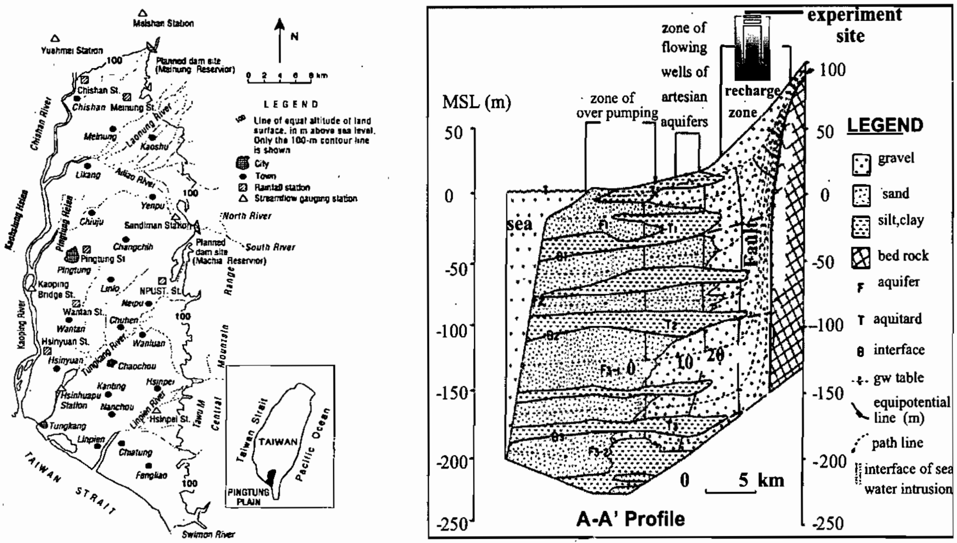


Fig. 1 Location of study area in Pingtung Plain, Taiwan, and geological profile (Ting, 1997).

Climatic regime

The climate of Pingtung Plain is sub-tropical; rainfall is alternately affected by typhoons in summer, producing most of the rainfall. The rainy season lasts from May to September. The groundwater of Pingtung Plain mainly comes from rainfall then through infiltration into the aquifer in the mountains and, consequently, the groundwater table is higher during the rainy season.

The maximum average rainfall is 3146 mm, 2980 mm, 2246 mm, 2152 mm, 2249 mm and 2460 mm in the Laonung, Ailliao, Chishan, Kaoping, Tungkang and Linpien river basins, respectively. The maximum average wind speed is 5.0 m/s in November and the minimum is 2.6 m/s. Average annual temperature in Pingtung is 25.2°C. Average annual evaporation is about 1723 mm.

Surface water

The Pingtung Plain mainly comprises flat areas of the Koaping, Tungkang and Linpien river catchments, as well as several small streams; the three main rivers finally flow into the Taiwan Strait. In the *Yearbook of Hydrology* (Water Resources Agency, 2007) the Ministry of Economic Affairs shows the average annual discharges as approx. 6330 Mm^3 for Koaping River, 952 Mm^3 for Tungkang River and 728 Mm^3 for Linpien River. Other parameters are given in Table 1.

In this study, MODFLOW was applied to evaluate the groundwater recharge dynamics between the recharge in the upstream reaches and pumping in the downstream reaches. The results from the groundwater model are then used to manage and deploy the water resources of Pingtung Plain, as below:

Table 1 River and hydrological characteristics in Pingtung Plain (WRA, 2007).

River	Inflow Location	Elevation (m)	Outflow location	Length (km)	Average slope	Basin area (km^2)	Average rainfall (mm)	Average discharge (Mm^3)
Koaping	Mt. Morrison	3997	Hsinyuan	171	1/150	3 256	2 547	6 330
Tungkang	Ailliao	1138	Tungkang	44	1/500	472	2 093	952
Linpien	Central Mt.	2880	Linpien	42	1/15	344	3 314	728

Model description

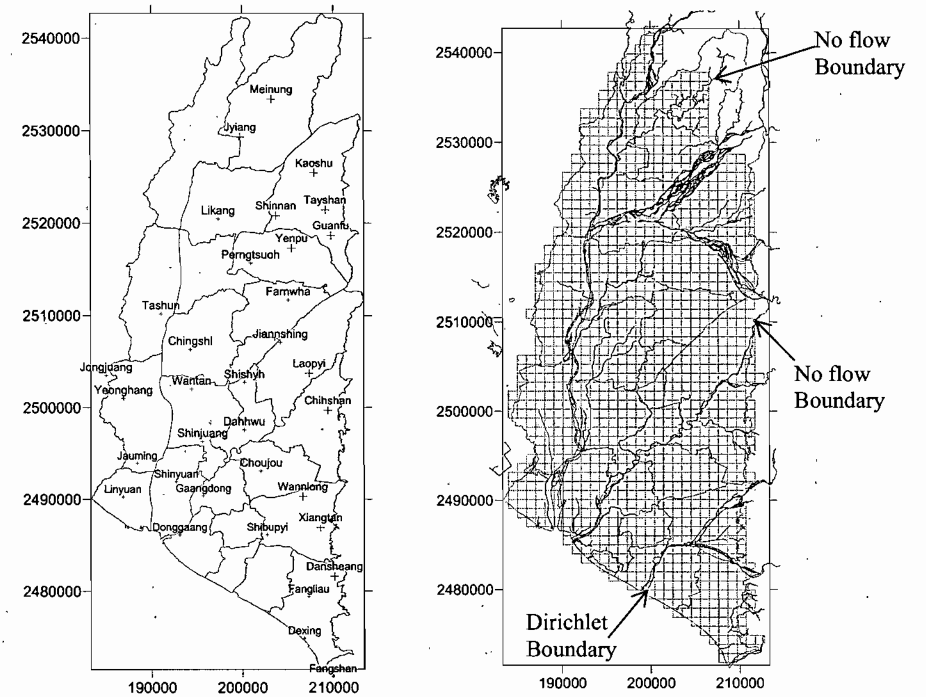
Defining a conceptual structure of the groundwater system is a necessary prerequisite to numerical simulation (SRWRO, 2007). The main purpose is to simplify the field problem and frame the associated field data through the conceptual model to create a rational quantification with hydrogeology and hydrological stresses (Ting, 1997). The hydrological stresses which need to be considered when simulating groundwater flow in the plain include: abstraction from wells, precipitation, recharge, interaction with the river, evapotranspiration and boundary conditions.

Model construction

An important tool to characterize the aquifer is its hydrogeological profile, which was prepared by the central Geological Survey and Ministry of Economic Affairs, WRA (2001). Figure 2 shows the observation wells; in total there are 52 aquifer monitoring stations and 127 wells set up in Pingtung Plain which provide the hydrogeological and observation data for the conceptual model.

Model grid and aquifer system

There are seven layered hydrogeological units in the aquifer, reaching down to about 220 m depth, to relate to the aquifer system data (Table 2) and observation wells as Fig. 2(a). Actually, the conceptual model can be simplified into a three-layer system: the first layer was defined as unconfined aquifer, and the second and third were defined as unconfined/confined interactive aquifer. Based on data availability and hydrogeological conditions, the grid spacing in both the x and y directions is 1000 m. The aquifer system is bounded by mountains in the north and east, by Funshin Hill in the west and by Taiwan Strait in the south, as in Fig. 2(b).



(a) Observation wells (b) Boundary conditions

Fig. 2 Location of observation wells and boundary conditions in Pingtung Plain.

Table 2 The depth of hydrogeological layers in study area (unit: m).

Toponym	Layer: F1	T1	F2	T2	F3-1	T3	F3-2
Linyan	44.5	22	14.5	24	68	0	57
Sipu	37	13	54	10	52	0	0
Jongjuang	54.5	0	65.5	0	78	6	6
Chingsi	47.5	0	57.5	4	89	6	6
Datan	40	5.5	44.5	15	70	7	48
Sinjhuang	31	27	43	0	81	0	28
Likang	44.5	0	79.5	0	81	0	5
Wanluan	53.5	5	54.5	13.5	49.5	22	12
Jiansing	51	0	64	0	83	3	11
Xishi	36.5	8.5	63	0	65.5	16	21.5
Shibupyi	23.5	23.5	45.5	23	62.5	18	15
Wannlong	80.5	6.5	50.5	0	72.5	0	0
Shinuan	26.5	18.5	38	12	80	0	35
Keefung	30	19	42	10	83.5	0	45.5
Majia	83.5	0	62.5	0	64	0	0
Chanliao	57	0	47	0	57.5	19	39.5
Yeonghang	62	0	41.5	6	70.5	4	36
Chunchou	43	0	79.5	0	82.5	0	15
Meinung	43	0	27	0	0	0	0
Haifong	52.5	0	62.5	0	89	6	10
Choujou	27	31.5	47.5	0	74	0	40
Konggang	33	29	22.5	16.5	73	0	46
Wantan	52	0	48	4.5	70	25.5	20
JiuRu	47	0	65	8	84	0	16
Yenpu	65	0	67	0	78	0	0
Kaoshu	80	0	72	0	46	0	0
Tayshan	65	0	70	0	75	0	0
Nei-Pu	45	7.5	59.5	0	68	7	23
Laopi	45	10.5	61.5	0	73	0	20
Dansheang	86	0	9	0	0	0	0
Dansheang	67	9.5	59.5	0	4	0	0
Fangshan	73	0	51	0	80	10	6
TaChung	47.5	8.5	51	0	84	0	29
Kanding	37.5	17.5	30	23.5	76.5	0	35
Fangshan	36	0	0	0	0	0	0

RESULTS AND SIMULATION

Water balance

The results of the water budget are given in Table 3. In term of the results, the well abstraction was approx. 1330 Mm^3/year. The river leakage into the aquifer was approx. 201 Mm^3/year, and the aquifer outflow to the river was approx. 167 Mm^3/year. The water balance clearly indicates that recharge is principally derived from precipitation on the plain and lateral inflow from numerous mountains and slopes during the rainy period.

Case study

The purpose of the case studies is evaluation of recharge so that we obtain the best simulations for the well abstraction study. Because of the artificial recharge of groundwater in the upstream area, the groundwater can be used downstream. The sites chosen for simulation are Kaoping Lake site and Wanlong Lake site, as shown in Fig. 3.

Table 3 Groundwater balances for Pingtung Plain in 2000 (m^3/year).

-	Inflow	Outflow
Storage change	827 284 670	889 898 716
Outside area	16 660 865	7 395 217
Well abstraction	0	1 338 587 726
Lateral inflow mountain	959 322 179	0
Precipitation	447 691 061	0
Evapotranspiration	0	48 501 500
River exchange	201 306 593	167 298 860
Total	2 452 265 369	2 451 682 019

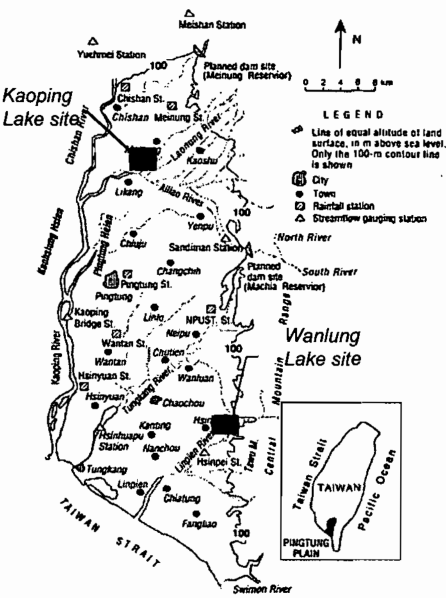


Fig. 3 Locations of Kaoping Lake and Wanlung Lake.

Case study A

In study A, the hypothesis is of artificial recharge in the rainy season (June–August) on Kaoping Lake; the simulation condition was 345 000 m^3/day recharged into the aquifer. In addition, these studies hypothesize that there are two flow lines of aspect to analyse water level variation in downstream, then to evaluate the water balance, as shown in Fig. 4. The groundwater variation was effected by the artificial recharge, with an average rise of 0.91 m and the aquifer storage increased by 683 266 m^3/year, as shown in Table 4 and Fig. 5, respectively.

Case study B

In the case of study B, the hypothesis is implementation of artificial recharge in the rainy season (June–August) on Wanlung Lake; 400 000 m^3/day recharged into the aquifer. In addition, these studies hypothesize that there are three flow lines of aspect to analyse water lèvel variation in downstream, then to evaluate water balance by the way, as shown Fig. 6. The groundwater variation was effected by the artificial recharge; the average rise was 1.21 m and the storage change was an increase of 2 471 765 m^3/year in the aquifer. The results are illustrated in Table 5 and Fig. 7.

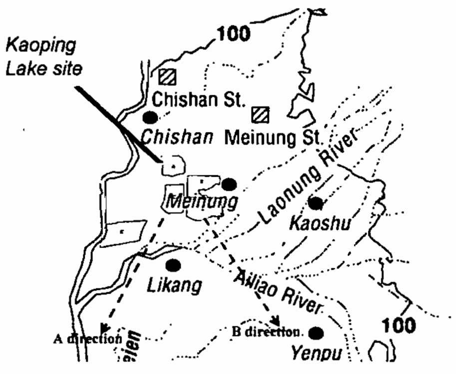
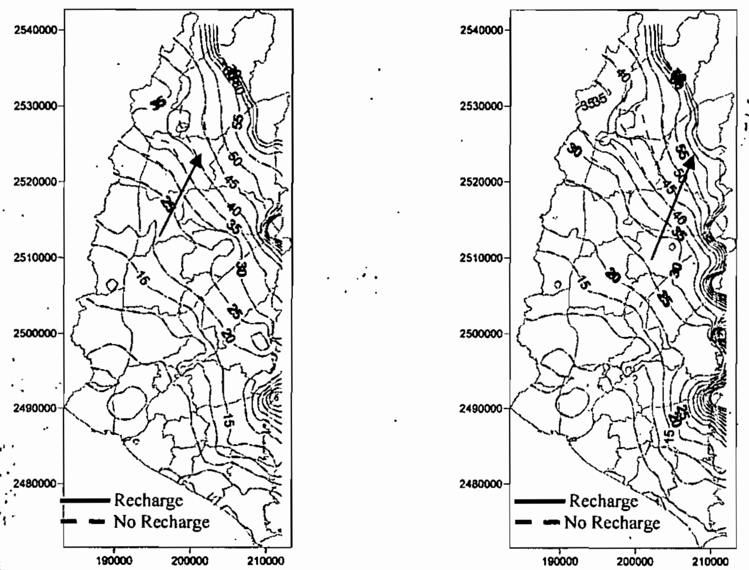


Fig. 4 Location of Kaoping Lake in Pingtung Plain.

Table 4 The variation of groundwater in downstream (unit: m).

Flow lines	Position of observation	June	August	October	December
	Downstream 2000m	1.16	4.43	3.45	2.4
A	Downstream 5000m	0.24	1.01	1.32	1.33
	Downstream 10000m	0.078	0.145	0.285	0.427
	Downstream 2000m	1.66	4.18	2.99	2.22
B	Downstream 5000m	0.065	0.46	0.73	0.79
	Downstream 10000m	0.014	0.024	0.102	0.175



June July

Fig. 5 Contour maps of recharge for layer 1.

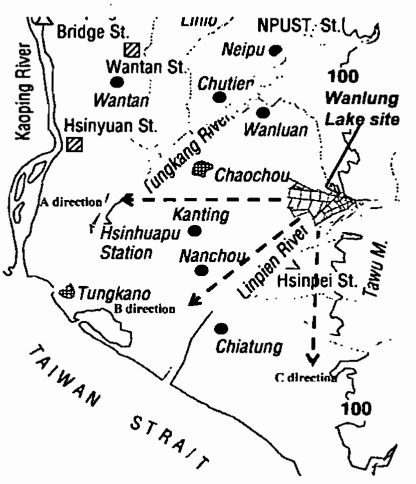


Fig. 6 Location of Wanlung Lake in the south of Pingtung Plain.

Table 5 the variation of groundwater in downstream (unit: m)

Flow lines	Position of observation	June	August	October	December
	Downstream 2000m	2.98	3.34	1.62	1.12
A	Downstream 5000m	0.67	1.52	1.23	0.94
	Downstream 10000m	0.0019	0.023	0.073	0.132
	Downstream 2000m	1.91	2.45	1.51	1.09
B	Downstream 5000m	0.06	0.38	0.59	0.64
	Downstream 10000m	0.0012	0.009	0.026	0.05
	Downstream 2000m	3.02	2.85	1.65	1.15
C	Downstream 5000m	0.34	1.0	1.04	0.93
	Downstream 10000m	0.05	0.23	0.35	0.41

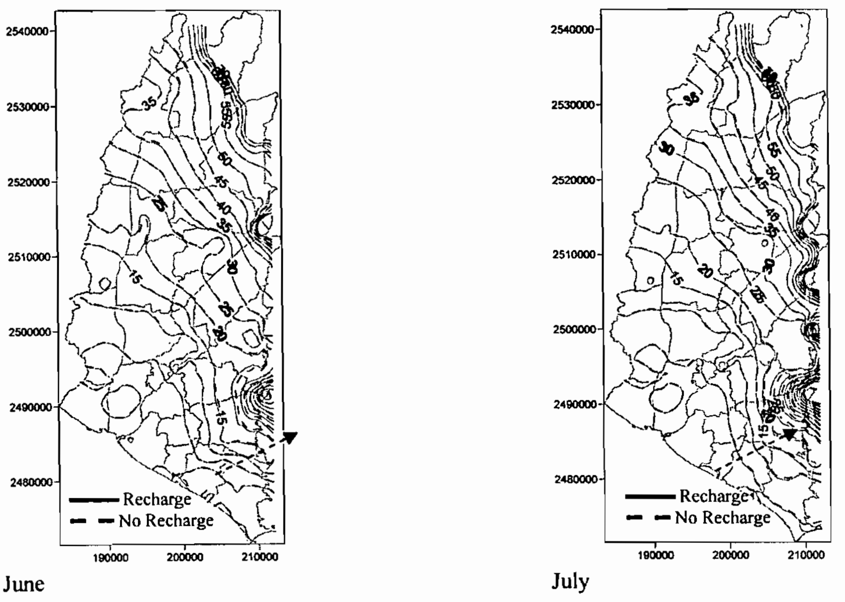


Fig. 7 Contour maps of recharge for layer 1.

CONCLUSIONS

Using the simulation model, the results show that the storage of the groundwater increased by 683 266 m^3/year and 2 471 765 m^3/year and groundwater table has risen by 0.91 m and 1.21 m in Kaoping Lake and Wanlung Lake, respectively. From the point of view of storage change, Wanlung Lake was the most suitable area to construct the artificial lake, and more so than Kaoping Lake. The variation of groundwater and storage in the aquifer are important factors in artificial recharge.

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2 Water for Food

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Impact of irrigation intensification on inter-sectoral water allocation in a deficit catchment in India

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Abstract Agricultural intensification across peninsular India has resulted in an increase in the demand for irrigation water leading to unsustainable extraction of surface and groundwater resources. This paper presents the case study of inter-sectoral conflicts arising from depleting resources in the Malaprabha sub-basin in Karnataka, India. Due to the changes in agricultural practices, drastic changes are happening in the hydrological regimes across the catchment including reduced streamflow and groundwater depletion. Consequently, this results in acute annual and seasonal scarcity of drinking water in urban and rural settlements. Hydrological modelling of the catchment is carried out using the ArcView integrated Soil and Water Assessment Tool (AVSWAT) to simulate streamflow. In order to study the anthropogenic impacts, various land-use scenarios are identified after considering various socio-economic and institutional constraints in the catchment. Considering the irrigation and drinking water supply requirements, the paper also presents outcomes of land-use changes on inter-sectoral water allocation that essentially prioritizes the drinking water demands of an urban settlement.

Key words water scarcity; irrigation; inter-sectoral water allocation; land-use change

INTRODUCTION

Agriculture is the largest freshwater consuming sector, accounting for more than 80% of the freshwater usage over the world. Population growth and the development of agro-based industries have increased the demand for agricultural produce. The growing market for agro-related products is now acting as an important driving force for increasing agricultural production. In addition to the shift from traditional cropping varieties to high-yielding varieties, that is happening in most parts of the world, a shift from rainfed to irrigated crops is also happening mainly in view of increasing the crop yields. Traditional crops which were cultivated to meet household food demands are gradually replaced by cash-crops like sugarcane. As a result more and more areas, which were traditionally rainfed, are now brought under intensive irrigation. The consequence is over-extraction of limited and less-renewable groundwater resources. Many of these river basins and aquifer systems are over-developed and are now facing severe water stress.

With increasing population, and fast economic and industrial growth, the problem of resource depletion and water scarcity is more severe in India. Since independence, the irrigated area in India has quadrupled. Availability of subsidized electric power and easy access to funds played a major role in increasing the number of bore wells and dug wells many fold after 1950 (Mall *et al.*, 2006). In the majority of the river basins in India, the current extraction rate is 50–95% of the total utilisable surface water resources (Majumdar, 2008).

This paper presents a case study from the Malaprabha catchment in India where large-scale changes are happening in agricultural practices. The impact of these changes on water resource availability is presented in this paper from the perspective of increasing inter-sectoral water allocation issues. A hydrological model is set up for the catchment to understand the impacts of current trends in agricultural and irrigation practices of the area on the availability and access to water resources.

LOOMING WATER SCARCITY IN MALAPRABHA CATCHMENT

Malaprabha catchment in Belgaum district, Karnataka, India, is one of the deficit sub-basins of the Krishna River basin. The Malaprabha Dam was commissioned in 1974 primarily to meet the

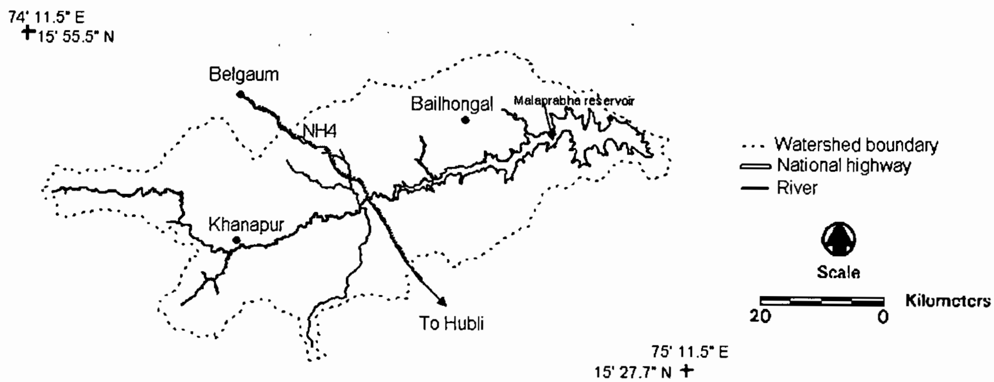

Fig. 1 Base map of the upper Malaprabha catchment.

irrigation and drinking water requirements. It has a gross and live storage capacity of 1070×10^6 m^3 and 830×10^6 m^3, respectively. The total area of the Malaprabha catchment (up to the Malaprabha Dam) is 2204 km^2 (Fig. 1). The area is characterized by the heterogeneity in topography, hydrometeorology as well as soil and land-use conditions. Hydrologically the catchment can be divided into three zones: zone 1 (upper parts of the catchment), zone 2 (middle part) and zone 3 (lower parts). Zone 1 and parts of zone 2 are characterized by hilly terrain and gravelly soil, cracking clay or loamy soil that assists percolation. Fairly dense forest covers almost 60% of zone 1. Zones 2 and 3 are dominated by agricultural areas, which were traditionally under rainfed cultivation. Annual average rainfall in the area varies from 2000 mm in zone 1 to 500 mm in zone 3.

Due to the variation in the hydro-meteorological characteristics, water availability in the lower catchment is closely related to the land-use practices and water extraction in zones 1 and 2. With the establishment of several sugarcane factories in the area, demand for sugarcane production has increased drastically during the last two decades. Vast areas of rainfed agricultural fields have been converted into sugarcane. On the other hand, the annual average rainfall of the area, particularly in zone 3 and lower parts of zone 2, being insufficient to meet the water demand, the newly introduced crops are supported by irrigation from the Malaprabha River or from groundwater sources. According to the statistics of the Karnataka Government, irrigated area in the Belgaum district has been increased from 10% in 1970 to 30% of the total cultivable land in 2003 (Statistical Abstract, Government of Karnataka, 2004). Due to over-extraction, groundwater resources are depleting at a fast rate resulting in reduced flow in the river during the post-monsoon season. Over-extraction of water in the upper catchments is causing water stress in the lower catchment leaving even the drinking water supply in crisis. Due to unplanned and unregulated land-use changes in the catchment, fair water allocation between the irrigation and the drinking water sectors is increasingly becoming difficult.

Long-term rainfall data for 18 stations distributed over the entire catchment (obtained from the Directorate of Economics and Statistics), streamflow observations at the Khanapur gauging station (collected from the Irrigation Department, Govt Karnataka) and reservoir inflow data for the Malaprabha reservoir (from the Karnataka Neeravari Nigama Ltd, Navilutheertha) are analysed to understand the changes happening to the rainfall–runoff process in the catchment. The streamflow was found to be decreasing in the last two decades. Though a corresponding reduction in rainfall is also observed, the average streamflow recorded at the reservoir during the period 2000–2003 was found to be much less than the period 1972–1975 when the average rainfall was similar. Further, the runoff coefficient (C), which is the ratio of runoff to rainfall, was found to be decreasing in recent years (Fig. 2). Thus it is found that reduction in the streamflow is not so much due to rainfall change, but due to changes in the land-use and irrigation practices.

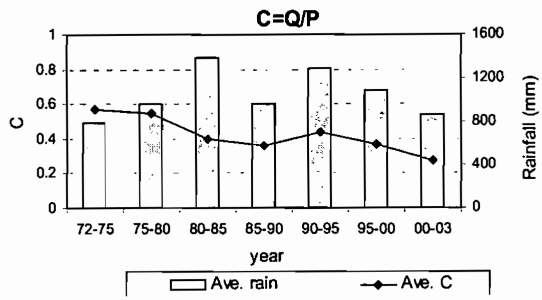

Fig. 2 Variation of rainfall and runoff coefficient for the Malaprabha catchment.

DESCRIPTION OF THE HYDROLOGICAL MODEL

The objective of the hydrological modelling is to set up a catchment-scale model to simulate various hydrological processes in the catchment. Modelling of the hydrological processes taking place in the catchment under the current land-use scenario and the impact of any possible change in land-use practices, on the streamflow, are the major interests of the current study.

In this study, a hydrological model of the area is set up using the ArcView integrated Soil and Water Assessment Tool (AVSWAT), which has the capability to incorporate spatially-distributed data, a reasonably user-friendly interface and is affordable for similar implementation in developing countries (Badiger & Bakken, 2007). The Soil and Water Assessment Tool (SWAT), developed at the Blackland Research Centre (Arnold *et al.*, 2000) is based on conceptualizing the physical processes in a watershed. It is a simple, but robust model which simulates various components of the hydrological cycle from precipitation to deep aquifer recharge. The model is developed for continuous time series with daily time steps, which can be used for estimating the long-term impacts of watershed management programmes. In this study, a version of SWAT integrated with ArcView, called AVSWAT (Di Luzio *et al.*, 2002) is applied to use spatially referenced data, thereby facilitating modelling with spatially varying parameters.

SWAT simulates various hydrological processes, viz., surface runoff, infiltration, direct evaporation from soil, plant transpiration, soil moisture storage, percolation, shallow and deep aquifer recharge, revaporation, return flow, transmission losses, bank storage, retention storage, ponds, reservoirs and wetlands, as well as water routing through the channels. The whole process can be considered in two phases: a land phase and a routing phase. In the land phase, SWAT considers water storage in four different layers, viz., canopy storage, root zone storage, shallow aquifer storage and deep aquifer storage (Fig. 3).

SWAT identifies hydrologically similar units in the catchment by using the soil and land cover information. These small units are called hydrological response units (HRUs). Water yield is estimated for each HRU separately. The sum of the water yields from each HRU in a sub-basin and the stream discharge from the upstream river reach are taken as the inflow to any channel reach. Once the water reaches the channel, it is routed to the watershed outlet allowing for transmission losses, direct evaporation and pumping for agricultural or domestic use (Neitsch *et al.*, 2005).

With its GIS framework and user-friendly interface, AVSWAT facilitates the input of geographically referenced data as well as the hydrometeorological data at multiple stations within the catchment. In order to facilitate modelling incorporating spatially-distributed data, the catchment is partitioned into small units called sub-basins and HRUs. By using the DEM, the model identifies the catchment area that drains through the specified outlet point. Further sub-basins and HRUs are delineated based on the user-specified outlet points within the catchment. Finally the output is generated in the form of tables showing the various components of the hydrological cycle in the land phase (both sub-basin and HRU level) and along different channel reaches.

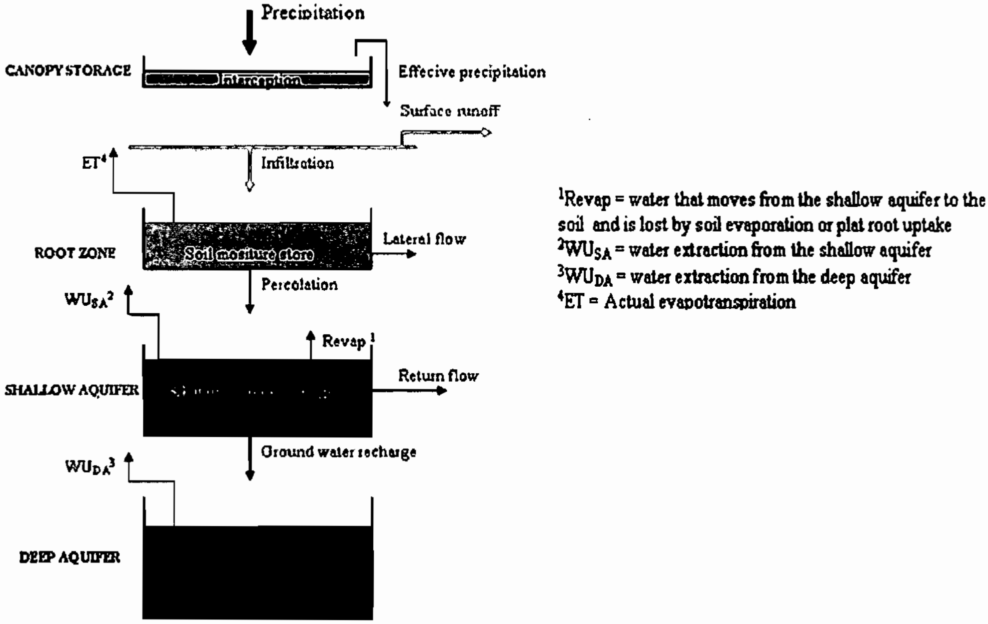

Fig. 3 Schematic representation of the hydrological processes in SWAT at the land phase.

Model application to the Malaprabha catchment

The model was applied to the Malaprabha catchment to simulate the current hydrological processes as well as the impacts of land-use changes on the streamflow. Shuttle Radar Topography Mission (SRTM) data at 90-m spatial resolution were used to generate the digital elevation model (DEM) of the area. The digital channel network of the catchment was generated by digitizing the major stream lines from the topographic sheets of 1:50 000 scale. The DEM and the digital channel network were used to identify the flow direction, and to define the catchment and sub-basins. Further, a digital soil and land cover map of the area were used as the input to delineate the different HRUs based on the hydrological properties (Reshmidevi *et al.*, 2008). The digital soil map of the area, along with the soil characteristics information, viz. percentage of sand and clay content, hydrological group, bulk density, number of layers, depth of each layer, and available water content of each of the soil types, were obtained from the National Bureau of Soil Survey (NBSS). The land-use map of the area was generated from remote sensing satellite imagery obtained from the National Remote Sensing Agency, India. IRS LISS III imageries of January, March and November 2007 were used to extract the land-cover information, particularly the cropping pattern. In the present study, eight land cover classes were generated, as shown in Fig. 4. Areas under paddy (water-intensive crop) and sugarcane (perennial irrigated crop) were grouped into two separate classes. All other irrigated crops (wheat, maize, etc.) were considered in a single class called "other irrigated crops". Less water-intensive, rainfed crops like ragi, bajra and jowar were classified into "unirrigated crops".

In this study, because the rainfall shows very large variations over the catchment, rainfall data from the 18 stations were collected from the Directorate of Economics and Statistics (DES), assuring proper accounting for the spatial variation in rainfall. Observed meteorological data from the observatory located near the Malaprabha Dam were collected from various sources including the Indian Meteorological Department (IMD) and used to customize the climate station. AVSWAT simulates the runoff at different layers and produces results at three different levels: HRU, sub-basin and reach. In the present study, the period June 2001–May 2004 was selected as the study period. Though the rainfall and streamflow data are available for a much longer period,

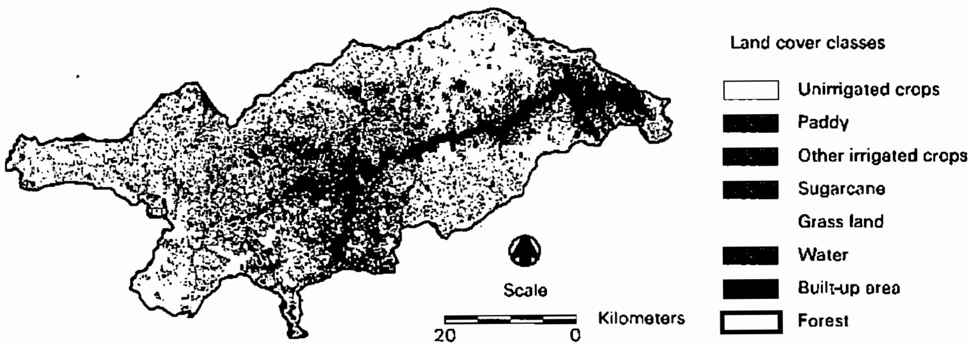

Fig. 4 Land-use and land-cover map of Malaprabha catchment (Reshmidevi *et al.*, 2008).

due to the lack of information about the earlier land-use pattern in the area, the study could not be extended pre-2000.

LAND-USE SCENARIO ANALYSIS

In order to study the impact of land-use changes on the streamflow in the catchment, four sets of land-use scenarios were built in AVSWAT.

Current scenario In the current scenario, the existing land-use/land-cover and irrigation scenario were assumed. The simulated streamflow was compared with the observed streamflow at Khanapur as well as with the reservoir inflow, and the model parameters were calibrated.

Base scenario The calibrated model parameters were used to develop further scenarios. The base scenario was developed by using the current land-use/land-cover map, but assuming rainfed cultivation in all the agricultural areas. With no water allocation for irrigation, this shows the maximum water available in the catchment.

Current trend The current trend of irrigation intensification was simulated by assuming a gradual variation from rainfed into irrigated sugarcane cultivation. A gradual variation from rainfed to sugarcane starting from 11% and increasing to 56% was assumed; these were termed as RS11, RS17, RS31, RS39, RS49, and RS56 respectively, where RS indicates the change from rainfed to sugarcane and the number followed indicates the percentage area converted.

Betterment scenario In order to understand how the water availability can be improved by controlling unsustainable agricultural practices, a set of scenarios were built by changing the sugarcane areas to rainfed crops. A change from 10% to 47% was assumed and these scenarios were termed SR10, SR20, SR30 and SR47, respectively where SR indicates the change from sugarcane to rainfed crops followed by the percentage area.

RESULTS AND DISCUSSIONS

Streamflow simulation from the current scenario for the study period was compared with the reservoir inflow. The simulated streamflow was found to be in good agreement with the observed streamflow giving a correlation coefficient of 0.95 and a Nash-Sutcliffe efficiency index of 0.71. The current streamflow after irrigation extraction was compared with the drinking water supply requirements for the Bailhongal Municipality (which is the largest rural settlement in the catchment area). The electricity bill at the Bailhongal water supply intake point was collected to estimate the normal pumping rate, which was estimated as 494 LPCD (L per capita per day; assuming a pump efficiency of 50%) for a population of 48 000. This translates to 0.275 m^3/s and was assumed as the minimum flow requirement at the intake point in the river. The current scenario was found to be causing water scarcity in the stream, with the streamflow during the peak summer of the dry years less than the production requirement at the Bailhongal intake point.

In order to assess the impacts of irrigation extraction on the streamflow, the base scenario was built into the model. From the analysis, 98% of the total streamflow in the year 2002–2003 (normal rainfall year) was observed during the monsoon period (June–November). Dry season flow was found to be only 2% of the total annual streamflow. Water yield from the sub-basins was studied separately for each zone and it was found that 82.5% of the water reaching the stream during the monsoon period is from zone 1, whereas the contributions from zones 2 and 3 are 11.8% and 4.7%, respectively. However, much of the dry season flow was found to be from zone 2. Water yields from zones 1, 2 and 3 during the dry season were found to be 7.2%, 62% and 30.8%, respectively, in 2002–2003. Assuming zero irrigation extraction from the stream, the resulting streamflow at the Bailhongal water supply intake point is shown in Fig. 5.

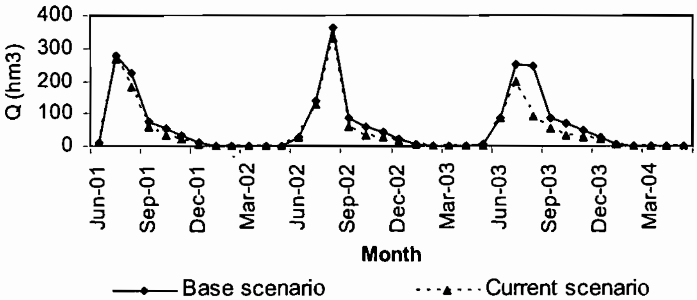

Fig. 5 Comparison between streamflow for base and current scenarios at the Bailhongal intake point.

From the comparison between the streamflow from the current and base scenario, irrigation extraction was found to be reducing the streamflow significantly, thereby reducing the water available for drinking water supply. Irrigation extraction was found to be reducing the streamflow in the monsoon as well as the dry season. During the three years of time considered in this study, when irrigation water was extracted from the stream with the current land-cover scenario, Bailhongal water supply requirement was found to be difficult to meet during the summer of below-normal rainfall years. As observed in the field, on average 2–4 months during the dry years were found to be critical periods when the simulated streamflow was found to be less than the estimated requirement. However, for the base scenario, water availability was found to be meeting the demand in all the months considered in this study, except March–April of the dry years.

Land-use scenario analysis

The land-use scenario analysis is divided into two parts. In the first part the trend of a shift from rainfed to irrigated sugarcane cultivation was assumed. Streamflow variation at the Bailhongal intake point for the scarce months (March–May 2002, May 2003, March–April 2004) under the six scenarios (SR11, SR17, SR31, SR39, SR49 and SR56) are shown in Fig. 6. Reduction in the streamflow due to the effect of increased irrigation was observed from the result. For the current scenario, in May 2003 and 2004, simulated streamflow was found to be higher than the demand. With an increase in the extent of irrigation, more periods of water stress was observed, e.g. May 2003 and May 2004 (Fig. 6). With the intensification/extensification of irrigation, there is a risk of prolonged water scarcity in the catchment. In addition to this, an average 35% increase in the intensity of scarcity is observed when 56% of the rainfed areas are converted to sugarcane as

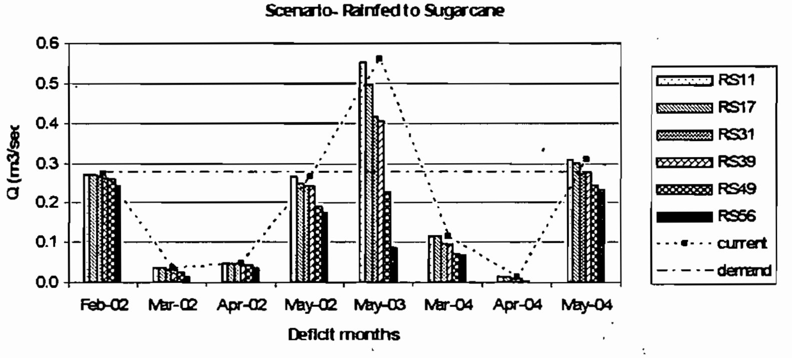

Fig. 6 Impact of irrigation intensification on streamflow.

shown in Fig. 6. Appropriate measures therefore need to be taken to prevent unsustainable change in the agricultural practices leading to water scarcity.

In the second part, improvement in the streamflow was studied by gradually changing the sugarcane areas back to rainfed cultivation. Streamflow at the Bailhongal intake point from the corresponding four scenarios (SR10, SR20, SR30 and SR47) for the six water-scarce months (February–May, 2002 and March–April, 2004) is shown in Fig. 7. For the below average rainfall years 2002 and 2004, significant improvement in the streamflow was observed when 10% of the existing sugarcane areas were converted into rainfed cultivation. With the SR10 scenario, water scarcity was found to be reduced from four months to two months (April–May) in 2002. In addition to this, an average 50% reduction in the intensity of scarcity was observed by replacing 47% of the sugarcane areas by rainfed crops.

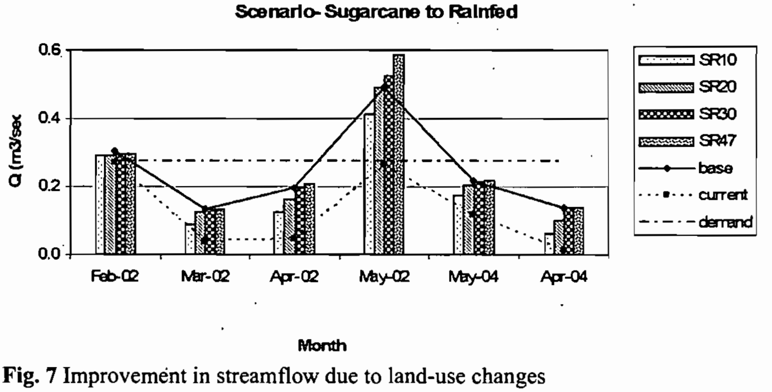

Fig. 7 Improvement in streamflow due to land-use changes

The results show the impact of land-use changes on the streamflow, particularly the dry season flow, in the Malaprabha catchment. With the shift from traditional rainfed cultivation to irrigated cultivation, the increased pumping from the stream to meet the irrigation requirement resulted in inequitable water distribution between different sectors. It was found that the current trend of unsustainable land-use change, in the form of the extensification of irrigation, if continued, will result in more severe water scarcity affecting the drinking water supply to the largest settlement in the catchment area. However, the scenario can be improved by reducing the irrigation extraction by adopting appropriate regulatory measures to control the unsustainable changes in the land-use and agricultural practices.

CONCLUSIONS

In this paper, hydrological analysis of the Malaprabha catchment using AVSWAT is presented from the perspective of identifying the water scarcity issues, causes and possible remedial measures. Long-term hydrometeorological data and streamflow observations were analysed to identify the water scarcity problem in the area in terms of scarcity of drinking water and reduced flow in the river, and the reasons for the scarcity. Giving priority to the drinking water supply for the Bailhongal municipality, over irrigation, the streamflow during the dry periods is compared with the demand at the intake point. In order to predict the impact of current land-use change on the future streamflow availability, scenarios were built by assuming gradual change from rainfed to irrigated-sugarcane cultivation. In this study attempts were also made to identify a possible remediation strategy in land use, wherein a gradual shift from irrigated sugarcane to the traditional rainfed cultivation and its impact on the streamflow was also studied.

Due to the variation in rainfall distribution, land-use conditions as well as the geological characteristics of the catchment, almost 80% of the streamflow and the groundwater recharge from the catchment are generated from zone 1. However, due to the post-monsoon rainfall, almost 60% of the post-monsoon runoff results from zone 2. Much of the streamflow generated from zones 1 and 2, as well as the groundwater extracted by zones 2 and 3, is used for irrigation. Major irrigation in the middle and lower catchment causes reduction in the streamflow. From the simulation results, the current land-use practice was found to result in 2–4 months of water scarcity at the Bailhongal intake point during dry years. More intense water scarcity and longer periods of scarcity can be expected with the current changing scenario. In addition, an average 50% increase in the intensity of water scarcity may result if 56% or more of the current rainfed areas were changed to sugarcane. Sustainable land-use practices can improve the situation. From the scenario analysis it has been found that a 10% change in the existing sugarcane area to rainfed crops could halve water scarcity at the Bailhongal intake point. Also, an average 35% reduction in the intensity of water scarcity could be achieved by converting 47% of the sugarcane areas to rainfed crops.

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Adaptive surface and groundwater management for vulnerable hydro-ecosystems in the northeastern region of India

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Abstract A multidisciplinary, long-term study showed that, on average, 16.2 to 141.1 tonnes of sediment yield per km^2 and more than 90% of rainwater was retained *in situ* in new land-use systems compared to 3621.3 tonnes of sediment and 64.4% rainwater retention in shifting cultivation. The mean values of base flow and surface flow were 104.6 mm and 31.0 mm in the new land-use systems as against 241.7 mm and 560.1 mm, respectively, in shifting cultivation. The groundwater recharge was maximum in livestock-based land use, followed closely by agriculture, horticulture and forestry land uses, the average being 34.7% of precipitation as against only 8.3% in shifting cultivation. Due to the reduction in runoff flows to river channels, the incidence of floods has reduced. The government has simulated these model land-use systems in different parts of the region for location-specific results so that farmers could adopt these as per agro-climatic situation and requirements.

Key words adaptive management; surface water; groundwater; vulnerable hydro-ecosystem

INTRODUCTION

The northeastern region of India has rich water resources, but their misuse and mismanagement have rendered them in a fragile state. Most of the surface water in the region is confined to river systems which are highly dynamic due to the high gradients and thus retain a very small quantity of freshwater for human use. Studies have shown significant effects of land-use change on hydrological fluxes. Surface water plays an important role in groundwater management, and knowledge of spatial and temporal distribution of surface water resources is required when developing water management strategies. Historically, human settlements have been established in areas with local water resources in the form of exploitable rivers or groundwater aquifers. However, fast growth in population has led to over-exploitation of the resources. An accurate description of the plant ecology requires an understanding of the interplay between precipitation, infiltration, and evapotranspiration. Hydrologists have considerable interest in land-use change and its hydrological consequences, both from the perspective of field monitoring (Bosch & Hewlett, 1982) and from a modelling perspective (Niehoff, 2002). However, in the long term, land-use change will also have an effect on soil physical properties.

Shifting cultivation is the major practice of land use in the northeastern region of India. The practice was alright when available land was abundant and the population was smaller. With the increase in demographic pressure in the region, the practice has become detrimental to natural resources as the shifting cycle has reduced from about 25 years to about 5 years. The misuse and mismanagement of rainwater has rendered the surface and groundwater resources in a fragile state in an otherwise vulnerable regional ecosystem (Sharma & Sharma, 2004). An understanding of the mechanisms that control groundwater interactions with surface water is crucial, both for the effective management of water resources and for the conservation of its associated ecosystem. Developing a range of appropriate adaptation options under location-specific situations will be of vital importance for integrated water resources management in a vulnerable hydro-ecosystem; otherwise, increased vulnerabilities to climate hazards will compound current water-related problems in the region. The extremely vulnerable environment and the risk of extreme events is due to the fast increase in population, intensification of agriculture and urbanization. Given the backdrop of limited development opportunities, there is a need to explore new concepts in water management with a more participatory and multi-stakeholder approach in the northeastern region. A multidisciplinary long-term study on various land-use options was, therefore, undertaken

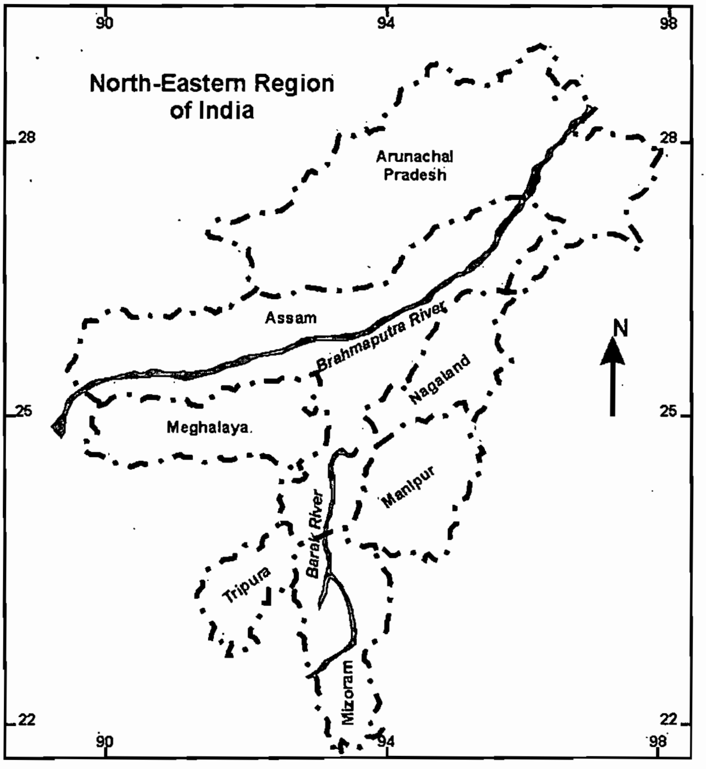

Fig. 1 Northeastern region of India.

to evaluate their performance with regard to runoff, soil and nutrient loss, *in situ* rainwater infiltration and its impact on the ecology and socio-economic conditions of the people of the region, so that integrated water resources management could be undertaken with adaptive management and active participation of the land users.

STUDY AREA AND METHODOLOGY

The site of study is the northeastern region of India (Fig. 1), with an area of 255 090 km^2, and comprising seven states: Arunachal Pradesh, Assam, Manipur, Meghalaya, Mizoram, Nagaland and Tripura. The experiment was conducted on hillslope micro-watersheds having slopes of 32.0% to 53.0%. The land-use systems are livestock based, forestry, agro-forestry, agriculture, agri-horti-silvi-pastoral (trees and pastures on top, horticulture in the middle and agriculture on the lower slope of the hill), horticulture and shifting cultivation (Table 1). The monitoring gauges were installed at the exit point of each watershed and suitable soil and water conservation practices, viz. bench terracing, trenching, half-moon terraces and grassed waterways, were followed except in forestry and shifting cultivation land uses. The runoff samples were collected and analysed for soil and nutrient concentration as per procedures outlined by Jackson (1973).

RESULTS AND DISCUSSION

Present scenario

Two major rivers in the region are the Brahmaputra and the Barak, which drain 194 400 and 78 100 km^2, respectively. The average annual flows are 537.2 km^3 from the Brahmaputra and 59.8 km^3 from the Barak River (Table 2). Shifting cultivation is prevalent, affecting 3869 km^2 annually;

Table 1 Vegetation cover and, soil and water conservation measures in different land uses.

Land use	Slope (%)	Crops	Soil and water conservation measures
Grasses and fodders	32.0	Maize, rice-bean, oats, peas, guinea grass, tapioca, broom-grass	Contour bunds, trenches, grassed waterways
Forestry	38.0	*Alder nepalensis, Albziia lebbeck, Acacia auriculiformis*	None
Agro-forestry	32.2	*Ficus hookerii,* Eucalyptus, pine apple, guava, beans, pulse crops	Contour bunds
Agriculture	32.4	Beans, radish, maize, paddy, ginger, turmeric, upland rice, oats, groundnut, grasses on risers	Bench terraces, contour Bunds, grassed water-ways
Agri-horti-silvi-pastoral	41.8	Ginger, *Alder nepalensis, Ficus hookeri, grasses,* guava, citrus, lemon, agricultural crops	Contour bunds, bench terraces, half-moon terraces, grassed waterways
Horticulture	53.2	Pear, peach, guava, citrus, lemon, vegetable crops	As above
Shifting cultivation	45.0	Mixture of crops	None

Table 2 Water availability in major rivers of the northeastern region.

River basin	Drainage area (10^3 km^2)	Average annual runoff (km^3)	Average runoff per km^2 (m^3)
Brahmaputra	194.4	537.2	276 300
Barak	78.1	59.8	765 600

however, the total affected area is 14 660 km^2. The fast increase in population in the northeastern region and great reduction in length of the shifting cycle have made the practice an uneconomical as well as resource depleting land-use system, causing soil erosion and resource degradation in the hills and silting of river beds and floods in the plains. The major problem is mismanagement of rainwater. The annual flood-affected area is 3760 km^2, whereas the area prone to floods is 35 840 km^2 or 14.05% of the geographical area. About 95.6% of the annual flood-affected area and 87.7% of the flood-prone area lie in Assam state. The region receives approximately 510 km^3 of rainwater at an average annual rate of 2450 mm. The runoff water goes untapped from the denuded hillslopes instead of infiltrating into the soil to recharge aquifers. About 8.86 million ha (34.7% of the area) of land has degraded due to mismanagement of water resources, with irreversible damage at some places. The region has 40% of the country's renewable freshwater. The surface water and groundwater resources of the region are 1487.4 km^3 and 31.22 km^3, respectively. The ultimate irrigation potential is 1485 and 1755 thousand ha from major and minor irrigation projects, respectively. About 601.2 million tonnes of soil is displaced though runoff, the quantities being 88.3, 90.7 and 422.2 million tonnes from shifting cultivation, other cultivated land and non-cultivated areas, respectively (Table 3). The runoff carries a nutrient load of 686.0, 100.2, 511.0 and 137.4 thousand tonnes of nitrogen, phosphorus, potassium and micronutrients, respectively (Sharma & Prasad, 1995). Some constituents, e.g. nitrate, are subject to chemical and biological processes in both groundwater and surface water passing through the region. In agricultural production, nutrients lost to water bodies do not contribute to the yield, but represent an economic loss. Integrated water resources management and adaptive management may be hindered by the complex hydro-political situation, characterized by natural water scarcity at some places and sometimes during the year, sharing of water, conflicting demands and intensive development. The uncertainty may also be related to climate change, which may aggravate the problem.

Social aspects of water resources management

For successful adaptive management of surface and groundwater, it is necessary to understand the socio-economic and socio-cultural scenario in the proposed area. The northeastern region of India is inhabited by several tribes. Shifting cultivation is not only a set of agricultural practices but

Table 3 Annual soil and nutrient loss from different states of northeastern region through runoff.

State	Soil loss (million tonnes): Shifting cultivation	Other cultivated areas	Non-cultivated areas	Total	Nutrients (10^3 tonnes): N	P	K	Micro-nutrients
Arunachal Pradesh	14.5	2.9	160.7	178.1	217	36.6	153	47.4
Assam	12.3	67.8	98.3	178.4	201	34.4	155	48.3
Manipur	20.4	2.8	40.8	64.0	76	7.4	63	10.0
Meghalaya	14.2	3.8	39.7	57.7	62	7.0	48	9.8
Mizoram	13.0	2.2	39.4	53.6	60	6.9	40	9.2
Nagaland	8.0	4.8	28.9	41.7	44	5.2	34	8.0
Tripura	5.9	6.4	15.4	27.7	26	2.7	18	4.7
Total	88.3	90.7	422.2	601.2	686	100.2	511	137.4

combines the whole nexus of people's belief, attitude, self-image and tribal identity. It is a transition between nomadic hunting and sedentary agriculture. The pattern, intensity and amount of annual precipitation being unique in the region, proper management of rain water can be helpful in managing water resources to a large extent. In the past, when land was abundant and population sparse, the rotational cycle of shifting cultivation used to be 25 to 30 years and the land had enough time for rejuvenation or revival of the vegetation cover. The soil fertility was maintained by *in situ* burning of forest vegetation and the production was enough to feed the limited population. However, with the increase in population, the rotational cycle has reduced to 2–10 years and the land does not get enough time for rejuvenation. Soil fertility is declining as there is limited material to burn and add to the soil. The annual area under shifting cultivation in the region is 3869 km^2, whereas the total area affected is 14 660 km^2 (Anon., 2000). The land tenure system in the region is unique. The land belongs to either: (i) the village chief, (ii) the community, or (iii) individuals. In the first two categories, the farmers have usufructuary rights over the land and so they have least interest in its development and as such do not make judicious use of available water resources. The prevalence of free range grazing by animals during the winter season (December–February), discourages cultivators from growing winter crops and integrated water resources management. Fast urbanization and change in life styles has also affected water resources management. The runoff water goes untapped from the denuded hill slopes instead of infiltrating in the soil to recharge aquifers (Sharma, 2003). Given this socio-economic scenario, a sound land-use policy is necessary for integrated management of water resources. More than 80% of the population in the region is engaged in agriculture and allied activities. Replacement of existing faulty land-use systems, which require deforestation to maintain soil health, is necessary for the adaptive management of surface and groundwater resources.

Effects of land use on the water yield and groundwater recharge

Potential groundwater recharge is estimated using the water balance approach, taking into account the crop water requirement, soil type and evapotranspiration from the bare soil. The precipitation reaches the groundwater and depending on groundwater intensity and the state of the soil, some rainfall runs away as runoff and some infiltrates into the soil zone (Hulme *et al.*, 2001). In the present study, the mean values of baseflow and surface flow were 104.6 mm and 31.0 mm in the new land-use systems, relative to 241.7 mm and 560.1 mm in shifting cultivation (Fig. 2(a)–(c)). *In situ* rainwater retention was maximum (99.4%) in livestock-based land-use systems, followed closely by agriculture (99.1%), agri-horti-silvi-pastoral (96.9%) and agro-forestry (90.4%), but only 64.4% in shifting cultivation (Fig. 2(d)). More *in situ* retention of rainwater helped the availability of soil moisture for the succeeding crops when the rainy season receded. It was reported (Anon., 1990) that more than 95% of rainwater can be retained *in situ* by following these land-use systems. The low rainwater retention in the shifting cultivation was due to the poor vegetation cover and greater slope, resulting in greater runoff mainly as surface and base flows. The results

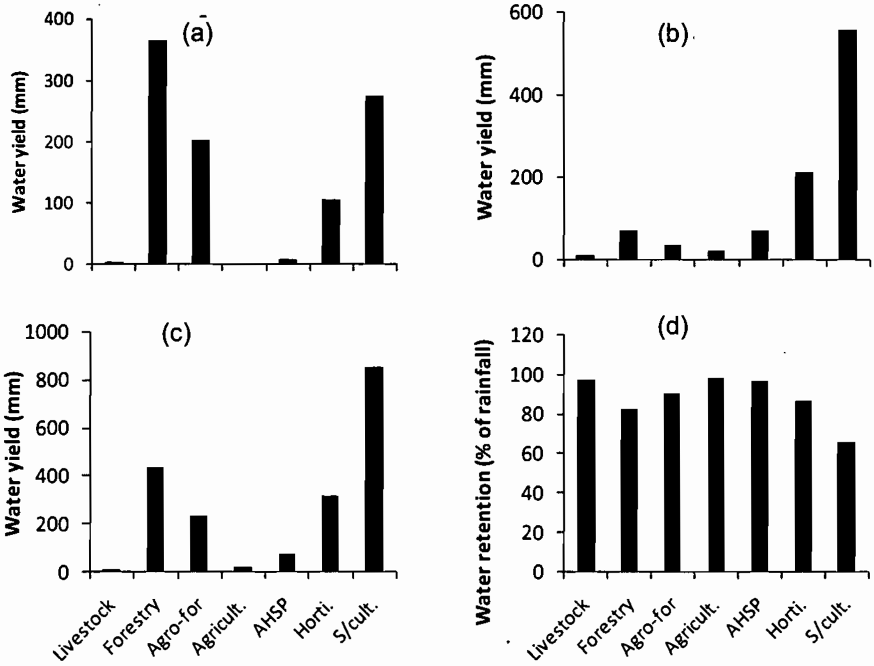

Fig. 2 Effect of land use on the water yield: (a) groundwater, (b) surface water, (c) total water yield and (d) and *in situ* water retention.

Table 4 Effect of land use, rainfall and their interactions on groundwater recharge (mm).

Land use	Rainfall (mm)						Mean
	2195	2705	2770	2599	2288	1992	
Livestock based	738	1212	1294	1101	835	555	956 (39.0)
Forestry	426	729	746	663	477	338	563 (22.9)
Agro-forestry	560	954	984	870	633	459	742 (30.2)
Agriculture	731	1219	1289	1102	831	570	957 (39.0)
Agri-horti-silvi-pastoral	679	1134	1198	1021	769	526	888 (36.2)
Horticulture	516	897	914	815	590	420	692 (28.2)
Shifting cultivation	152	260	274	231	168	125	202 (8.2)
Mean	543 (22.1)	915 (37.3)	957 (39.0)	829 (33.8)	615 (25.1)	426 (17.4)	

Figures in parenthesis are groundwater recharge as % of rainfall. The rainfall data are for 6 years.

obtained indicated that suitable soil conservation measures need to be followed on hillslopes to reduce the soil loss due to runoff, maximize *in situ* retention of rainwater and increase groundwater recharge. Direct groundwater recharge by rainfall is affected by the type, amount and duration of precipitation, initial soil moisture content, soil infiltration characteristics, topography and vegetation cover (Hussein, 2001). The average groundwater recharge was 39.0% of rainfall in livestock based and agriculture land-use systems, followed by agri-horti-silvi-pastoral, agroforestry, horticulture, forestry and shifting cultivation. In shifting cultivation, the groundwater recharge was only 8.2% of rainfall. The groundwater recharge increased with increase in rainfall (Table 4). The sudden spurt in population growth has put tremendous pressure on land and water resources, while development has not kept pace to accommodate the increased growth rate. The pressure on the freshwater system continues to grow for domestic use, agriculture, industry, energy and disposal of effluents, not only due to population increase but also due to a change in life style of the people (Sharma, 2001). Since food productivity is highly dependent on spatial and temporal changes in water availability, the future needs for water have to be met from water resources similar to those existing at present.

Impact of rainfall and land use on soil loss

The results of the study showed that, on average, 16.2 to 141.1 tonnes of soil loss per km^2 was recorded from new land-use systems (Fig. 3), as against 3621.3 tonnes from shifting cultivation (Fig. 3). The average sediment yield was only 0.44%, 2.68%, 1.47%, 0.31%, 0.73% and 2.27% of that of shifting cultivation in livestock based, forestry, agro-forestry, agriculture, agri-horti-silvi-pastoral and horticulture land-use systems, respectively. The amount of rainfall received had a significant impact on the sediment yield in runoff. Highest average sediment yield in new land-use systems was 141.1 t km^{-2} when the annual rainfall was 2770 mm and minimum 3.1 t km^{-2} when the annual rainfall was 1992 mm, cf. 4499.7 t km^{-2} and 2669.4 t km^{-2}, respectively, in shifting cultivation (Fig. 3). On average, the sediment yield varied from 405.4 to 704.3 t km^{-2} when the rainfall variation was from 1992 to 2770 mm, annually. In the new land-use systems, forestry recorded the maximum sediment yield under various rainfall regimes. This may be attributed to the fact that the land coverage with vegetation was much less in forestry than other land uses due to the tree planting distances. The soil loss was very low in the newly tried land-use systems due to reduced runoff because of the good vegetation cover and water and soil conservation measures undertaken. These land-use systems could be adopted to replace shifting cultivation in the region, depending on topography, slope and nearness to the market. The adoption of the new land-use systems will help in the adaptive management of surface water and groundwater with the active involvement of the stakeholders. The livestock-based land-use system can be adopted where there is a demand for milk or a nearby market for animal produce. The agri-horti-silvi-pastoral land-use system also suits the specific agro-ecosystem of the region well.

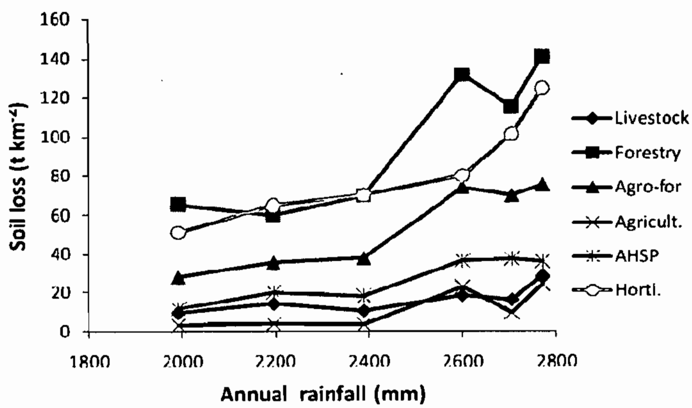

Fig. 3. Effect of land use and rainfall on soil loss through runoff.

CONCLUSIONS

Indiscriminate use of rainwater, the prevalence of shifting cultivation, and rapid and unplanned urbanization are a real threat to the management of surface water and groundwater resources in the northeastern region of India. The increase in runoff means a loss in potential groundwater recharge. Adequate plans need to be developed to accommodate the needs of the growing population in the region without jeopardizing the natural balance and the ability of the ecosystem to cope with the requirements. Surface water plays an important role in groundwater management and knowledge of spatial and temporal distribution of surface water resources is required when developing water management strategies. With new sustainable and eco-friendly land-use systems in place, the farmers have the option to change from shifting cultivation and opt for settled cultivation. Maximum rainwater could be retained *in situ* and the soil can retain sufficient moisture for growing winter crops. This would also help to reduce runoff and soil loss and to improve the environment. These land-use systems have been simulated at different locations, with a watershed approach, for demonstration and adoption. A keen interest has been shown by the farmers. The level of adoption, with modifications in cropping pattern according to the soil, climate and physiography; is encouraging, though no systematic survey has been done to ascertain the percentage of adopting families.

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Groundwater externalities of large, surface irrigation transfers: insights from the Godavari–Krishna river link, India

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Abstract The Krishna basin in South India is a fast closing basin. Consumptive water use, mainly for agriculture, accounted for 90.5% of the basin net inflow. The adjoining Godavari basin is comparatively water rich and it is planned to transfer 5325 million m^3 of Godavari waters to Krishna through P-V Link. Such a transfer should make the groundwater-irrigated area more sustainable, a part of the remaining rainfed area will come under irrigation and current benefits shall increase by 65%. MODFLOW results showed groundwater recharge increased by 28% due to supplement irrigation return flow. Annual estimated seepage from the main system was estimated at 183 million m^3 per year leading to an average water table rise of 1.83 m. This is also likely to affect 16% of the command with water logging. Integrated planning of surface and groundwater resources and a scientific demand management through optimization of cropping systems have the potential to effectively harness the benefits of the river linking initiative.

Key words inter-linking of river basins; groundwater modelling; groundwater externalities; water-logging; Krishna-Godavari basin, South India

INTRODUCTION

The economic growth of a country is critically linked to its water security (Grey & Sadoff, 2005). Coping with annual floods and droughts, both occurring at the same time in different regions, has been a major concern for India over the millennia. Designed to address these concerns, the National River Linking Project (NRLP) of India envisages transferring water from the potentially water-surplus to the water-scarce river basins and will build 30 river links and approximately 3000 storages to connect 37 Himalayan and peninsular rivers to form a gigantic South Asian water grid (Amarasinghe & Sharma, 2008). The Godavari and the Krishna are two large rivers in the peninsular region of India, with quite variable hydrological conditions and their delta regions lying in the same state of Andhra Pradesh.

The Godavari River is the second largest river in India with a catchment area of 312 812 km^2 and a long-term average annual surface flow of 110 km^3, of which 76 km^3 (69.1%) is estimated as non-utilizable (natural annual flow is exceeded 75% of time; NWDA, 1999). The adjoining Krishna River basin is the fourth largest in India with a total catchment area of 258 948 km^2 and a long-term average annual surface flow of 78 km^3, of which 58 km^3 (74.4%) is considered to be utilizable (Amarasinghe *et al.*, 2005). The cultivable area in the basin is about 20.3×10^6 ha and groundwater is the predominant source of irrigation. Due to massive surface and groundwater irrigation development, the annual river flow at the Krishna outlet has decreased to approximately 36% of its pre-development level, and certain studies have reported on the "closure" of the basin (Biggs *et al.*, 2007; Venot *et al.*, 2008). The Godavari (at Polavaram) to Krishna (at Vijayawada) Link (P-V link) is the most downstream link in both the Godavari and Krishna basins, and the one which is scheduled for construction in the near future (partly under construction). The Indira Sagar Right Main Canal (ISMRC) of this link envisages providing surface irrigation to a command area of 140 000 ha. Besides providing surface irrigation to the intended command, the large conveyance and distribution network and the actual surface application of the imported water will have significant impacts for the existing groundwater regime and use patterns.

This study examines the groundwater externalities due to introduction of surface irrigation in the P-V Link command through review of available material on the area and the project, primary

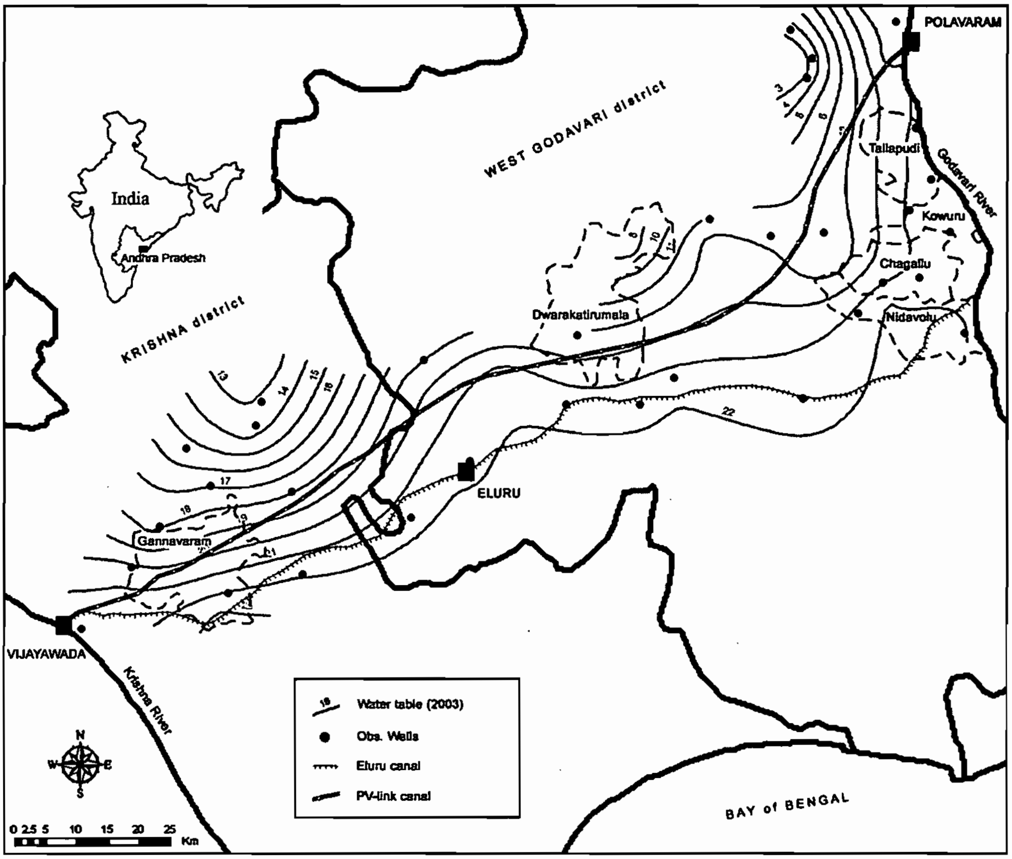

Fig. 1 Location map of the Polavaram (Godavari) to Vijayawada (Krishna) link.

surveys in the proposed command and groundwater simulation model studies. The paper provides a prognosis of the changes in groundwater regime and the accompanied production benefits from the proposed link.

STUDY AREA

The Indira Sagar Right Main Canal (ISMRC) command area stretches over the Krishna and West Godavari districts (Fig. 1). The area has a hot and semi-arid to sub-humid tropical climate. The average annual total rainfall is 900–1100 mm. About 70% of the precipitation occurs during the southwest monsoon (June–September) and 20% occurs during the northeast monsoon (October–November). The general topography of the area is mostly plain with a few local high mounds and occasional hills. Altitude decreases slightly from north to south before reaching sea level. The top soils within the area are generally red earths, black cotton soils and river alluvium. Up to 200 m of sedimentary deposits of Mesozoic to Quaternary age overlie the Archean Khondalite shield. The geology of the study area is characterised by high spatial variability with alternation of clayey or sandy layers (alluvium), sandstones or fractured rocks, but all the aquifers are more or less connected at the large scale. A major unconfined aquifer comprised of the Mesozoic sandstones and weathered/fractured Khondalite, flows from northeast to southeast with very low gradients, following the topography. The mean depth of the water table was estimated at about 17.43 m below ground level in May 2003 but can vary considerably (Sharma *et al.*, 2008).

METHODOLOGY

To assess the impact on groundwater of the surface water transfer through canal linking, the following strategy has been used: two steady-state groundwater models were implemented, respectively without and with the Indira Sagar Right Main Canal for the dry season (so-called MR1 and MR2) and then for the monsoon season (MK1 and MK2). The MR1 and MK1 models simulate the actual groundwater state of the area, whereas the MR2 and MK2 models simulate the future state with the ISMR canal. The influence of the canal on groundwater is obtained by comparing the groundwater balance and piezometric levels of the two states.

Water level data

Time series of pre- and post-monsoon groundwater levels were collected from the Andhra Pradesh Ground Water Development (APGWD) Department. For the study area, 59 observation points were available for West Godavari and Krishna districts from 1996 to 2005, which represent a total of around 1000 groundwater level measurements. Annual pre- and post-monsoon data (May and November) are sufficient for the general water table annual fluctuation (Marechal *et al.*, 2006). Long-term time series were used to select measurement points that are representative of the general groundwater level. Finally, 34 wells were retained, showing a slow long-term depletion trend estimated at about 0.05 m year^{-1} to 0.10 m year^{-1}.

Other data

Additional data were collected on land-use pattern, groundwater irrigation conditions, cropping patterns and crop yields, and the net returns. Farm and irrigation use surveys of 155 farms spread across the proposed command area of the P-V link were made through a well-designed questionnaire to understand the existing conditions and make an estimate of the expected production benefits/losses after the implementation of the link canal.

GROUNDWATER MODELLING

Model set-up

Due to data quality and scarcity, the physical modelling of the area has to be as simple as possible for providing significant results. The physically-based groundwater model MODFLOW (Harbaugh *et al.*, 2000) was chosen for its robustness, flexibility and simplicity. The most contrasted geological formations, in terms of hydrodynamic characteristics (permeability, K and specific yield, Sy), were represented in a four-layer model, in order to reproduce the geological complexity without introducing too many numerical instabilities. The first layer simulates the superficial formations (alluvium) with good porosity and medium permeability; the second layer simulates the Deccan Traps formation with good porosity and permeability; the third simulates the Rajamundry sandstones as highly porous and permeable; and the fourth simulates the fractured Khondalites with very low porosity but high permeability. Spatial segmentation was defined to be fine enough for transferring flows from the canal through adjoining cells. In accordance with low hydraulic gradients (<2‰), a grid spacing of 1 km was used. The grid was rotated to the North 39° direction, keeping cell boundaries perpendicular to the general flow lines in order to facilitate the numerical model convergence. The ground surface was described based on the 90-m resolution SRTM DEM degraded at 1-km^2 resolution to fit with the model resolution.

The flow domain was limited by the Godavari River to the east, as a river condition in MODFLOW, with a conductance coefficient based on the river's mean width, bottom and water level elevation (Harbaugh *et al.*, 2000). The western limit of the model followed the piezometric ridge close to the Krishna River and a no-flow boundary was established. The canals were surrounded by two buffer zones in order to minimise boundary effects. Flow originated in the vertical plane along the 65 m water-table contour where lateral flux was specified and exited through the vertical plane along the 5 m water-table contour. The specified influx was based on the water table Darcy flow calculation. The MODFLOW River package was used to simulate the influence of the canals on the groundwater, depending on the hydraulic gradient between the surface water body and the groundwater system. The canal width is independent of the model grid resolution (Harbaugh *et al.*, 2000).

Hydrodynamic properties of the aquifers are scarce in the area and only a few values interpreted from short-time pumping tests have been collected. The reliability of these very local measurements is difficult to assess and spatial local variability is also very high, especially in the alluvium and fractured rocks. However, the geology is well documented so hydrodynamic parameters can be calibrated based on usual values for a given geological formation. A single homogeneous permeability value was assigned to each geological formation.

For the MR1 model, the groundwater recharge from rainfall is nil (dry season), recharge from canal seepage (Eluru canal) and return flow from irrigation (percolation of excess water) act as groundwater recharge. For the MR2 model, seepage from the P-V link canal and the return flow from the additional irrigation increase the potential groundwater recharge. Seepage through the canals is directly calculated by the MODFLOW River package. Return flow to the aquifer from irrigation is estimated considering the net area irrigated for each crop and their specific common return flow coefficient in the region (i.e. rice 51%, sugar cane 15%, vegetable 25%, flower 12%; Dewandel *et al.*, 2007). Groundwater withdrawals for irrigation supply are estimated based on crop water requirement, net area sown and the groundwater irrigation practices of the farmers (Massuel *et al.*, 2008). After the commissioning of ISRMC, it is assumed that the same cropping pattern is extended to the new command area and that the proportions of surface and groundwater irrigation remain the same.

RESULTS

Calibration, validation and sensitivity

Calibration was performed only on the MR1 model (*Rabi* (winter) season model), where direct groundwater recharge from rainfall does not occur and where the Indira Sagar canal is not simulated. The uncertainty of the fixed variables is therefore minimized and leads to an easier aquifer characteristic calibration; only the permeability parameter is tuned. The difference between the calculated and observed groundwater levels is minimized based on 14 piezometric control points selected for the quality of measurement and their representativeness.

A best fit is obtained with a RMS of 3.79 m (Fig. 2). This result is acceptable compared to the observed data range (normalized RMS = 5.8%). Furthermore, the absolute residual mean is 3.0 m which means that with an equivalent specific storage of 4%, the calibration error in water table height is 0.09 m. General flows of the water table are well simulated except for some specific local areas. Calibrated permeability for the alluvium is 5.5×10^{-4} m s^{-1}, 9.7×10^{-5} m s^{-1} for Rajahmundry sandstones, 1.1×10^{-6} m s^{-1} for Deccan Traps, 1.6×10^{-5} m s^{-1} for Tirupathy sandstones and 4.2×10^{-5} m s^{-1} for Khondalites. They are in the range of usual values (Domenico & Schwartz, 1997), except for the Deccan Traps which have a very low calibrated permeability, almost certainly due to the difficulty of representing the geometry of the basalts in the model.

Validation is performed on the MK1 model by applying the direct groundwater recharge from rainfall (Dewandel *et al.*, 2007), all other thing being equal to MR1. The MK1 simulation flow efficiency is calculated with reference to the piezometry of the *Kharif* season. The fit remains good, RMS is 4.20 m and absolute residual mean is 3.80 m. The MK1 model therefore confirms the reliability of the MR1 model calibration.

Sensitivity tests have been carried out for different variations of permeability values. A general increase of 30% leads to a general increase in the water table of 0.89 m and enhances the leakage from Eluru Canal of 12%. The specified flow decreases by 35%. A decrease of 30% leads to a general drop of the water table by 1.19 m and reduces the leakage from Eluru Canal, while the flow through artificial boundaries decreases to 31%. The model is therefore quite sensitive to the artificial boundary conditions even if the impact on the general piezometry is low.

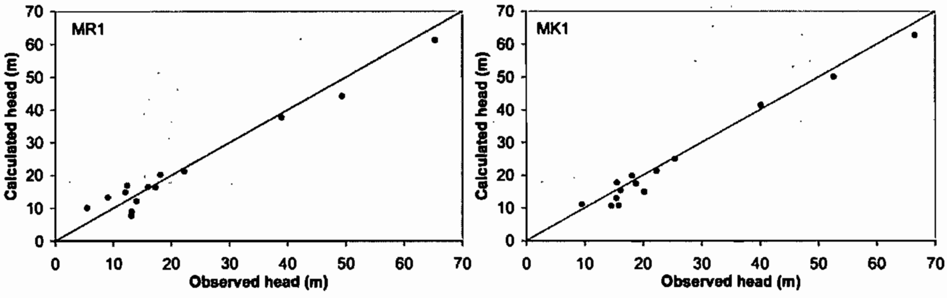

Fig. 2 Observed *versus* calculated groundwater heads for calibration (left) and validation (right).

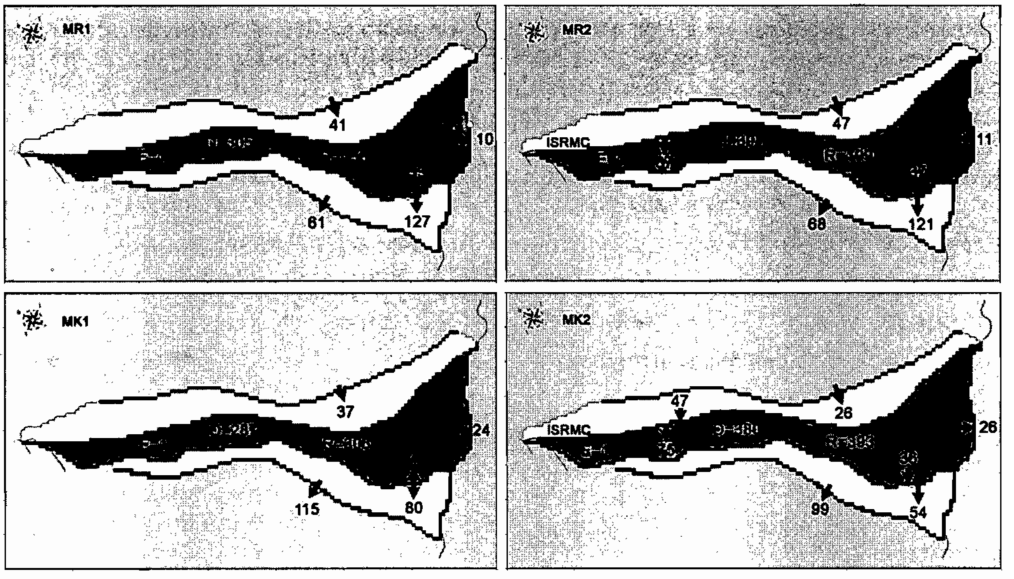

Fig. 3 Groundwater budget for the four simulated states in mm. E (direct groundwater evaporation), D (groundwater discharge), R (groundwater recharge) for the inter-canal zone (dark grey).

Groundwater impact assessment

The general groundwater budget simulated by the model is presented in Fig. 3. Before ISRMC, simulated groundwater discharge (D) dominates in the dry season with 305 mm while total recharge (R), which mainly comes from irrigation return flow, is only 156 mm. In the monsoon season calibrated R reaches 306 mm and D decreases to 287 mm. According to the model, outflow to the Godavari River (baseflow) is significant (10–24 mm). Eluru Canal seepage has more influence on the downstream than on the ISRMC future command area, especially in the monsoon season (40% of the leakage flows in the ISRMC command area in *Rabi* (winter) compared to 5% in the dry season). After ISRMC, groundwater flow through the natural and artificial boundaries (north and south) is not significantly affected since discharge increases in the same proportions as recharge (~28%). According to the model, due to water-table rise, the Eluru Canal would act as a drainage or water transfer system in the wet season.

Recharge (R) and discharge (D) in the model are defined as:

$$R = GW_{rf} + SW_{rf} + R_{rain} + R_{other} \tag{1}$$

where R is the calibrated total net groundwater recharge; GW_{rf} is the return flow to the aquifer from groundwater source excess irrigation; SW_{rf} is the return flow to the aquifer from surface water source excess irrigation; R_{rain} is the direct recharge from rainfall, and R_{other} are the other potential recharge sources.

$$D = P_{irri} + ET_{veg} + ET_{crop} + P_{other} \quad (2)$$

where D is the calculated discharge from the aquifer; P_{irri} is the groundwater draft for irrigation purpose; ET_{veg} is the natural vegetation draft; ET_{crop} is the cropped vegetation draft and P_{other} the withdrawals for other uses.

The parameter values have been assessed based on irrigation statistics data as explained in the methodology section (Table 1). Model recharge and discharge calculations appear to be relevant with statistical estimations for solving equations (1) and (2). According to the model, groundwater abstraction for irrigation supply accounted for 90% of the total groundwater discharge. Total return flow to the aquifer from excess irrigation ($GW_{rf} + SW_{rf}$) is not dependent on the proportion of groundwater irrigated area, which gives strong reliability to the recharge calculations.

According to the groundwater model in steady state, the P-V Link canal has a significant influence on the groundwater budget: (1) directly through the canal seepage, and (2) indirectly by excess irrigation seepage and additional groundwater abstraction due to extension of the command area. It is therefore important to recall that predicting the canal's impact on groundwater is tricky and has to be closely linked with the assumptions that led to the results. In the model, groundwater recharge rises by 28%, due to additional excess irrigation return flow in the new ISRMC command area. Annual estimated recharge from ISRMC seepage is 130 mm/year (183 × 10^6 m^3/year), which is consistent with the estimates of designed total transmission losses of 260 × 10^6 m^3/year. The annual balance between the situations before and after ISRMC shows a net increase in recharge of 73 mm (Table 2). Assuming an average aquifer equivalent specific yield of 4%, a water-table rise of 1.83 m can be anticipated from the increase in recharge.

The impact on groundwater condition of the increase in recharge was also evaluated. The stage of groundwater development (SGD) was calculated (draft/availability) after ISRMC and compared to the situation before ISRMC. Assuming addition of 73 mm on water availability, all other things being equal, it appears that the categories of five administrative units (*Mandals*) are changing: "Over-exploited" becomes "Critical" for two *Mandals* (Kovvuru and Nidadavolu), "Critical" becomes "Semi Critical" for one *Mandal* (Dwarakatirumala) and "Semi-critical" becomes "Safe" for two *Mandals* (Chagallu and Tallapudi). Further, when the system is balanced, the general piezometric level will stabilize on average at 2.20 m and 1.90 m above the actual mean monsoon and dry season levels, respectively. As a consequence, water-logging from high water table

Table 1 Annual groundwater balance before and after commissioning of ISRMC and warnings of basin closure emerge during dry periods.

Annual recharge, R (mm)		GW_{rf}	SW_{rf}	R_{rain}	R_{other}
Before ISRMC	462 (156+306)	178	193	90	1
After ISRMC	592 (199+393)	235	236	90	31
Annual discharge, D (mm)		P_{irri}	ET_{veg}	ET_{crop}	P_{other}
Before ISRMC	592 (305+287)	527	65		
After ISRMC	771 (391+380)	703	68		

Table 2 Computed water budget before and after ISRMC and net balance in the command area (excluding boundary fluxes).

Condition	Annual recharge, R (mm)	Annual discharge, D (mm)	Annual leakage from canals (mm)	Net recharge (mm)
Before ISRMC	462	–592	57	--
After ISRMC	592	–771	179	--
Balance	130	–179	122	73

conditions may potentially increase. According to the simulations, the potential area of water-logging (<2 m b.g.l.) could increase by 16% in the dry season and by 19% in the wet season and cover respectively 342 and 390 km^2 of the total 1582 km^2 of the inter-canal area. This negative externality occurs mainly in the vicinity of the P-V Link canal and particularly in Gannavaram *Mandal*.

Expected crop production benefits

The farmers survey results on the net benefits were used to estimate the current agricultural net returns and the future potential benefits. Annually, the net returns from groundwater dependent agriculture in the proposed command is about INR1.62 billion, which is threatened due to a diminishing resource and declining water levels. After commissioning of the link, the groundwater irrigated area will become more sustainable and the remaining rainfed area shall also come under irrigation. Overall, cropping intensity will increase from 124% to 150%. The projected estimates show that the total benefits will increase to INR 2.785 billion/year (67.9% increase, due to enhanced crop production from the greater irrigated area and increased cropping intensity (Table 3).

Table 3 Projected net returns in the P-V Link command.

Crop	NWDA assessed area under crops: Irrigated area (ha)	Unirrigated area (ha)	Total (ha)	Irrigated area: Yield (t/ha)	Net returns, INR/ha	Total net returns, INR×10^5	Unirrigated area: Yield t/ha	Net returns INR/ha	Total net returns INR×10^5	Current net returns, INR×10^5	Projected net returns after P-V Link, INR×10^5
Rice	60 672	713	61 385	5.0	12 158	7 377	1.69	7473	53	7 430	7 463
Black gram	6 150	1 465	7 615	0.6	9 995	615	0.4	7465	109	724	761
Maize	2 553	201	2 754	4.8	8 800	225	1.87	3154	6	231	242
Chillies/ cotton	2 972	25	2 997	1.0	11 145	331	0.5	5676	1	333	334
Ground nut	6 786	810	7 596	3.1	24 496	1 662	1.58	7299	59	1 721	1 861
Sub-total*	96 458	7 159	103 617		16 805	16 210		9247	662	16 872	17 094
Unirrigated area to be developed		35 953	35 953								5 931
Total expected net returns											23 025
Total expected net returns with cropping intensity of 150%											27 853

*including other minor crops

DISCUSSION AND CONCLUSIONS

Many assumptions have been made to simplify the model implementation but consistent results demonstrate the relevance of the different hypotheses. During the calibration, it was shown that the model layering has to be simplified to give more stability to the model. Hypotheses made on the recharge forcing from rainfall or surface irrigation return flow need to be confirmed by a surface water balance analysis (land use, cropping pattern, irrigation techniques, etc.). For higher accuracy of the groundwater balance, transient state modelling should be used in order to include time variability and specific storage constraints. Such a complex model would require coupling with a fully distributed surface model for net recharge forcing, by taking into account irrigation uses, the cropping calendar and irrigation canal water balance. Given the data availability, the output improvement should not be significant compared to the increase in complexity.

The groundwater modelling presented here can be considered as preliminary relevant results and indicate the influence of the P-V Link canal on the groundwater balance. For the inter-canal zone, according to the steady state simulations, the water table should rise by 2.20 m and 1.90 m on average for the *Rabi* and *Kharif* seasons, respectively. Potential areas of waterlogging could increase by 16% in the *Rabi* and by 19% in the *Kharif* season. The net groundwater recharge should be enhanced by 73 mm (103 × 10^6 m^3).

Introduction of surface irrigation systems in the sub-basin is likely to improve the groundwater regime and yet may not be sufficient to sustainably meet the current and future water requirements. Conjunctive management of surface and groundwater resources and scientific demand management through optimization of cropping systems have the potential to effectively harness the benefits of this river-linking initiative.

About 16% of the proposed command is likely to witness water logging, especially in Gannavaram block, if appropriate remediation measures are not put in place, and this will be a serious negative externality of the Project. Integrated planning of surface and groundwater use in the affected and adjoining areas can handle this. The worst affected areas can be put under paddy–paddy crop rotation for higher economic benefits (Sharma *et al.*, 2008).

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Modelling water resources in the Sana'a basin, Yemen, using a WEAP model

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Abstract The Water Evaluation and Planning system (WEAP) provides database and scenario-driven forecasting, policy analysis, and resource management capabilities. A comprehensive set of adaptation strategies to address water scarcity has been investigated using WEAP, and the relative costs and benefits weighed using multi-criteria analysis (MCA) methods, with broad input solicited from stakeholders in Sana'a basin in Yemen. Results of the WEAP include information on: the groundwater storage in the aquifers, unmet water demand and demand site coverage. The results of this study will assist decision makers in the field of water resources planning and management in Yemen and similar countries in the region who seek to better manage their water resources and water forecasting for the future.

Key words resource planning; Water Evaluation and Planning System (WEAP); Sana'a basin, Yemen

INTRODUCTION

The Republic of Yemen (ROY) is located at the south-southeast part of the Arabian Peninsula, bordered on the north by Saudi Arabia, on the south by the Arab Sea and the Gulf of Aden, on the east by the Sultanate of Oman, and on the west by the Red Sea. Yemen is a semi-arid country covering approximately 536 000 km^2 and characterized by five major zones that span elevations from sea level to 3700 m above sea level: (a) hot and humid coastal plain; (b) temperate highlands; (c) high plateaus and the Hadhramout–Mahra uplands; (d) desert interior; and (e) islands. Yemen's landscape possesses no permanent river systems and has limited and irregular rainfall, ranging from 50 to 500 mm annually among the regions.

Rain falls primarily in spring and summer, and is determined by two main mechanisms: the Red Sea Convergence Zone and the Inter Tropical Convergence Zone. Temperature depends primarily on elevation and, in the coastal areas, is determined by distance from the sea. Mean annual temperatures range from less than 12°C in the Highlands (with occasional freezing) to 30°C in the coastal plains.

ASSESSING CURRENT VULNERABILITY

Hydrologically, the Sana'a basin can be divided into 22 sub-basins. There are six major aquifers in the Sana'a basin: Northwestern, Northeastern, Central, Eastern, Southwestern and Southern. The urban area surrounding Sana'a draws water from the Central Plains aquifer. Trends in water consumption have put enormous strain on the basin's limited groundwater resources. Indeed, Sana'a has experienced dramatic declines in its groundwater table in recent years, a fact that has been cited by all stakeholders as the region's most pressing development challenge. In 2004 alone, the groundwater table dropped by about 7 m in the eastern region and about 8 m in the western region of the basin. These declines come in addition to water table declines of previous years, all of which reflect increasing access to groundwater in recent years.

ASSESSING FUTURE VULNERABILITY USING WEAP

Several assumptions were integrated into the reference scenario of the Water Evaluation and Planning (WEAP) system. Population growth in Sana'a city was assumed to continue at high rates:

5% per year to the end of the planning period in 2026. Agriculture in sub-basins 9 and 16 (surrounding Sana'a city) was assumed to continue recent patterns of decline by about 3% per year, compared to 1% per year decline in sub-basins south of Sana'a. However, agriculture in the sub-basins north of Sana'a was assumed to increase by about 3% annually, consistent with recent patterns and government plans. In addition, the treatment capacity of the Sana'a waste water treatment plant (WWTP) was assumed to increase to 0.1×10^6 m^3/day in 2012, consistent with expansion plans.

MODEL RESULTS

Calibration

The WEAP simulates precipitation, runoff, groundwater infiltration, changes in soil moisture and evapotranspirative and irrigation demands of land cover in each of the catchment nodes. The model shows the good calibration between measured and calculated values (Fig. 1).

ADAPTATION STRATEGIES

When considered basin-wide, implementation of any individual strategy decreases depletion of total basin aquifer storage relative to the reference condition (Fig. 2). Similarly, as any given strategy reduces the groundwater depletion in the various aquifers, total unmet demand across the basin diminishes as well, being highest under reference conditions (Fig. 3). For the purposes of assessing the relative success of each strategy in terms of potential water savings for each aquifer, a metric can be defined as the water remaining in each of aquifers under a particular strategy condition at the time of that aquifer's full depletion under reference conditions (Table 1).

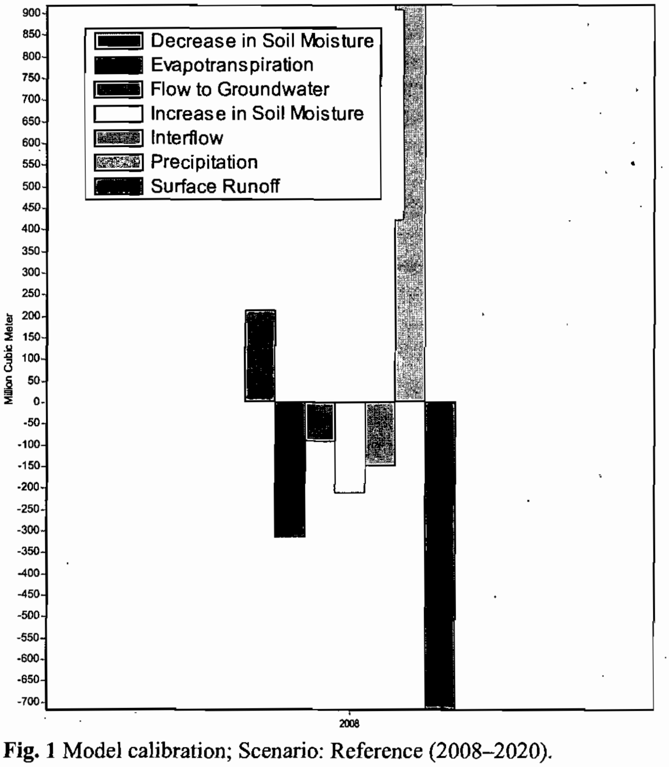

Fig. 1 Model calibration; Scenario: Reference (2008–2020).

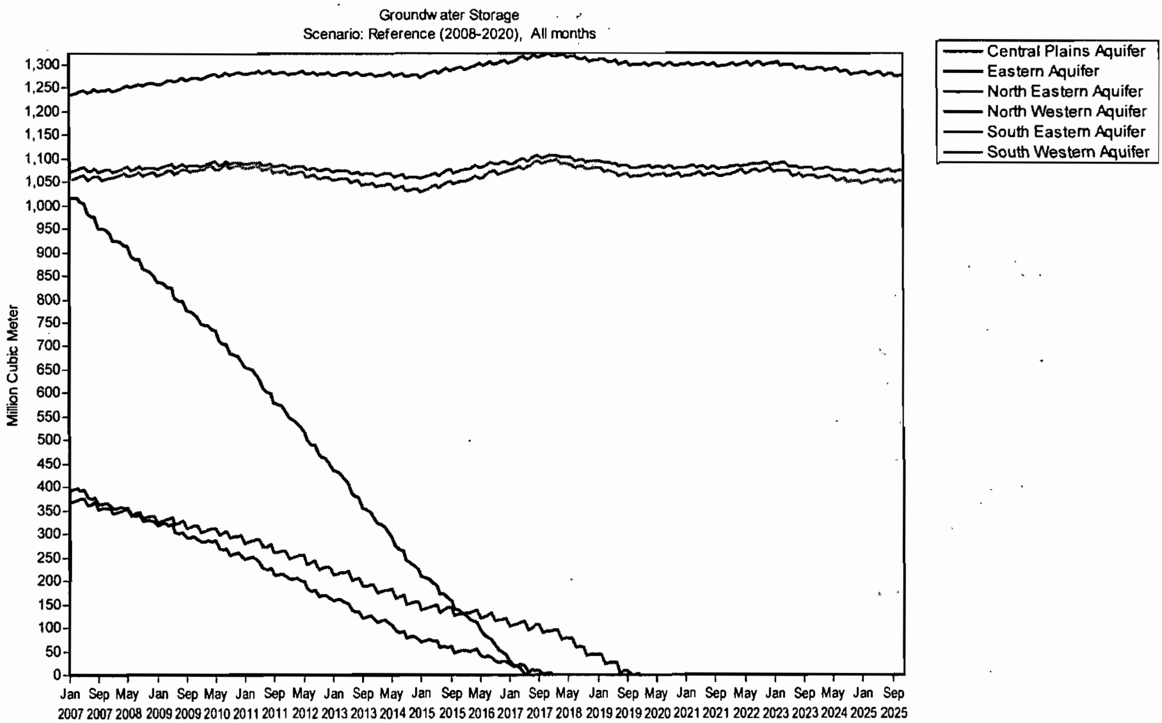

Fig. 2 Groundwater storage in the Central Plains aquifer under different strategy scenarios.

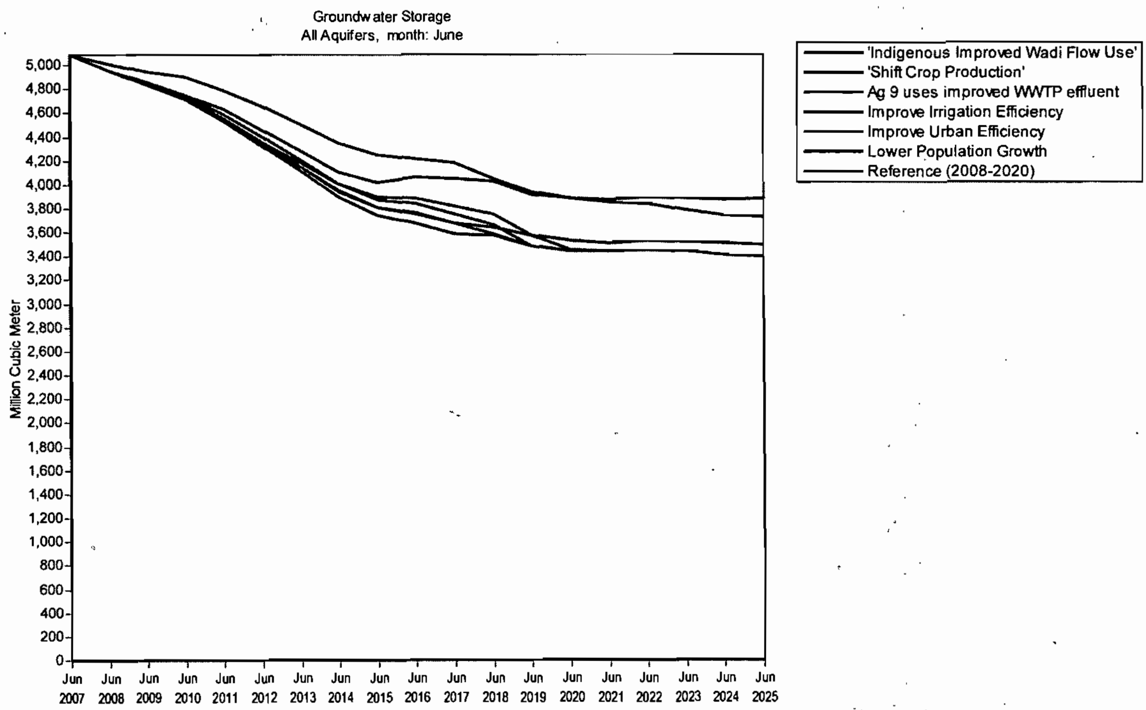

Fig. 3 Groundwater storage for all aquifers under different strategy scenarios.

Table 1 Groundwater savings (in 10^6 m^3) for strategies relative to reference condition at time of full aquifer depletion in reference condition.

Scenario	Aquifer: Central Plains	Eastern	Northeastern	Northwestern	Southeastern	Southwestern
Reference	0 June 2017[a]	0 October 2017[a]	Sustained	0 January 2020[a]	Sustained	Sustained
Improve wadi infiltration	54	62	Sustained	97	Sustained	Sustained
Shifting from Qat	2.5	12	Sustained	35	Sustained	Sustained
WWTP effluent	239	NA	Sustained	NA	Sustained	Sustained
Improve irrigation efficiency	136	144	Sustained	142	Sustained	Sustained
Lower population growth in Sana'a city	91	NA	Sustained	NA	Sustained	Sustained
Improve urban efficiency	173	NA	Sustained	NA	Sustained	Sustained

[a] Date of full aquifer depletion.

Those strategies that are implemented in all of the sub-basins (e.g. introducing irrigation efficiency or promoting improved indigenous wadi flow infiltration) rather than targeted in specific sub-basins (e.g. improving urban efficiency in sub-basin 16 and using WWTP effluent for irrigation in sub-basin 9) provide greater savings of total groundwater reserves (Fig. 3). Note that the irrigation efficiency and wadi infiltration scenarios preserve groundwater reserves through different routes, as the former decreases water demand compared to the reference (Fig. 4), while the latter scenario increases groundwater supply (Fig. 5).

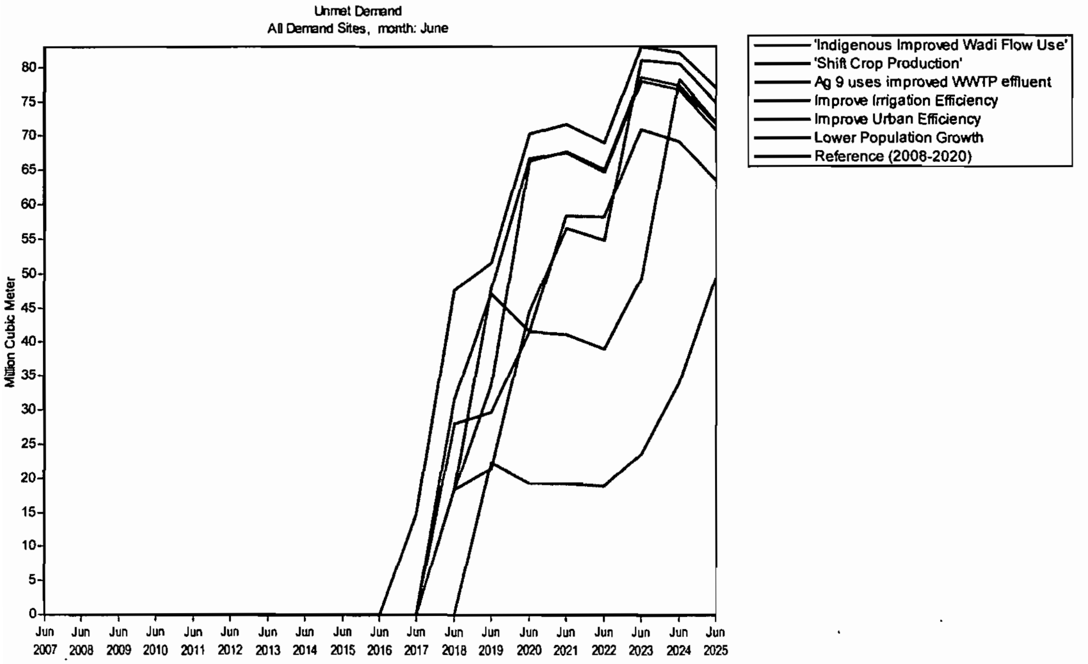

Fig. 4 Unmet demand for all demand sites under all scenario conditions.

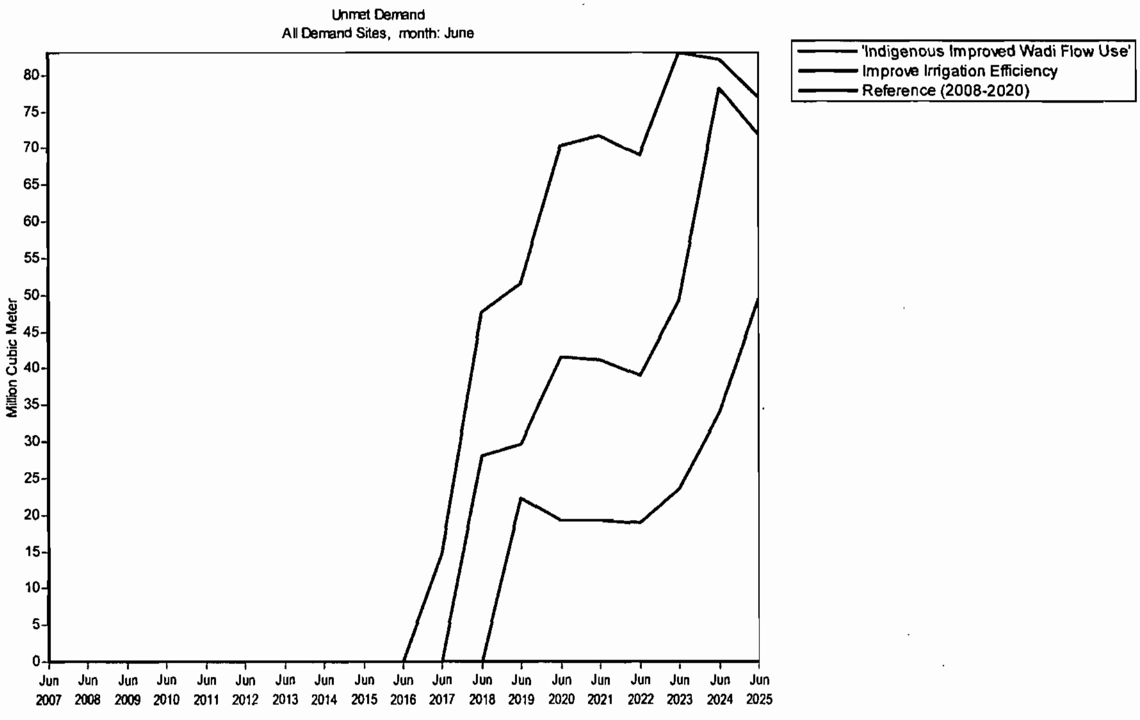

Fig. 5 Basin-wide irrigation water demand under the improved wadi flow and improved irrigation efficiency scenario conditions.

LESSONS LEARNED AND STRATEGIC RECOMMENDATIONS

The key lesson learned from the study is that Sana'a basin will continue to suffer from a pressing water crisis in the absence of strategies to stabilize water supply and demand patterns. Of particular note is the fact that climate variability and climate change are less influential than current and predicted patterns of agricultural and household water consumption. At the present time, annual withdrawals from groundwater resources exceed renewable resources by wide and unsustainable margins, and will likely continue into the future in the absence of a vigorous intervention policy. Indeed, the analysis in this study has revealed that a collapse of water supply systems will likely take place toward the end of the next decade in several important aquifers, suggesting that intervention is urgently needed. At the technological level, improved efficiencies through drip irrigation and improved water distribution systems will have demonstrable effects when combined with other supporting adaptation initiatives:

- The choice of adaptation strategy depends on the dictates of the particular case study region, including both physical and stakeholder inputs. While pilot adaptation measures were not implemented as part of the study, considerable effort was spent on scoping out the highest priority pilot scale adaptation for each area with help of local institutions.
- Stakeholders identified improvement of the traditional methods for wadi flow use as the highest priority initiative. The farming communities along this wadi are well aware of the need to harvest wadi storm flows. The area is littered with small structures in varying states of disrepair designed primarily to divert spate flow onto adjacent land to replenish soil moisture and groundwater. Approximately 23 check dams would be constructed on the main watercourse of Wadi Asser watershed to reduce runoff flow and to enhance groundwater recharge.

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Quantitative analysis of ecological protection objectives in the Yellow River basin

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Abstract The primary driving factor for ecosystem disturbances in the Yellow River basin is the degradation of ecosystem structure and function caused by water shortage. Ecosystem health is determined by a series of thresholds, and if these thresholds are exceeded the ecosystem can only be restored to a healthy state with the greatest difficulty. Due to spatial variability and zones of different ecological importance, the Yellow River basin has been delineated into 51 zones, of which 13 were identified as preferential zones. Vegetation coverage rate and population scale were chosen as indicators and analysed with models for optimal vegetation coverage rate and water resource carrying capacity, respectively. The ecological status of most zones meets the optimal protection objectives, except Ningxia and Inner Mongolia, the Henan region in the middle reach and the delta.

Key words scarce water resources; preferential protection zones; optimal vegetation coverage rate; optimal population scale; the Yellow River basin

INTRODUCTION

Ecological protection objectives are an important basis for directing water resource management in watersheds and for ensuring sustainable use of water resources. The protection objectives should meet the needs of ecosystem health in the watershed and of economic development. Nowadays, most ecological protection objectives are general qualitative descriptions, like halting deterioration trend of ecosystem and improving ecological status. Such objectives are difficult to use in practice, and do not directly enumerate indicators like forest coverage rate, area of desertification control, or wetland area. The determination of these objectives lacks scientific theory and a quantitative derivation process. As such, they are not conducive to real-world application.

The most severe ecological problem of the Yellow River basin is the loss of water and fertile land. The solution needed consists of planting vegetation with a reasonable coverage rate. Population levels are closely related to the degree of exploitation of water resources. This paper takes vegetation coverage rate and population level as indicators of ecological protection objectives and quantitatively analyses them to see which can be expected to be operable.

There has been much research about the optimal level of the vegetation coverage ratio. Most of this research was based on the requirements for soil and water conservation in small watersheds and neglected the need for economic development of land and water resources. As such, this previous research is not useful for application at regional scales. This paper will quantitatively analyse the optimal vegetation coverage rate (OVCR) in accordance with natural conditions and requirements for economic development.

When water resources restrict development, taking water resource as constraints is an effective method to determine a rational population level (Chen *et al.*, 2002). This paper will use a water resources carrying capacity model to calculate optimal population levels, which is the number of people that the available water resources can support under reasonable exploitation and utilization.

STUDY AREA DESCRIPTION

The Yellow River lies between 95°53′ and 119°05′E, and between 32°10′ and 41°50′N as shown in Fig. 1. The river originates on the Tibetan Plateau and ends in the Bohai Sea. It passes through

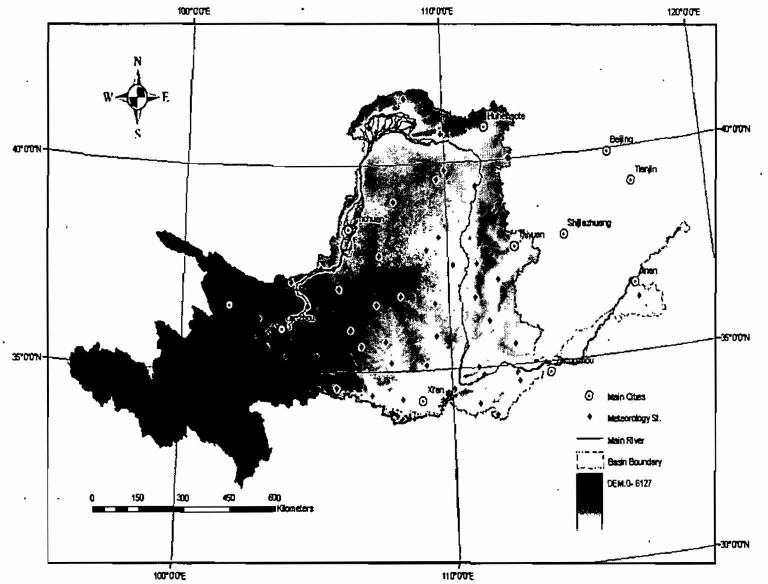

Fig. 1 Map of study area and location of meteorological stations.

nine provinces and autonomous regions, such as Qinghai, Sichuan, Gansu, Ningxia, Inner Mongolia, Shaanxi, Shanxi, Henan and Shandong, with an area of 753 000 km^2. The annual average temperature of the watershed is 8.6°C, annual average precipitation is 466 mm, with a decreasing trend from southeast to northwest, and the total number of sunshine hours varies from 1400 to 3300. Climate in the watershed is clearly diverse, varying from arid in the west to humid in the east. The terrain fluctuates severely, dropping from west to east by three levels. There are also five main vegetation zones, including forest, forest steppe, grassland, desert, and Tibetan Plateau.

METHODS

Method of calculating rational vegetation coverage rate

You *et al.* (2005) have put forward the concept of OVCR, which is calculated with three parameters, namely ecologically-allowable coverage rate, ecological demand coverage rate, and economic demand coverage rate. These three parameters are compared and optimized to find the best balance between ecology and economy.

Ecologically-allowable coverage rate is the maximum vegetation coverage rate that can be reached under local natural conditions, especially taking precipitation into account in arid and semi-arid regions. This indicator reflects the capacity of supporting vegetation under natural conditions. In the Yellow River watershed, water is the main limiting factor for plants, and water requirement of plant growth mainly depends on natural rainfall. This rate is calculated by the local water balance equation:

$$Y_1 = (P + R - E)/QS \tag{1}$$

in which Y_1 = ecological allowable coverage rate; P = regional annual precipitation; E = regional annual evapotranspiration; R = water quantity that can be extracted from rivers while keeping the ecological balance of the river and downstream reaches; Q = regional typical plantation water consumption per unit area; and S = total regional area.

Ecological demand coverage rate means the required coverage rate to sustain good local ecological conditions and prevent deterioration. Soil and water loss is the main ecological problem in the Yellow River watershed, and vegetation can effectively control soil and water loss. So in the study area, the ecological demand coverage rate indicates the threshold at which soil loss is not more than the acceptable soil loss rate. The mathematical model for the vegetation needed to control soil erosion is $M = ae^{-bF}$ and can be used to derive the formula for the ecological demand coverage rate:

$$Y_2 = -(\ln M / a) / b \tag{2}$$

in which Y_2 = ecological demand coverage rate; M = local allowable soil loss amount; a, b = regression coefficient obtained by regression analysis of local water and soil loss, respectively.

The economic allowable coverage rate means the vegetation coverage rate while sustaining the present economy of the local population. In arid and semi-arid areas, the economy and ecology both compete for land and water resources. So the lower vegetation coverage rate when meeting economic land and water usage meanwhile is established as the economic allowable coverage rate:

$$Y_3 = \min\left[(S - S_1 - S_2 - S_3)/S, (P + R - E - Q_1 - Q_2 - Q_3)/QS\right] \tag{3}$$

in which Y_3 = economic allowable coverage rate; S = regional total area; S_1 = land area for secondary and tertiary industry and urban construction; S_2 = agricultural land area; S_3 = area of land which is not fit for vegetation; Q_1 = industrial water consumption; Q_2 = agricultural water consumption; Q_3 = domestic water consumption.

The determining principle for finding the optimal coverage rate is to ensure that the ecosystem does not deteriorate preferentially and the ecology level does not fall. So the method first compares ecological allowable coverage rate to ecological demand coverage rate. When the former is less than the latter, it indicates that the study area is in a poor ecological condition due to the certain deficient resources, like water, for its bad natural endowment. Such a region is not fit for further economic development and the aim is to try the best to maintain the ecosystem. So in this case, the optimal coverage rate is equal to the ecological allowable coverage rate.

If the ecological coverage rate is more than the ecological demand coverage rate, the natural resources are adequate for ecological requirements, and the economy can be developed. Then the ecological demand and the economic allowable coverage rates are compared. If the former is less than the latter, it indicates that the region has sufficient for the ecology and the economy. In this case, the optimal coverage rate is equal to the economic demand coverage rate.

If the ecological demand coverage rate is more than the economic allowable coverage rate, the land and water resources are insufficient for meeting both the economy and ecology. In this case, the importance of ecology and economy are weighted, where the proportion is determined by the regional development programme and the optimal coverage rate. The ecological demand coverage is a threshold of an ecosystem. When this threshold is not reached, the ecosystem probably deteriorates. So we first ensure the ecological demand coverage rate, which is taken as optimal coverage rate at this time.

Method of optimal population level

Domestic and productive water consumption ultimately serves human requirements, so the water carrying capability can be expressed by the population level that the available water resources can support. Hence the optimal population level is the maximum number of people supported by available water resources with the allocation between domestic and productive water taken into consideration. In this paper, the calculation method was a modified version of a regional water resource carrying capacity model (Feng *et al.*, 2003). They thought that the regional water resources carrying capacity was determined by three factors: (a) regional water resource quantity, which generally means regional water that can potentially be supplied, (b) regional water resource exploitation and utilization capability, and (c) regional water consumption structure and level, which reflect regional social and economic structures and level of economic development. We set up an optimal population level model for the Yellow River basin as follows:

$$WC = \max\left\{\sum_{i=1}^{n}\{(W_\alpha \cdot B_i)\cdot W_{ui}/r_i\}\right\} \quad \text{S.T.} \sum_{i=1}^{n} B_i = 1.0 \quad 0 \le B_i \le 1.0 \qquad (4)$$

in which WC = optimal population level, the maximum that water can support; W_α = amount of water available in the Yellow River basin. W_{ui} is the efficacy factor of water resource use, assuming there are n social and economic aspects that depend on water resources, which is capable of supporting the ith water consuming aspect per unit of water. Here we assumed there were two aspects of water consumption regarding GDP production and domestic life in the Yellow River watershed. Water resource efficacy factors were reciprocals of per unit water of GDP production usage and domestic usage quota per person. B_i is the proportion of the ith water consuming aspect by water resource allocation, and r_i = ideal water requirement per person of the ith water consuming aspect. We assumed as given the GDP per person and the domestic water consumption quota for ideal water requirements in the Yellow River basin.

DATA DESCRIPTION

Precipitation records from meteorological stations (shown in Fig. 1) in the range of the preferential protection zones from 1980 to 2000, were collected to provide annual totals with yearly recurrence rates of 25% (wet), 50% (normal) and 75% (dry). Water extractable from rivers was calculated by taking the annual runoff in a control watershed and subtracting the minimum ecological flow. For most parts, minimum ecological flow was taken as 10% of the observed average flow. Minimum ecological flow was taken as 30% of observed average flow in the regions above Tangnaihai, which have a fragile ecosystem. According to the Classification Standards of Soil Erosion (S190-2007), when erosion strength is slight, the average erosion modulus is 1000 t km$^{-2}\cdot$year^{-1}, which was taken as the allowable soil erosion quantity in the protection zones. Industrial and agricultural land use areas were used to determine the economic allowable coverage rate. Present land use was obtained from the land-use map of 2006 in the Yellow River basin, which was determined with remote data of the Beijing-1 satellite.

For the optimal population level analysis, annual water availability of every preferential protection zone was obtained for the period from 1980 to 2000, and wet, normal, and dry years were determined. Efficacy factors of the protection zones were determined by the water use levels of 2006.

RESULTS AND ANALYSIS

Delineating watershed

The Yellow River basin has a large watershed area, and there is spatial variability in the exploitation and utilization of water resources and importance of different ecosystems. It is necessary to regionalize the basin according to ecosystem importance, hydrological characteristics, ecological background and human activity levels. To allow for the ready analysis of ecological protection objectives and regulations, regionalization was based on the county boundary map. There were four kinds of regionalizing indicators: ecological importance (forest coverage rate, biology diversity, soil erosion modulus, wetland, nature reserves), hydrological characteristics (precipitation, evaporation, runoff coefficient and groundwater resource modulus), ecological background (terrain, air temperature and sunshine hours, etc.) and human activity level (population density, domestic water consumption per person and productive water consumption per person). The watershed was regionalized into 51 zones, whose ecological importance was assessed by the RIAM (Rapid Impact Assessment Matrix) model (Christopher, 1998). Thirteen zones were identified as preferential protection zones to calculate ecological protection objectives as shown in Fig. 2. The four-digit numbers of each zone in the figure represent the code, and the names are shown in Table 1.

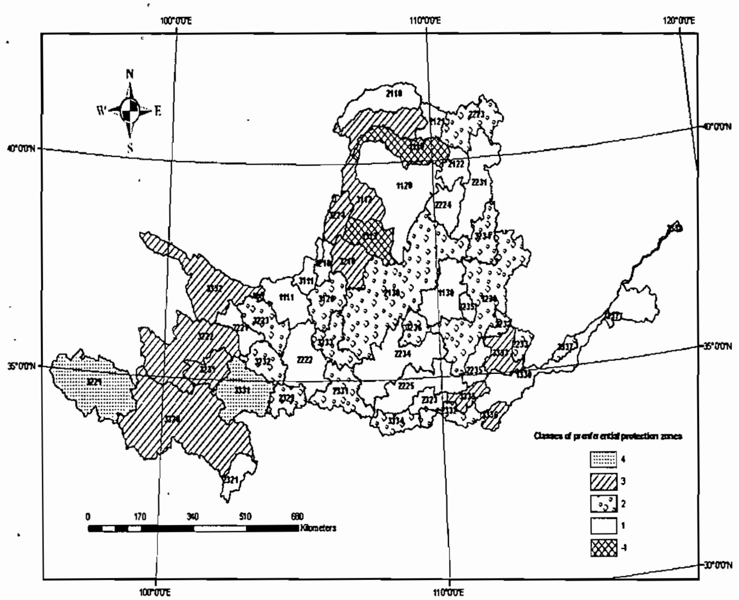

Fig. 2 Regionalization and preferential protection zones of the Yellow River basin.

Table 1 Code and name of preferential protection zones in the Yellow River basin.

Code	Name
3221	Maduo and the Up Reach
3331	Tao River and Xia River
3320	Between Maduo and Ruoergai
3222	Trunk between Tongde and Xunhua
3231	Tao River Source
3332	Huangshui River
3210	Qingshui River and Kushui River
3224	Ningxia Irrigation Zone
3112	West Odors and Hetao Irrigation Zone
3333	West Qin River and Henan Yellow River Wetland
3335	Little North Trunk and Sanmenxia Reservoir
3336	Henan Funiu Mountain
3338	Estuary Delta

Optimal vegetation coverage rate

The OVCR of preferential protection zones is shown in Table 2. Empty cells in Table 2 indicate that the corresponding water condition could not sustain vegetation growth. These conditions were found in the Qingshui River and Kushui River Zone, West Odors and Hetao Irrigation Zone, Ningxia Irrigation Zone, and during dry years in the Estuary Delta Zone and Henan Funiu Mountain Zone.

To find the difference between the desired and present status, OVCR in normal years and the present coverage rate were compared. The Tao River and Large Xia River Zone's OVCR was the highest at 79.9%, while in the Hetao Irrigation Zone it was the lowest at 22.3%. In the remaining zones this varied from 29.2% to 77.9%. The Tao River and Xia River Zones also had the maximum present coverage rate, which was 92.3%, and the present rate in the Estuary Delta Zone was the lowest at 7.2%. Present coverage rate of the zones above the Huangshui reach were more than 70%, and in these zones the present rates were mostly more than their OVCR. The present rate of zones in the upper-middle reach varied from 20% to 70%. There were five zones where the present coverage rate was less than OVCR of a normal year, such as the Huangshui River Zone with a difference of 7.2%, the Ningxia Irrigation Zone with a difference of 13.3%, the West Qin River and Henan Yellow River Wetland Zone with a difference of 16.1%, the Henan Funiu Mountain Zone with a difference of 5.8%, and the Estuary Delta Zone with a difference of 63.5%.

Table 2 Vegetation coverage rate of protection zones in the Yellow River basin (%).

Code	Y_0	Y_1 (25%)	(50%)	(75%)	Y_2	Y_3 (25%)	(50%)	(75%)	Y (25%)	(50%)	(75%)
3221	71.9	53.0	38.0	17.5	74.8	51.8	36.4	13.5	53.0	38.0	17.5
3331	92.3	87.0	79.9	73.0	57.3	83.1	79.9	68.4	83.1	79.9	68.4
3320	88.3	54.6	43.6	30.7	57.3	53.2	41.8	28.1	54.6	43.6	30.7
3222	70.7	94.5	87.7	81.5	57.3	70.8	70.8	70.8	70.8	70.8	70.8
3231	90.6	76.4	64.5	52.1	57.3	75.6	63.5	50.9	75.6	63.5	52.1
3332	70.5	88.4	81.4	74.7	65.5	77.9	77.9	73.4	77.9	77.9	73.4
3210	56.8	66.4	30.0		33.6				33.6	30.0	
3224	31.6	93.4	81.1	69.4	44.9				44.9	44.9	44.9
3112	37.6	58.2	22.4		53.0				53.0	22.4	
3333	28.4	100	97.7	66.3	44.5	28.5	28.5	15.7	44.5	44.5	44.5
3335	60.9	68.2	52.1	31.9	38.3	60.9	42.4	9.7	60.9	42.4	31.9
3336	50.7	77.7	56.5	29.2	72.6	50.7	34.5		72.6	56.5	29.2
3338	7.1	100	98.5	40.8	70.6	7.1	7.1		70.6	70.6	40.8

Y_0: present vegetation coverage rate; Y: optimal vegetation coverage rate. 25%, 50% and 75% stand for occurrence rates of yearly precipitation (wet, normal, dry).

The five zones with present coverage rates that do not meet the OVCR were analysed as follows. In the Huangshui River Zone, most of the land and water resources ware exploited less intensively where there was potential to plant more vegetation to increase its OVCR.

The Qingshui River and Kushui River Zone (3210), the West Odors and Hetao Irrigation Zone (3112) and the Ningxia Irrigation Zone (3224) are situated in the Ningxia and Inner Mongolia dry zone, where average annual precipitation is only 170 mm to 290 mm. Precipitation is consumed for evaporation in dry years, and soil moisture cannot sustain vegetation. Water shortage is the main reason for ecological vulnerability. Also domestic water use and productive water use, especially irrigated agriculture, consumed much water, which aggravated ecological water shortage.

Present coverage rates of the Henan Funiu Mountain Zone and the West Qin River and Henan Yellow River Wetland Zone were 50.7% and 28.4%, respectively. Soil in these regions is loess, which is easily eroded by water. So the demand of vegetation coverage rate to prevent soil erosion was higher, and correspondingly the optimal coverage rate was higher.

The natural water condition in the Estuary Delta Zone can sustain a vegetation coverage rate of 41% in dry years. According to local soil characteristics, the coverage rate must be more than 70% to ensure that soil erosion is limited. The present coverage rate was only 7.1%: The large gap between these is caused by agriculture, which occupies 79% of the land area.

In the zones whose ecological allowable coverage rates were less than the ecological demand coverage rates, the ecological condition was vulnerable. Ecological allowable coverage rates in normal years and ecological demand coverage rates were compared. Fragile zones included the Maduo and the Up Reach Zone, the Maduo and Ruoergai Zone, the Qingshui River and Kushui River Zone, the West Odors and Hetao Irrigation Zone and the Henan Funiu Mountain Zone. The Maduo and the Upper Reach Zone and the Maduo and Ruoergai Zone lie in the source region of the basin, where soil parent material is sandy slate, argillic limestone and weather gravel deposits, which are easily reduced to sand and eroded by wind.

Optimal population level

Water use was divided between domestic water consumption and water consumption for GDP production. According to existing water using quota (Chen, 2004; Yuan *et al.*, 2007), water requirement quota of adequate, well-off and rich level were determined as shown in Table 3. The optimal population level that can be supported by available water resources for each protection zone was calculated by the water resource capacity model, and the results are shown in Table 4.

Table 3 Living level standard and per capita domestic water consumption quota.

Indicators	Simple adequate	Well-off	Rich
Per capita GDP (Dollar)	500	1000	4000
Domestic water consumption quota (L/d)	120	260	300

Table 4 Optimal population level in preferential zones (10^4).

Code	Rich			Well-off			Simple adequate			Present
	25%	50%	75%	25%	50%	75%	25%	50%	75%	
3221	222	163	113	488	358	249	1026	752	523	25
3331	43	37	32	78	68	58	165	143	123	283
3320	351	263	189	756	565	404	1587	1186	850	181
3222	232	171	119	512	375	261	1075	788	548	26
3231	66	49	34	146	107	74	306	225	156	8
3332	174	130	93	383	286	204	804	600	429	38
3210	38	37	36	90	88	86	189	183	179	119
3224	47	45	44	117	112	108	243	233	226	143
3112	30	29	28	80	78	76	169	163	159	135
3333	151	128	108	314	267	225	670	568	478	304
3335	144	115	89	246	196	152	525	419	325	119
3336	227	188	148	398	330	260	850	704	555	166
3338	7	6	5	26	23	19	56	48	41	22

25%, 50% and 75% stand for occurrence rates of yearly precipitation (wet, normal, dry).

For comparison, present population density and optimal population density at present economic levels in a normal year are displayed in Fig. 3. Present economic levels of the protection zones were evaluated by taking the per capita GDP as standard. Of the 13 protection zones, in the Tao River Zone the level was adequate, the Estuary Delta Zone has reached a rich level, and the other zones are well-off. It can be seen from Fig. 3 that there were obvious differences among zones with respect to optimal population density. The Henan Funiu Mountain Zone has the highest density with 662 people per km^2, West Qin River and Yellow River Wetland in Henan Zone followed with 228 people per km^2, then Ningxia Irrigation Zone with 136 people per km^2. The next eight zones, mainly the ones lying upstream and in the delta had values from 50 to 100 people per km^2. The West Odors and Little North Trunk Zone had the lowest density of 27 people per km^2.

Present population density within zones varies largely as well, with a maximum of 422 people per km^2, and a minimum of 8 people per km^2. The population was denser in most zones on the middle and lower reaches, and population density in upstream zones was generally less, with the exception of the Tao River and Large Xia River Zone, with 114 people per km^2.

There were several zones whose present population density was more than the optimal population density, such as the Tao River and Large Xia River Zone, with an excess of 81 people per km^2, the Qingshui River and Kushui River Zone with an excess of 30 people per km^2, the Ningxia Irrigation Zone with an excess of 43 people per km^2, the West Odors and Hetao Irrigation Zones with an excess of 21 people per km^2, the West Qin River and Yellow River Wetland in Henan Zone with an excess of 83 people per km^2 and the Estuary Delta Zone with an excess of 182 people per km^2.

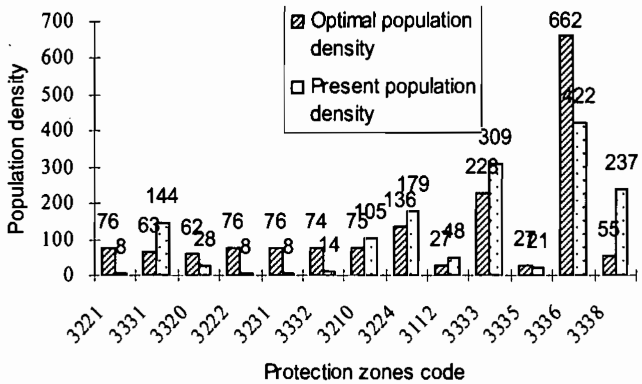

Fig. 3 Optimal and present population density of protection zones.

CONCLUSIONS

The selected preferential protection zones play important roles in the ecosystem of the Yellow River basin. OVCR and optimal population levels of the protection zones have been analysed using the optimal coverage rate and the water resources carrying capacity methods. The main results can be summarized as: (a) in most preferential protection zones, present vegetation coverage rate reached the OVCR. Due to poor natural water conditions and excessive agricultural water consumption in Ningxia and Inner Mongolia, vegetation cover did not meet OVCR. Also, the high soil conservation demand in Henan at the middle reach and the large proportion of farmland in the Delta at the lower reach caused vegetation cover levels to be lower than the corresponding optimal rates. (b) There were six zones where the present population level exceeds the optimal level given the carrying capacity of available water resources: the Tao River and Large Xia River Zone, the Qingshui River and Kushui River Zone, the Ningxia Irrigation Zone, the West Odors and Hetao Irrigation Zone, the West Qin River and Yellow River Wetland in the Henan Zone and the Estuary Delta Zone. Especially, in Ningxia and Inner Mongolia, the Henan region in the middle reach and the delta, the present status of both vegetation coverage rate and population scale did not reach optimal levels.

We have presented a method to obtain regulation target values for sustainable water resources utilization. There is still room for further improvement. The water resources carrying capacity model for optimal population level is simple and the GDP water consumption and domestic water consumption do not reflect the complete social and economic implications of water use. A next logical step would be to include industrial and agricultural water use data to improve the model.

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Indirect water management: how we all can participate

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Abstract Life Cycle Assessment (LCA) represents a methodological framework for analysing the total environmental impact of any product or service in our daily life. After tracking all associated emissions and the consumption of resources, this impact is expressed with respect to a few common impact categories. These are supposed to reflect major societal and environmental priorities. However, despite their central role in environmental processes, to date hydrological and hydrogeological aspects are only rarely considered in LCA. What are the reasons? The origin of LCA plays a major role; it has been mainly applied in the industrial sector. Here, if at all, water turnover and use is described, but less emphasis is paid to the related effects. This incompleteness can be also found in water footprint or virtual water based evaluations. Our approach, presented here, fills this gap, and reveals how a revised LCA and the related water footprinting can serve as a consistent baseline for indirect water management involving producers, consumers, and local stakeholders.

Key words IWRM; LCA; freshwater use; impact assessment; wheat; agriculture; virtual water

INTRODUCTION

Integrated water resources management (IWRM) stands for a systematic approach for sustainable, equitable, and efficient development of water. It promotes the coordinated implementation of multifaceted programmes that address water supply and demand, while complying with environmental requirements in a basin or a watershed (e.g. Meire *et al.*, 2008). According to the Dublin principles, IWRM should be based on a participatory approach, involving users, planners, and policy makers at all levels. Its challenges are to combine approaches and standpoints of multiple disciplines, to understand and predict the characteristics of each particular real case, and to involve the variety of associated stakeholders. In this sense it offers recipes for local or regional management plans, and thus commonly is constrained to a limited area. The principal perspective of water management is unidirectional, and originates from the case-specific problems that need to be solved. Thus, solution strategies focus on direct effects. In this fashion, the role of external pressures may be acknowledged but is taken as given and is not subjected to further discussion.

In this paper, we focus on these "external pressures" and present a concept that also enables control of the underlying factors. Naturally, the scale of such a different perspective will be coarser than usual in IWRM applications, and is only complete on a global level. As will subsequently be shown, an orientation for the smallest scale, for individuals' daily decisions, can be derived by using a holistic concept. Even if not living within one particular watershed, people can ultimately exert indirect water management. The methodology presented follows life-cycle thinking, and hence is embedded in a standard life cycle assessment (LCA) framework. In the following section, the underlying principles of LCA will be explained, and its link to established IWRM practice will be elaborated. As a side effect of embedding indirect water management in LCA, trade-offs with other environmental concerns can be shown in a consistent manner.

LIFE CYCLE ASSESSMENT, LCA

LCA is a modular procedure to quantify and compare the environmental burdens of any product, service, or good. It is based on ISO standards (ISO 14040 series), and involves sequential procedures such as "Goal and Scope Definition", "Inventory Analysis" (LCI), "Impact Assessment" (LCIA) and "Interpretation" (e.g. Guinee *et al.*, 2002, Fig. 1). The purpose is to fully examine the environmental burdens or benefits, that is, those that stem from the entire life cycle of a product, from cradle-to-grave. For example, a bottle of wine would not only be characterized by the net water in the bottle, but all the water that is spent for grapevine irrigation, grape-crushing, as well as water that is consumed for glass production, bottling, transport and all the effects from recycling and disposal. LCA follows a holistic paradigm, often at the expense of accurate estimations. This fact does not necessarily mean a disadvantage, as it gives LCA the space to perform a combined investigation of multiple different safeguard subjects and assessment criteria. These are distinguished by different standard impact categories such as global warming, ozone depletion, land use, depletion of natural resources, and toxic effects. The normative values that reflect the impacts within each category are scientifically based, but often have to be understood as rough reference figures for comparative assessment. This is due to the specific assumptions for calculating indicators and their commonly approximate nature.

Compared to standard impact categories within LCA, water is special. In contrast to other abiotic resources such as coal or crude oil, it can be replenished. In contrast to CO_2 emissions, there exists an immense spatial dependency of impacts on natural water bodies. Total global freshwater resources are sufficient, but not evenly distributed and often scarce in regions of high demand. Natural resources are difficult to assess if their value is not just local (e.g. on a community or industry scale), but on a societal level (like health effects). Setting up functional relationships in order to derive a generally valid and practicable evaluation is tedious due to the complex, insufficiently understood, and uncertain natural processes involved. This is also true for those effects and processes connected to natural water systems.

LCA that includes the environmental effects of water use and consumption means global *indirect* water management. It supports goal-directed consumer behaviour that aims to reduce pressure on natural water systems. By developing a hydrologically-based assessment of potential impacts from human interaction with natural water bodies, "greener" products can be favoured. Let us consider the wine example: for different wine labels, tracking the effects of irrigation in different grape cropping regions would reveal that brand with the lowest local impacts. If consumers are not only guided by taste but also such an "ecolabelling", they could choose products based on sustainable and environmentally friendly water management. The focus of such an assessment can be on comparisons between regions but also on absolute values (thresholds).

In the following we briefly review some studies that are dedicated to the question of how to account for hydrological and freshwater issues in common LCA concepts. Then, we concentrate on the most water consumptive sector, agriculture, and show how to compile the water-efficiency of cropping and regional water availability. As an illustrative example, global wheat farming is chosen, as wheat is one of the crops with the greatest acreage worldwide.

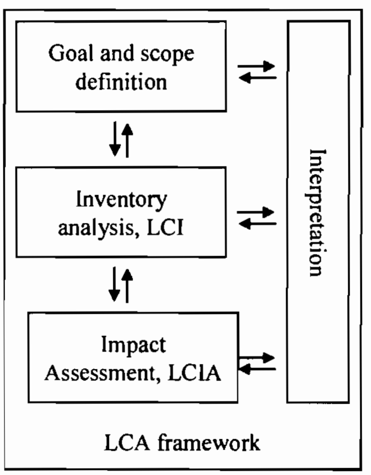

Fig. 1 LCA components (adopted from ISO 14040, 2006; ISO 14044, 2006).

RELATED WORK

Despite the complexity of accounting for the impacts in hydrological environments in a both comprehensive and expressive way, several approaches have been suggested to overcome the rudimentary state of the art. For example, Owen (2002) presented a conceptual overview of requirements for integration of water resources into LCA. This work represents a general introduction into this transdisciplinary field, and structures potential impact types related to water quantity and quality. However, it focuses on a verbal discussion and does not embody substantial methodological advancements. The simplified method for water assessment by Bösch *et al.* (2007) is oriented toward theoretical cumulative energy demand, but does not account for regional differences related to water use. As stated above, such regional dependencies are essential for balancing water resources (Vörösmarty *et al.*, 2005), particularly for products with a globalized value chain. Lundin & Morrison (2002) focused their LCA on urban water systems, and presented an impact assessment methodology suitable to urban water supply. Other studies, even when water supply is of central concern, simply place raw water extraction from aquifers beyond the system boundaries (e.g. Landu & Brent, 2006).

Hydrological impact assessment is discussed in several studies (e.g. Shiklomanov, 1997; Vrba & Lipponen, 2007), but such judgements have not been incorporated in a LCA framework so far. The distance-to-target method of Ecological Scarcity 2006 accounts for regional aspects by assessing freshwater use on a country level (Frischknecht *et al.*, 2008). National water-stress values are used to derive impact factors based on a defined threshold. While this method is a good first step to quantitatively assess potential water stress, it does not differentiate between water consumption (e.g. evaporation) and other water use (e.g. use of water as a cooling agent, returning the water to the watershed after use). Vince *et al.* (2008) describe an LCA approach for potable water production. However, effects such as reservoir depletion due to freshwater extraction or ecosystem impacts are not evaluated.

LCA of water use is closely related to virtual water principles and assignment of water footprints to products or countries (Chapagain & Hoekstra, 2004). The underlying idea is to quantify the total water that is needed within a lifecycle and thus, to calculate virtual water contents following a concept equivalent to LCA. However, and this is a fundamental difference, by only summarizing the water volumes spent during a product's life cycle, a general statement cannot be made about the severity of the related environmental impacts. In fact, virtual water volumes may be even misleading, by suggesting that water use and consumption is directly proportional to environmental burden. This rigorously ignores the regional dependencies of freshwater resources, and neglects the fact that water turnover and extraction can even be advantageous, such as in wet seasons or areas. For agricultural products, available virtual water databases represent comprehensive information sources, which report water requirements for several crops and countries. Still, these data cannot directly be used in LCA. For example, we need to quantify irrigation water and not total crop water requirements for tracking the water impacts from agriculture, as on-site precipitation is indirectly assessed by land use. In the following example, we demonstrate how such a conceptual modification influences the assessment.

EXAMPLE STUDY: GLOBAL WHEAT FARMING

Agricultural production is one of the most important economic activities and responsible for about 70% of the global anthropogenic freshwater withdrawals, while only 20% and 10% are used by industry and the municipalities, respectively (WB, 2004). Furthermore, freshwater scarcity has been recognized as one of the most crucial environmental issues (UNESCO, 2006) and several regions around the world are already facing this problem.

Global agriculture is facing substantial pressure from increased food and bio-energy demand affecting water and land availability. Globalized trade also requires global water resource management approaches, as local water management can hardly cope with the global market demand and its adverse effect on the producing regions. Conservation activities in one catchment might lead to greater water import from other regions where the pressure on water might be higher. In addition to focussing on optimal water resource management in single catchments, the global dimension of agricultural trade and embedded water has to be considered to arrive at meaningful assessments of water resource use. The key question is this: how can we optimize the water efficiency of crops and thereby minimize environmental impacts in the global context?

The virtual water concept and water footprint calculations are intended to tackle these questions. However, the underlying concept has so far been limited to the differentiation of blue (irrigation) and green (soil moisture) water. In order to account for the environmental impact caused by water use for crop production, we suggest calculating water deficiencies. The purpose of these calculations is to quantify the severity of the consumed water with respect to environmental issues and describe unsustainable water management involved in the production of a good. We developed a comprehensive environmental impact assessment method for water consumption which evaluates the damage potential regarding ecosystem and human health for major watershed levels worldwide (Pfister *et al.*, 2009). This assessment is based on a combination of different vulnerability studies and water scarcity measures with global coverage. A crop's content of "deficit water" is calculated by applying the resulting, spatially-resolved, impact factors on blue water data.

We computed regional irrigation requirements at a high level of spatial resolution (10 arc minutes grid), based on remote sensing data for the 50 globally most traded products. Deficit water footprints of crop production were generated for each grid cell, for each country, and based on FAO-based international trade data for the average product on the global market. For instance, one ton of wheat on the global market carries a virtual deficit water content of 0.23 m^3/kg and a virtual blue water content of 0.98 m^3/kg (Fig. 2). For the top ten wheat exporting countries, these values vary between 0.03 and 0.83 m^3/kg for deficit and 0.23 and 3.3 m^3/kg for blue water, respectively.

Figure 2 shows the global maps for the related impacts. These maps depict the differences of applying standard virtual water volumes and a deficit based approach. As expected, when considering deficit water, there is substantially higher impact in the dry regions of northern Africa and central Asia. In contrast, wheat production in countries such as Russia, Canada and Brazil is judged as more environmentally sound. In these countries, on the average, freshwater resources are sufficient and thus less threatened by intense agricultural activities. These discrepancies indicate the need for a more comprehensive impact-oriented perspective and the potential of global assessment approaches to support a balanced water management.

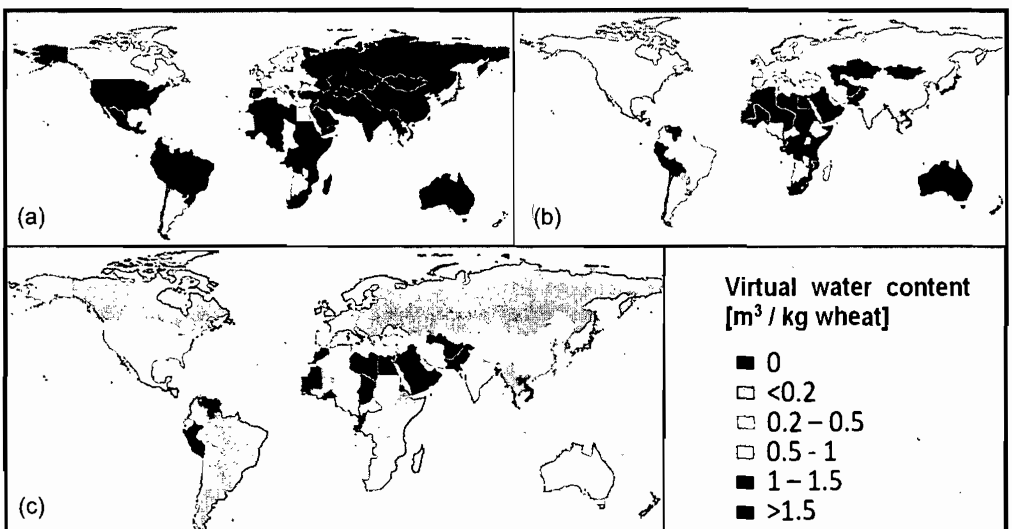

Fig. 2 Virtual water content of wheat: (a) total virtual water, (b) virtual blue water, and (c) virtual deficit water.

CONCLUSIONS

The proposed contribution reviews the state of the art in a new interdisciplinary field: the integration of environmental assessment of freshwater use in Life Cycle Assessment (LCA). It is demonstrated that current concepts, such as standard water footprinting, have to be improved to reflect the true environmental burdens inherent to freshwater-demanding products. This is scrutinized by computing the standard and deficit virtual water contents of global wheat cropping. The new deficit-oriented perspective represents a promising and more plausible concept to identify those daily-life products of higher impacts.

Consumers, end or mid-point users and any protagonist of the global supply chain have the power of choice. Utilizing a realistic, hydrologically-based operational assessment method and including this into a common LCA-framework is essential to arrive at meaningful ecolabels or sustainability certifications. If we consider relative water deficits, we establish a direct connection to case-specific IWRM, and ultimately give control to any actor on the scene to exercise indirect water management. A standardized procedure and assessment of water in LCA further helps to illuminate additional and less subjective (e.g. political) aspects of water allocation problems.

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3 Water Quality

(Proc. of JS.3 at the Joint IAHS & IAH Convention, Hyderabad, India, September 2009). IAHS Publ. 330, 2009.

Studies on the quality of the waters of Morocco and training in environmental geochemistry

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Abstract The collaboration of the Department of Earth Sciences of the University of Cagliari, Italy, with the Ministry of Energy and Mines of the Reign of Morocco initially, and later with the University Mohammed V of Rabat-Agdal, started in the mid 1990s. The objective of the collaboration project was two-fold: (i) the hydrogeochemical characterisation of the qualities of Morocco's surface and groundwaters, and (ii) the training of technicians and researchers in Environmental Geochemistry, in particular in the protection of water resources. Morocco faces a serious challenge in terms of water resources management over the near- and medium-terms, both in terms of quantity and quality. The Sebou River basin holds 30% of the country's surface water resources and 20% of its underground water. Although it represents only 6% of the total land area of Morocco, 18% of the country's population live in the basin. It is therefore very densely populated and polluted. The basin is very rich in water resources, agricultural land and forest reserves, with the Middle Atlas as its highlight. It also has great tourism and economic potentials and contains some major economic hubs, such as Fès, Meknes and Kenitra. All of these activities result in significant wastewater discharges from household, industrial and agricultural sources. In the catchment of Oued Sebou, the most important watercourse of northern Morocco, the Sebou River is the primary source of water for a variety of purposes (i.e. drinking, agriculture, industry, recreation). However, it receives waste from vast rural areas and several large cities along its banks. We analysed different physico-chemical parameters to determine the level of pollution of the Sebou River basin. The results indicate that sites located close to the most urbanized and industrialized areas are severely impaired. This often condemns tens of kilometres of the Sebou and leads to a threat and an important handicap for some reaches of water located downstream of the towns of Fès and Meknés. At times the groundwater presents anomalous values which can be attributed to the fertiliser and antiparasite compounds used in agriculture, in the case of NO_3 and some metals (Cu, Hg).

Key words Sebou; surface water; underground water; wastewater; downstream; untreated wastewaters; pollution; Fès and Meknés, Morocco

INTRODUCTION

The Sebou River rises in the Middle Atlas, with the name of Oued Guigou, and flows for about 500 km towards the Atlantic Ocean, receiving water from various tributaries on the way, of which the main ones are oueds Ouergha, Inaouène, Lebene, Mikkes, Rdom and Beht. Annual average precipitation varies from 400 mm in the High Sebou and the valleys of the Beht, to 1800 mm on the Rif. The course is irregular in space and time; the High Sebou, upstream of Allal Al Fassi Dam, is characterised by a perennial flow, while other tributaries, especially Ouergha and Inaouène, have a seasonal regime depending on precipitation, with important floods. Despite the high annual evaporation, the Sebou basin is one of the richest in water resources in Morocco, as it amounts to an average of 5000 million m^3, corresponding to 28% of the national freshwaters. Groundwaters also contribute to the hydraulic importance of the basin to the country, with 20% of the national assessed quantity, and an annual exploitable volume of 800 million m^3. The volume of water utilized at present in the Sebou basin is about 1680 million m^3, including 1000 million m^3 from freshwaters and 680 million m^3 from groundwater sources.

Agriculture is the most important economic activity in the basin and the main water-consuming sector, with annual water supply for irrigation of 1500 million m^3, 970 million m^3 from freshwater (97% of all, mobilised freshwaters), and 530 million m^3 from groundwater. The usable agricultural area in the region is 1 880 000 ha (20% of the national potential). The irrigable area is 375 000 ha, of which 268 800 ha are now irrigated (116 800 ha from large irrigation schemes and 152 000 ha from small and medium irrigation schemes) (Agence du Bassin Hydraulique du Sebou, 2003).

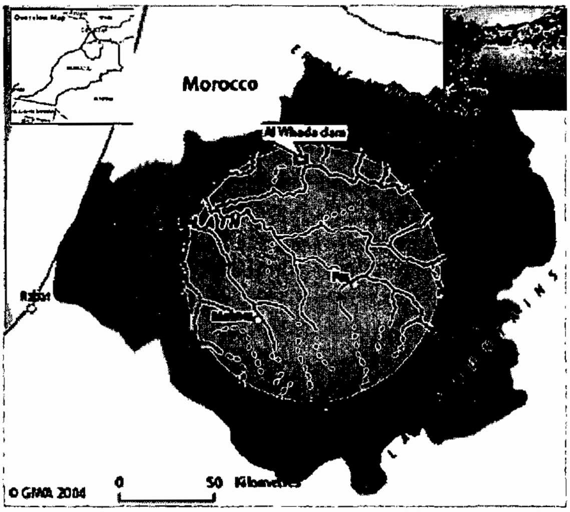

Fig. 1 Sebou River basin.

The basin (Fig. 1), some 40 000 km^2 in area is among the most densely populated regions in Morocco. Within the basin there are large urban centres, extensive agricultural areas and both modern and artisan industries, all of which contribute to the basin's water pollution problems. The major urban centres in the upper basin are Fès and Meknes. The lower basin meets the sea near the cities of Kenitra, Rabat and Sale. Agricultural activities centre around Gharb where two million hectares, 25% of the country's arable land, are cultivated. Approximately 200 industries of varying scale operate in the Sebou basin. There are thus a range of pollution sources: urban, agricultural, industrial, and artisan; and a range of organic, chemical and heavy metal pollutants (Arthurton *et al.*, 2002).

GEOLOGICAL SETTING

The geology of the basin is rather complex. The Palaeozoic base consists of metamorphic rocks such as schist and quartzite, and of magmatic rocks of the Cambrian and Permian (Ternier *et al.*, 1950) (Fig. 1).

The Mesozoic rocks are marls and red gypsiferous and saliferous Triassic clays flush in the Middle Atlas, and reach 700 m in the plateau of Meknes (*Ressources en Eau du Maroc*, 1975). The Lias consist of the sandy dolomites, marly dolomites and the bedrock of dolomites of the Lower Lias (Essahlaoui, 1997), and limestones of the Middle Lias, and marly limestone of the Upper Lias and of the Dogger. The Jurassic in the basin has a thickness of 250 to 300 m Essahlaoui, 1997).

The Cretaceous is characterized by a calcareous facies. The Tertiary is mainly represented by the Miocene grey marl which can reach 2000 m in thickness. It constitutes an aquifer with groundwater in the Gharb plain (Kacimi, 1992). It is also formed Missinien marls which appear in Oued El Kell. The Pliocene is represented by sands (Ait Brahim, 1991) with a variable thickness between 60 and 80 in the Sais plain, by sandstone and conglomerates, and by lacustrine sediments (Kamal & Chehtan, 1993). The Quaternary period is represented by river terraces, silt and alluvium.

The Sebou basin has three principal aquifer (Fès/Méknès, Maamora and Gharb). The heterogeneous Plio-Quaternary coastal aquifer of the Mamora basin is the most significant aquifer in Morocco. It is composed of sandstones, conglomerates, limestones and more or less argillaceous sands.

THE SEBOU BASIN WATER QUALITY

The Sebou is the most polluted river of the country and partially caused by the municipal discharges of Fès, Meknès, Sidi Kacem and Sidi Slimane (BCEOM, 2003). More than 50% of the sampling sites present bad–very bad quality, due to organic matter (BOD5 and COD) and phosphorus.

As for groundwater quality, the average values are good to medium for the three principal aquifers (Fès/Méknès, Maamora and Gharb) (Amharref *et al.*, 2007); but locally high concentrations of nitrates (more than 50 mg/L) occur in agricultural zones with strong agricultural activity, and high salinity occurs in the Gharb coastal section caused by high abstraction rates (Matee, 2006).

The Sebou River conveys the discharges of oil-works, tanneries and sweet manufacturers which are concentrated there, as well as all the domestic discharges of the city of Fès, causing a net degradation of the quality of the Sebou River water. This deterioration reaches a critical threshold during the period of active oil-working. On account of these important discharges, the regime of dissolved oxygen in the river of Sebou remains overdrawn for more than 70 km downstream of Fès. Anoxic states (0 mg 0_2/L) are registered regularly during the period of oil-works activity (October–March), in the course of which the load of oxidizable material expressed in DCO reaches values of the order of 600 mg 0_2/L.

The degradation of the quality of the Sebou sometimes reaches levels such that the processing capacities of both existing water treatment stations are widely exceeded. This causes stops of these stations, and treatment costs (for reagent and extra energy) increase from \$0.14/m^3, to \$0.7/m^3.

Most of water used for irrigation in the Gharb area comes from the Sebou River (*Ressources en Eau du Maroc*, 1975). Unfortunately, the domestic wastewaters of the city of Kenitra (north-west Morocco) are emptied into the Sebou River without preliminary processing (Fig. 2) and contribute to significant pollution (Malaki *et al.*, 2001; Saddiki, 2002; El Wartiti *et al.*, 2007).

In the upper basin, urban domestic waste and artisan industrial waste are the predominant sources of water pollution. The city of Fès produces large volumes of untreated domestic waste and artisan waste such as tannery chemicals, resulting in particularly severe pollution downstream from Fès. Preliminary research suggests the Sebou River may be dead for the first 100 km below Fès. Levels of chromium and lead in the water downstream of Fès are 10 to 100 times internationally acceptable levels.

The lower Sebou basin is one of the most polluted areas in the kingdom, and has been since the 1970s. The primary sources of water pollution in this area are fertilizers and pesticides, and agro-industrial waste from sugar and olive oil processing facilities.

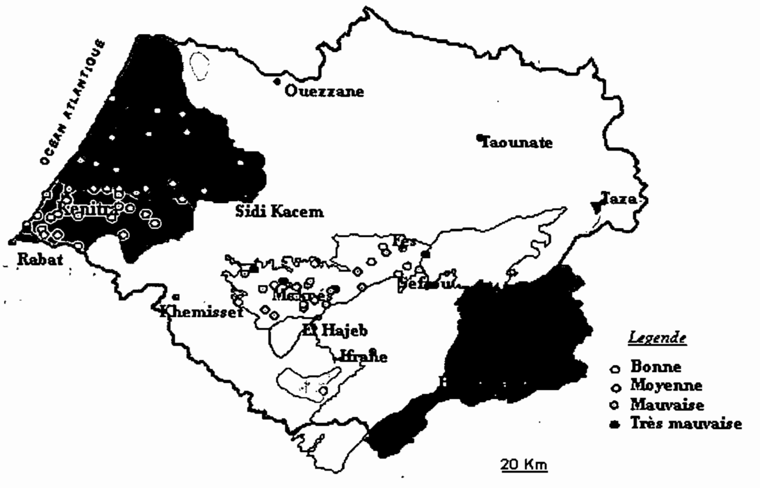

Fig. 2 The Sebou basin water quality.

Groundwater sources are also affected. In Gharb, saline, chloride and nitrate levels in the groundwater are generally very high. Outside Fès, Sidi Slimane, Souk el Arbaa and Si Yahya du Gharb groundwater is considered hazardous, particularly with regard to nitrate levels. The ground water of Saiss is evolving in the same direction. The quality of both surface and groundwater in the Sebou basin is declining due to pollution from urban, industrial, and agricultural sources. As for underground waters, which are more often geologically protected, they are generally of better quality. However, important underground water aquifers on the Sebou basin are polluted by excessive use of fertilizers and pesticides from the agriculture sector. The quality of underground water suffers from excessive pumping, especially on the coastal border.

Critical pollution levels have been reached near the cities of Fès, Meknes, Sidi Kacem, Sidi Slimane, Taza, Khemisset and Tiflet. Domestic household wastewaters constitute the principal source of organic pollution (BOD5 and COD). Nitrate and phosphate pollution is of domestic and agricultural origin. In the agricultural areas, seasonal increases in agroindustrial processing wastes coincide with annual low water levels, increasing the concentration of pollutants and magnifying their effects. In addition to the organic wastes generated by urban areas, bacteriological and toxic wastes also pose serious threats to the local populations.

Piper diagram

The classification of the surface waters of Sebou is represented in the Piper diagram (Fig. 3). This diagram shows that the water is characterized by a composition intermediary between chloride and bicarbonate (for anions) and calcium and alkali (Na) (cations). Some samples differ from this; in fact we can see that the water is rich in alkaline earth bicarbonates (water samples of Fès-Meknes), the chemical composition is related to the existence of an interaction with the dolomitic limestone formations of Mesozoic.

Moreover, the diagram relating to the anion analyses shows an evolution in the water's composition of water from a prevalence of bicarbonate on the level of the Middle Atlas to a

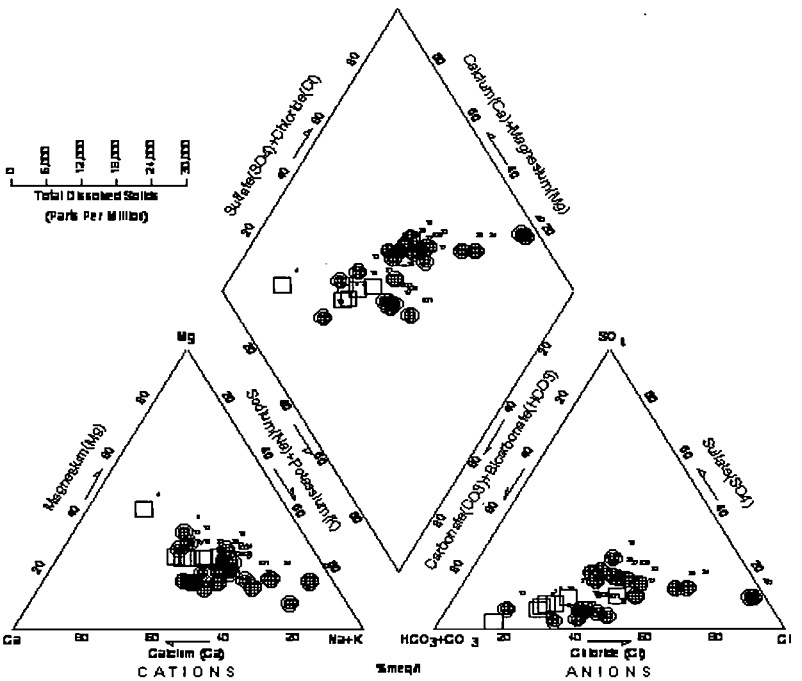

Fig. 3 Piper diagram of the surface waters of the Sebou River. Black circles: upper Sebou; grey circles: low Sebou; squares: wadi.

prevalence of chloride in the Gharb. Whereas, the samples of water taken in Prérif, Gharb and the central plate have a chemical composition dominated by the chloride and alkaline (Na), which could be explained by leaching of the marly marine sediments of the Tertiary and the Quaternary age (PréRif, Gharb, central high plateau) which contain injections of salt deposits and Triassic gypsum.

Salinity

Figure 4 shows the spatial variation of TDS in surface water. It ranges from 2725 mg/L in Lebène, to 492 mg/L in the Inaouene. In the Fès system, TDS varies between 500 mg/L and 1100 mg/L, while in the Meknès system, it varies between 450 mg/L and 1000 mg/L.

The TDS increases immediately after the domestic and industrial discharges of Fès (Fig. 4). It decreases after the confluence with the Ouerrha because its waters are less saline and dilute the Sebou waters. However, it remains constant after the confluence with Lebène and Innaouène because the TDS laden waters of Lebène are diluted by less saline waters of Innaouène before arriving at Sebou (El Wartiti *et al.*, 2007).

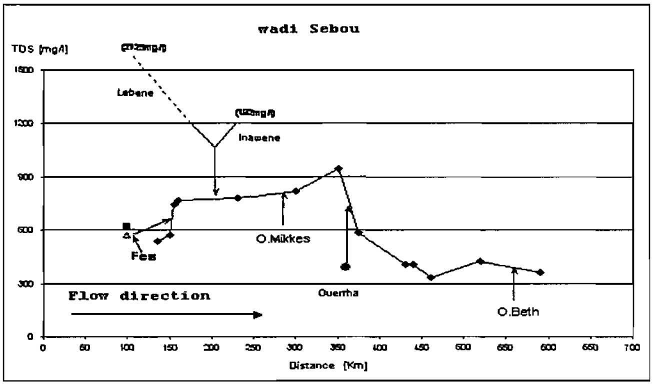

Fig. 4 Evolution of salinity along the wadi Sebou.

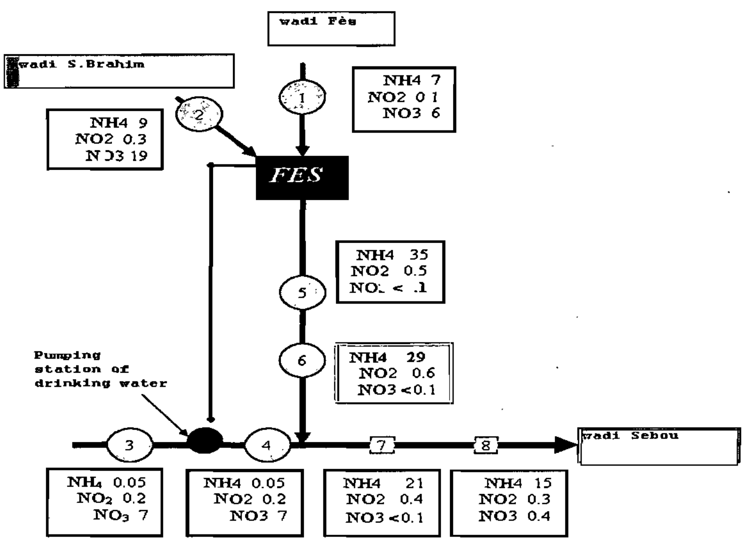

Fig. 5 Concentrations of the nitrogenized species (NH_4, NO_2, NO_3 *in mg/L*) in the surface waters of the Sebou River, upstream (1, 2, 3 and 4) and downstream (5, 6, 7, and 8) of Fès.

The main contaminants of these waters are the nitrogenized species and toxic metals, in particular chromium and mercury which greatly exceed the Moroccan standards. The Fès and Meknès systems of the basin of Sébou provide a clear example of surface water pollution in Morocco.

The main anomalies in nitrogen species are located downstream from the big cities and of the villages of the basin. They are caused by domestic and industrial waste dumped directly into the wadis. In certain wadis of the basin, nitrate concentrations (Fig. 5) rarely exceed the Moroccan standard (50 mg/L); e.g. the case of water samples collected on the wadi Fès downstream from the city of Fès and another on the wadi Tiflet downstream from the city of Tiflet. The results of the surface water analyses carried out downstream and upstream from Fès are shown in Fig. 5; downstream of Fès, wadi Fès shows a concentration of ammonia of 35 mg/L and Wadi Sebou has a concentration of 21 mg/L and 15 mg/L.

The results of analyses of chromium, copper, mercury, nickel and zinc of water samples from the Meknes system are represented in the Fig. 6, which shows high concentrations of mercury (4 and 9 ppb) downstream from the city centre;, nickel reached 198 ppb and zinc 575. in the water samples analysed.

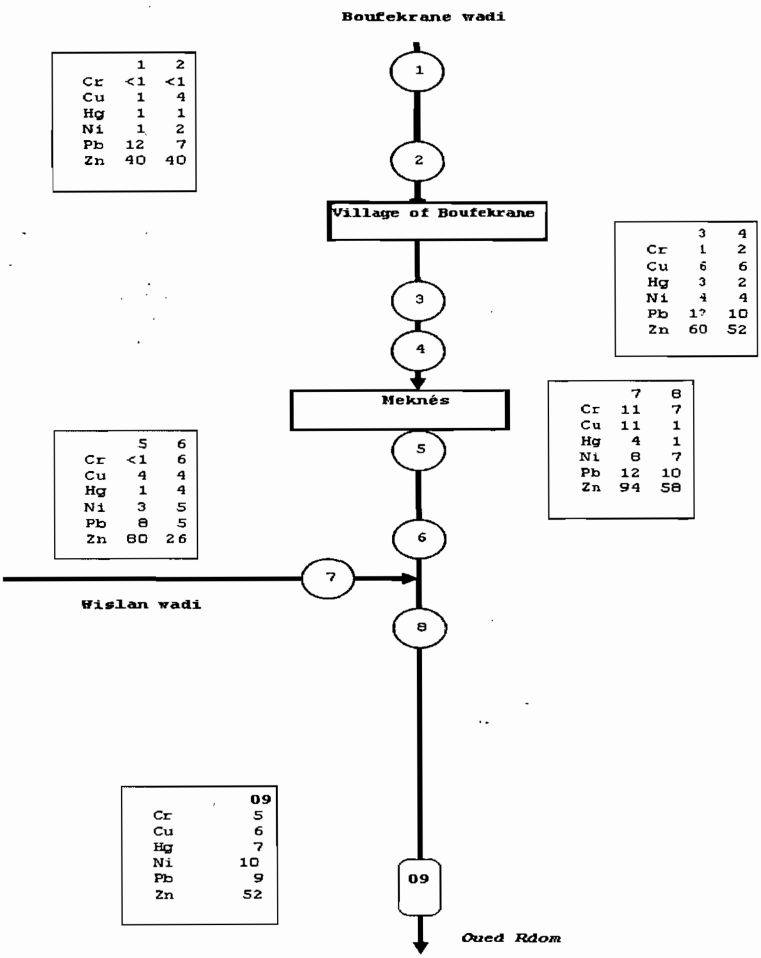

Fig. 6 Concentrations (in ppb) of heavy metals in water of the Meknès system.

CONCLUSION

Our results show that the surface water quality of the Sebou basin, upstream of the wastewater outlets of the urban centres, is good. However, downstream the water quality is a threat to public health. To mitigate this problem, urgent installation of wastewater treatment plants is recommended.

The Sebou River is the main provider of water to the Gharb plain, a major agricultural region where fertilizers and pesticides are widely used. Located along its banks, various industries (paper mills, sugar plants, tanneries, food industries, wool mills, chemical plants, etc.) contribute to the development of large towns which are not equipped with facilities for the treatment of either industrial or domestic waters. It is a real example of water pollution in a developing country.

Currently, the Sebou basin is threatened by intensive pollution, considering the enormous volume of dismissals issued from the city of Fès. The total pollution from this city increases to 100 000 m^3/day; it will reach 120 000 m^3/day by 2015 (RADEEF, 2007).

The main areas of pollution in the basin of Sebou wadi are located at the confluence of the Fès wadi and Sebou wadi downstream of Fès city, where significant contamination of water by all kinds of organic waste, such as household waste, that contribute to increased levels of: the reduced nitrogenous species, the phosphates and the number of the coliform groups, faecal and total.

The content of toxic heavy metals in the water was mainly high downstream of Fès, Meknes and the small town of Moulay Driss (nickel reached 198 ppb and zinc 575 in the water samples analysed). This type of contamination is the result of the artisan industrial activity (tanneries), which characterize these historical cities.

The surface waters of the Sebou basin have deteriorated because of anthropogenic activity and the absence of treatment plants, and this will undoubtedly cause deterioration of the surrounding environment. The major problem with the water resources of Morocco is that they are irregularly distributed in both space and time. The long episodes of drought combined with increasing water requirements generate imbalances in both quantity and quality in the majority of the aquifers.

The recent results obtained by this project have consolidated the relationships between the partners, and encouraged extension of the themes of study to Remote Sensing and Environmental Geology and to the cooperation with other Maghreb Countries (some common activities have already been conducted in Tunisia). The problems faced during the collaboration are the basis of an INCO project presented to the UO Commission on the Sixth Framework Programme FP6.

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Water resources challenges in Nigeria: pathways to water security and sustainability

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Abstract The challenges to water resources in Nigeria of scarcity or shortages, flooding and poor quality, have become very complex, especially in the rural areas. These challenges are triggered by rapid population growth, changes in climate (especially in rainfall patterns) and pollution of different sorts with the end effects borne mostly by the rural people. The study on the rural communities of the southwestern parts of Anambra State of Nigeria illustrates the need for integrated management of water resources through advanced data collection and monitoring techniques, as well as local participation because the food security, health and well being of the people are at great risk.

Key words changes; water protection; data; Nigeria

INTRODUCTION

Water is a basic need in man's environment and the United Nations interest in water as a 21st century global resource lies in the fact that it is central to agriculture management, ecosystem services, sanitation, human health and natural disasters, and hence a key factor to meeting the millennium development goals (MDGs). The complexity of water problems impact sustainable development and the quality of life, threatens food security, human health and natural ecosystems (Luijten, 1999) especially at the local level. The challenges of recurrent water shortages; rain harvesting and water contamination through receptacles and reservoirs, especially when not properly protected from dust and other pollutants; water crises resulting during dry periods when stored rainwater is exhausted; the seasonality of most rural water sources of rivers and streams due to changes in rainfall patterns; the pollution of unprotected water consumption sources by activities of bathing, washing, food processing, refuse dumping as well as the economic activities of fishing, sand dredging, brick making and worship, all result in changes in physical (colour, taste and odour) as well as in chemical and micro-biological properties, thereby deteriorating the quality of water (Field Survey, 2006). To meet the challenges of the ever-changing water sector in Nigeria, the June 2003 initiative "*Water for people, Water for life*", a programme aimed at achieving 100% water supply coverage for all State capitals in Nigeria by the year 2007 was initiated, but unfortunately was not achieved. Nigeria is also presently off-track from achieving similar millennium development goals (MDGs) by the year 2015.

STUDY AREA

The Ihiala and Ekwusigo Local Government Areas and their rural communities in Anambra State, Nigeria, were selected because of the poor state, insecurity and unsustainability in the management of their water resources. The local population list in Table 1 shows the population for each of the study communities. The climate is tropical with distinct wet and dry seasons. The rainy season occurs from the month of April to October with heaviest rainfall in the month of July, making rainwater harvesting feasible (considering the number and distribution of rain days) while the dry season is from November to early March. The natural vegetation lies within the tropical rain forest belt of southwestern Nigeria. The people are mostly farmers, craftsmen and industrialists.

Table 1 Study LGAs and their selected rural communities.

Ihiala LGA communities	1996 Local population	Ekwusigo LGA communities	1996 Local population
Azzia	11 384	Oraifite	29 649
Amorka	7 371	Ichi	10 329
Isseke	9 383	Ihembosi	9 610
Mbosi	8 339		
Okija	44 280		
Orsumoghu	10 167		

Source: National Population Commission; 1996 projection figures.

METHODOLOGY

The concept of water supply protection (BAHC, 2007) was adopted so as to achieve protection of public potable water supply sources through adequate knowledge of the natural and human components of water chemistry and pollution through methods of water sampling and field exercises. The inadequate protection of most water sources of rivers, streams and springs, as well as man-made water sources such as hand dug wells and boreholes, from indiscriminate human activities, pollutants and contaminants, has resulted in water insecurity, water-related diseases and in most cases, deaths. This creates a need for water quality monitoring and evaluation, especially at the rural scale as an integrated management technique for water resources sustainability.

Water quality data generation through water sampling and laboratory testing was carried out in the selected rural communities and analysed using the World Health Organization (WHO) and the Nigerian Federal Ministry of Environment (FMEnv) standard limits. The water sampling exercise was carried out during the dry season or low flow period due to the relatively high concentrations and easy access to remote communities at that time of year. The water sample collection points are as detailed in Table 2.

Table 2 Geographical co-ordinates of water sample points.

Name	Date	Location	LGA	X Coordinates			Y Coordinates		
Ekulo Ibollo stream	8/12/2006	Oraifite	Ekwusigo	6	0	35	6	49	41
Private, free borehole	8/12/2006	Ihembosi	Ekwusigo	5	55	44	6	51	40
Private free borehole	8/12/2006	Ichi	Ekwusigo	6	02	52	6	52	0
Urasi Okija stream	7/12/2006	Okija	Ihiala	5	55	16	6	51	30
Omaiyi stream	7/12/2006	Azzia	Ihiala	5	53	10	6	52	38
Private, free and time regulated borehole	7/12/2006	Orsumoghu	Ihiala	5	52	5	6	54	52
Commercial borehole	7/12/2006	Amorka	Ihiala	5	45	12	6	52	48
Roof catch rain harvest	7/12/2006	Mbosi	Ihiala	5	51	35	6	54	25
Private commercial borehole	7/12/2006	Isseke	Ihiala	5	50	2	6	54	56

Source: Field Survey (2006).

RESULTS

The *Ekwusigo LGA* samples (Table 3) were very acidic, except for the Ekulo Ibollo stream, Oraifite with pH 6.64; but presence of brownish deposits, high turbidity value of 27.8 NTU and iron concentration (2.43 mg/L). Ichi borehole sample had a low iron (Fe) level of 0.25 mg/L.

Samples from *Ihiala LGA* were all acidic and had iron (Fe) levels below the 0.3 mg/L WHO limit. However, samples mostly from free private boreholes and streams had very acidic pH, while most roof-catch rainwater harvest samples had alkaline pH values. This indicates a tendency that health risks could result from intake of water with these parameters.

Table 3 Chemistry of sampled water sources in Ihiala and Ekwusigo LGAs of Anambra State.

LGA/ towns	Sample point	GPS Long./Lat.	Appearance	pH	Turbidity (NTU)	Nitrate (mg/L)	Ammonia (mg/L NH_3)	Hardness (mg/L $CaCO_3$)	Total iron (mg/L Fe)	Salinity (mg/L Cl^-)
Method				ASTM D1293	ASTM D1889	APHA Std Mtd 4500 - NO_3^- E	ASTM D1426	ASTM D1126	APHA Std. Mtd. 3500 – Fe D	ASTM D512-89
WHO Limit			Clear and Colourless	6.5–8.5	5	50	1.4		0.3	250
Ekwusigo										
Ihembosi	Private, free borehole	5°55'44' 6°51'40'	Clear and Colourless	4.53	0.13	5.28	0.45	3	0.08	2.5
Oraifite	Ekulo Ibollo Stream	6°00'35' 6°49'41'	Brownish with small brownish deposits	6.64	27.8	7.48	0.96	21	2.43	1.8
Ichi	Private free borehole for public use	6°02'52' 6°52'40'	Clear and Colourless	4.49	0.48	5.72	0.00	2	0.25	1.0
Ihiala										
Okija	Urasi Okija River	5°55'16' 6°51'30'	Colourless with few brown deposits	5.29	0.90	7.92	0.26	2	0.20	1.9
Azzia	Omaiyi stream	5°53'10' 6°52'38'	Colourless with some brownish deposits	4.96	2.75	3.96	0.15	2	0.22	1.4
Orsumoghu	Private bore-hole free and time regulated	5°52'05' 6°54'52'	Clear and Colourless	4.51	0.59	7.04	0.40	2	0.13	1.4
Amorka	Commercial Borehole 25litres @ 5 NAIRA	5°45'12' 6°52'48'	Clear and Colourless	4.58	0.27	8.8	0.37	4	0.18	NA
Mbosi	Roof Catch	5°51'35' 6°54'25'	Clear and Colourless	5.56	0.32	7.48	0.00	NA	NA	8.3
Isseke	Private, commercial borehole	5°50'02' 6°54'56'	Clear and Colourless	4.91	0.36	7.48	0.39	2	0.09	NA

Source: Field Survey, 2006. NA = not analysed due to insufficient sample; --- = no sample collected; salinity was measured as chloride; ASTM = American Society for Testing and Materials; APHA = American Public Health Association; D = various test methods for particular tests; WHO Limit = World Health Organization Limit for potable (drinking) water.

The oxygen test results (Table 4) on dissolved oxygen (DO), biological oxygen demand (BOD_5) and chemical oxygen demand (COD) for pollution analysis carried out in the laboratory show that the DO levels for all samples were lower than the 9.0 ppm oxygen saturation for freshwater at 20°C and the 7.5 ppm FMEnv limit. The Ekwusigo and Ihiala DO values of 5.5 and 4.9, respectively, show that samples are not really deficient in oxygen. However, the high BOD_5 and corresponding COD values of 75.0 and 184, respectively, for Ihiala suggest that the samples contain high oxidizable organic matter components and, thus, are likely pollution sources.

Pollution deteriorates water quality, rendering it unfit for its intended uses, with traditional risks of infectious and parasitic diseases as resultant effects. Gonzalez & Sherer (2004) emphasized that pollution and agricultural wastes, water contaminants by foreign matter such as micro-organisms, chemical runoff, and industrial or other wastes, or raw sewage, affect large volumes of water supply every year, and worse still, causes sickness and death in those with no choice but to drink tainted water that is unsafe. Therefore, contaminated water could be the source of large outbreaks of water diseases such as cholera, dysentery, typhoid fever, diarrhoea, hepatitis and cryptosporidiosis. The most probable number (MPN) and the membrane filtrate (MF) methods were used to determine the total coliform, faecal coliform (*Escherichia coli*), and the aerobic mesophilic bacteria and fungal counts of the water samples.

A very high total bacterial count for Ekwusigo was recorded (Table 4), 102 ctu/mL indicating high presence of total mesophilic bacteria and disease-causing organisms was recorded. Ihiala LGA (16 ctu/mL) is slightly above the WHO limit of 10 ctu/mL and is fairly acceptable. The coliform counts for both samples indicate the presence of coliforms.

Table 4 Microbial count values in Ihiala and Ekwusigo LGAs; source, Field Survey (2007).

Laboratory analysis	Sample names and locations		
Oxygen tests	FMEnv limits	Osemeham community borehole Akwukwu, Ekwusigo LGA	Omaiyi stream Azzia, Ihiala LGA
DO	7.5	5.5	4.9
BOD_5	0.0	30.0	75.0
COD	N/A	56.0	184.0
Microbiol tests	WHO limit		
Total (mesophilic bacterial count (ctu/mL)	10	102	16
Coliform count (ctu/mol)	-	6	1
Faecal coliform count (cfu/mol)	0	1	0
Micro-organisms isolated	-	Proteus sp., *Enterobacter* sp., *Seriatia* sp.	Chromogenic bacteria, *Citrobacter* sp., *Staphylococcus* sp., *Aspergillus* sp., *Cladosporium* sp.

PATHWAYS TO SECURITY AND SUSTAINABILITY

The challenges of changes and fluctuations in water supply, increased water demand due to high population growth, lack of integrated policies and financial constraints are all set-backs to the Nigerian water sector. This creates the need for security and sustainability through adoption of traditional and scientific knowledge for water solutions in the rural areas. The strategy will support indigenous, private, non-governmental and government organisations' participation in the rural water sector, thereby improving enhanced water harvesting, conservation, protection and even periodic training on rural surface water sources to meet qualitative and quantitative criteria. Reinforcement of environmental conservation practices in the study area through public education and enlightenment programmes will improve the knowledge of geophysical environmental conditions. This will equally regulate indiscriminate activities that pollute water sources, and improve the protection and treatment for threatened water sources against eutrophication, siltation and human contamination, while ensuring the continued existence of the rural water sources for domestic consumption and health safety.

CONCLUSION

The research adopted the concept of water sources protection and data generation through field studies, and water sampling and analysis in the Ihiala and Ekwusigo LGAs of Anambra State, Nigeria, to show the need for water sustainability and security. The global positioning system of sample points as well the laboratory analysis of water samples using standard limits are integrated methods of data collection to show the location, water source component and its pollution parameters. The study area therefore requires adequate monitoring and timely evaluation for sustenance rural water.

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Integrated water resources management of the Idemili River and Odo River drainage basins, Nigeria

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Abstract Adequate water supply is a pre-requisite for the sustainable development of any economy. In many parts of Nigeria, such as Anambra State, functioning water supplies are totally lacking leaving the population in water scarcity. Poor waste disposal/management exacerbates water management issues resulting in the contamination of available water supply sources. Socio-economic consequences of poor water management also abound, increasing poverty and affecting women and children most. Field and laboratory studies aimed at assessing the sustainable integrated water resources management of the Idemili River and Odo River drainage basins in southeastern, Nigeria, were carried out for sustainable water development of the area. The water table occurs at depths of 7 to 108 m while the calculated aquifer parameters of transmissivity and hydraulic conductivity range between 0.48 and 19.50 m^2/day and 0.06 to 3.75 m/day, respectively. The pH of the water resources of the basins is 5.6–7.0 indicating slight acidity. Major ions such as Mg^{2+}, Ca^{2+}, Cl^-, NO_3^- and SO_4^{2-} occur at concentrations within the WHO recommended permissible limits. High concentrations of Fe^{2+} ranging from 0.01 to 0.9 mg/L were observed in the groundwater of the basins and were attributed to the inherent geology of the area. The areas of high head in the aquifer coincided with areas of groundwater recharge and intensive gully erosion/landslide development associated with the dominant Awka-Umuchu-Orlu escarpment. These areas are recommended for water catchment structures for rainwater harvesting, artificial recharge and gully/landslide control measures. Sustainable water projects for the domestic water supply to the rural poor are recommended based on the quantity/quality of the water resources of the basins.

Key words Nigeria; water quality; geophysical methods; hydrogeochemistry

INTRODUCTION

Water resources management is essential for the sustainable development of any economy. The daily water need of a population cannot be substituted given that water is life and should be treated as such in both supply and management. Lack of a functional water scheme in parts of southeastern Nigeria is common, especially in Anambra State, exacerbating development. Rainwater is wasted and allowed to cause environmental disasters, pollution and erosion through flooding. Poor environmental management creates havoc on the water supply, hygiene and sanitation of the area, exacerbating public health. Water for food security of the unprecedented populations of many developing countries has yet to receive successful attention. Groundwater resources in developing economies are exceptionally important as a source of relatively low-cost and generally high-quality municipal and domestic water supply (GW-MATE, 2008). The implications of climate change result in high dependency on groundwater resources without adequate plans for sustainable supplies. The basins of study are located in areas vulnerable to severe land degradation in the form of gully erosion and landslides that threaten, among other things, effective integrated water resources management. Great portions of the socio-economic group of poor nations, such as Nigeria, dwell in areas affected by water access problems. The management of water resources, including their protection from pollution, can be facilitated through long-term groundwater planning in order to ensure informed and participatory decision-making (Waltina & Elke, 2008).

The resources of the Idemili River and Odo River drainage basins are still underdeveloped and under used. The integrated management of the water resources of the basins will enhance the sustainable water situation, provide opportunities for economic development and social improvement. It will also advance water and food security of the region by employing best agricultural practices in the valuable flood plain of the basin.

LOCATION AND METHOD OF STUDY

The study area lies within latitudes 6°00'N and 6°18'N and longitudes 6°45'E and 7°20'E, covering an area of about 1706 km^2 within the Anambra Basin (Fig. 1). The area falls within the rainforest belt of West Africa. Two climatic conditions exist in Nigeria and the study area: the rainy season and the dry season. While the rainy season in characterized by heavy thunderstorms that last from April to October, the dry season extends from November to March, with high temperatures and a dusty atmosphere. The mean annual rainfall is about 2000 mm (Floyd, 1965). The high rainfall intensity and the concomitant large volumes of runoff are a significant environmental issue.

Vertical Electrical Sounding (VES) using the Schlumberger electrode configuration was carried out at various locations in the study area. The acquired apparent resistivity data were inverted and interpreted for different geoelectric units using the RESIST automated inversion software. The geoelectric units obtained were correlated with available lithologic logs of drilled sites (Fig. 2). Samples from both the surface water and the groundwater resources were analysed for physical and chemical properties to determine the potential of the basins integrated resources management.

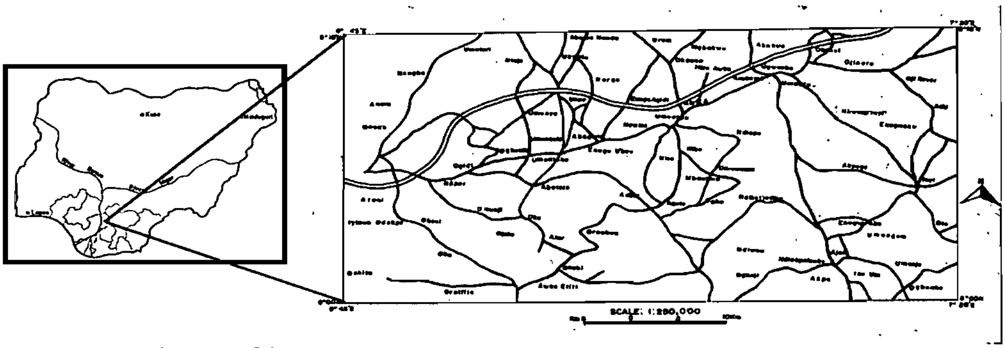

Fig. 1 Location map of the area.

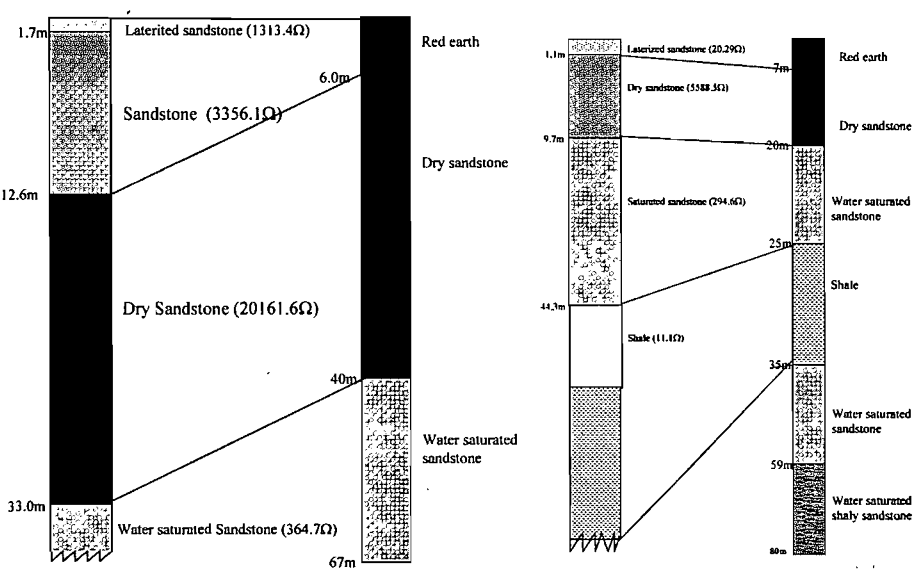

Fig. 2 Correlation of adjacent borehole with VES result.

Geology and hydrogeology

The study area falls within Anambra Basin. The basin stratigraphy is largely a result of the palaeohydrology of the region (Druhan *et al.*, 2008). The basin is associated with the Santonian tectonism that resulted in the folding and uplift of the Abakaliki–Benue trough and the subsidence of the Anambra platform to form a major depocentre for clastic materials (Reyment, 1965). The main geological units of the study area are the Nanka Sands (Eocene), which are overlain by Ogwashi-Asaba Formation (Oligocene) and underlain by Imo Shale (Palaeocene), Fig. 3(a).

Lithologically, the Nanka Sands consist of distinct units of sands, shale-siltstone and finely laminated shale. Sand subunits comprise uncemented, medium to coarse grained and pebbly quartz sand, with thickness varying from 50 to 90 m (Nwajide & Hoque, 1979). The Nanka Sands form the north–south trending Awka-Umuchu-Orlu cuesta, the main topographic high in the basin. The sandy units of the formation form a thick viable aquifer (Egboka *et al.*, 1985). The Imo Shale, which resulted from the major transgression of the Palaeocene, consists of dark-grey to bluish-grey shales, siltstone, mudstone and sandstone lenses. The main sandy facies is the Ebenebe Sandstone. The shale is highly fractured and fissile and lies east of the Nanka Sands. The area is drained by numerous surface waters in the form of rivers, streams and lakes (Fig. 3(b)). While Idemili River and its tributaries drain the east of the study area, the Odo River drains the western part, establishing both a surface water and a groundwater divide.

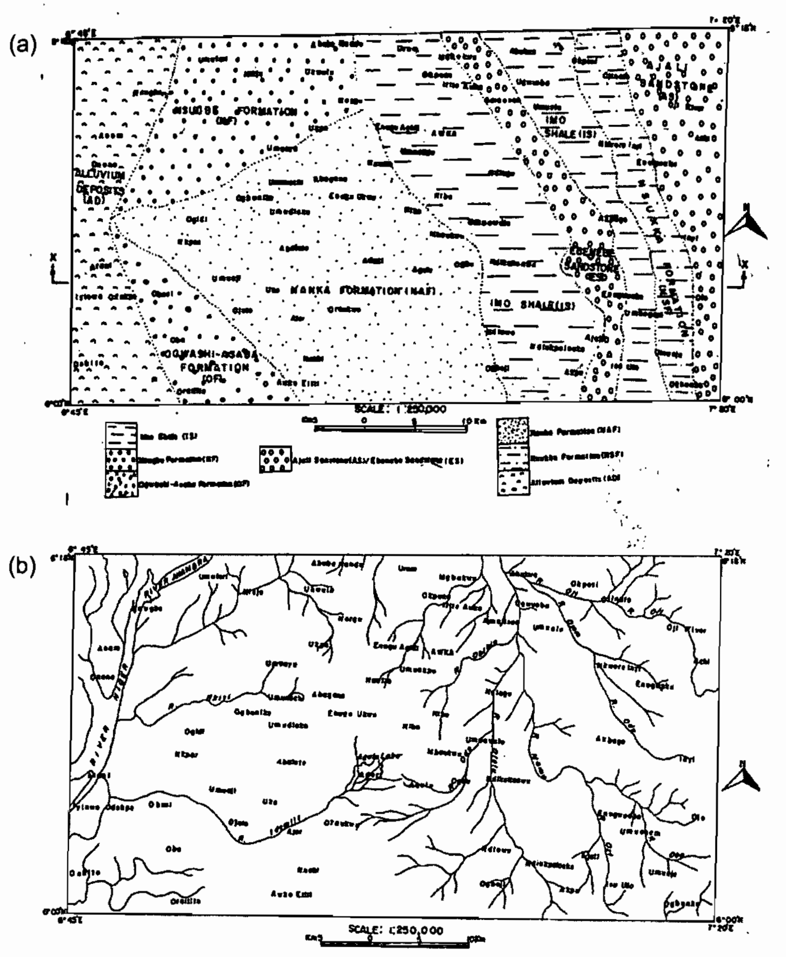

Fig. 3 (a) Geology map of the basins. (b) Drainage map of the basins.

The knowledge of local geological and hydrogeological conditions, as well as the understanding of the groundwater flow regime, can contribute significantly in solving regional problems. Hydraulic conductivity and aquifer depth are fundamental properties that describe the subsurface hydrology. In the present study, vertical electrical sounding using a Schlumberger Array was employed in the porous media of the Nanka Formation to determine the groundwater potentials of the basins. In a porous media such as the study area, hydrogeological properties of the aquifers do not generally vary rapidly. As a result, a direct linear relation between resistivity and hydraulic parameters (conductivity, K, and transmissivity, T) is expected to exist. The high cost of pumping tests in aquifer hydraulic study cannot be underestimated. Thus K and T were estimated from geophysical studies in combination with some data from pumping tests. The Nanka Sands aquifer has high permeability with a plasticity index of 12.50–36.57% (Ifeoma, 2008) thereby increasing its porosity. The aquifer is recharged locally by precipitation. Numerous seepage faces at the base of many gully sites that characterize the area, coupled with the few rivers and streams, form the outlet for the discharge of the aquifer. The depth of the wells tapping the upper unconfined aquifer ranges from 7 to 108 m, with very shallow wells located close to the river sources.

RESULTS AND DISCUSSION

Of the 40 VES measurements carried out for this research, only eight locations have information from previous pumping tests of existing boreholes useable for estimation of hydraulic parameters. Resistivity readings are processed to produce sections of thickness and resistivity of subsurface electrical layers (Corriols *et al.*, 2007). The transmissivity Tc values calculated from the VES results range from 0.48 to 19.50 m^2/day while the equivalent values obtained from pumping tests vary from 1.07 to 33.74 m^2/day. The hydraulic conductivity Kc values estimated from the VES results range from 0.06 to 3.75 m/day. These values compared favourably with those obtained from pumping test analyses in the area.

The water samples were analysed at Ideyi Consult Limited, University of Port Harcourt, Nigeria. The detailed chemical analysis of the water resources of the basins is presented in Table 1. The total dissolved solids ranges from 16–151 mg/L. The presence of this parameter in high concentration is often an unrecognised problem in rural water supplies. The values obtained are well below WHO (2004) standards. The pH of the water samples is 5.68–7.4 indicating slight acidity and may be attributed to environmental activities. The value of the total hardness (TH) is

Table 1 Physiochemical parameters and WHO Standard.

Location	Source	pH	Total Hardness	TDS mg/L	HCO_3- mg/L	SO_4^{2-} mg/L	Cl^- mg/L	NO_3^- mg/L	Mg^{2+} mg/L	Ca^{2+} mg/L	Fe^{2+} mg/L
Agulu	Gw1	6.31	18	28	18.30	<1	4.25	4.95	0.93	5.20	0.02
Ukpo	Gw2	6.30	17	28	14,95	<1	7.09	9.43	1.02	5.07	<0.01
Oko	Gw3	6.16	28	38	19.52	<1	7.09	11.67	1.44	7.36	<0.01
Awkuzu	Gw4	7.4	13	22.3	8.0	5.0	4.2	0	5.0	8.0	0.003
Nise	Gw5	7.0	70	130	-	21.6	6.0	23.9	38.9	20.0	0.3
Awka	Gw6	5.6	18.5	63	-	4.0	5	0.06	9.7	2.4	0.5
Nteje	Gw7	6.2	65	151	25	21	16.4	0	19.2	66.8	0.7
Ufuma	Gw8	6.0	48	55.4	1.6	5	10	8.8	12	36	0.05
Idemili River	Sw1	6.58	10	16	10.98	2	5.90	0.47	0.66	0.83	0.09
Odo River	Sw2	5.86	21	43	3.14	20	3.78	3.06	2.45	4.31	0.03
Agulu Lake	Sw3	6.40	10	27	-	-	5.70	0.8	5.0	5.0	0.29
Mean		6.34	28.95	54.7	12.36	7.97	6.86	6.31	8.75	14.63	0.22
WHO	HDL	6.5	100	500		200	200	50	50	75	0.03
(2004)	MPL	8.5	500	1500		400	500	70	150	200	1.0

10–65 mg/L. These values are within the range classified by Sawyer & McCarty (1967) as soft water. From Table 1, the mean concentration of the cations is in the order: Ca^{2+} >Mg^{2+} >Fe^{2+} while the anions is in the order of HCO_3> SO_4^{2-} > Cl> NO_3^-. The values of the parameters are generally well within the WHO standard for drinking water quality. The low concentration of NO_3^- in surface water bodies is an indication of low agricultural activities and under-utilization of the resources of the basins, while the relatively high value obtained in sample GW5 may be attributed to sewage contamination due to unplanned housing development. The concentration of Fe^{2+} ranges from 0.01 to 0.9 mg/L in the sampled water supply sources. About 80% of the samples have concentrations above the WHO acceptable limit of 0.3 mg/L. The high concentration of this cation is associated with the inherent geology of the basin.

Relatively strong correlation was obtained between TDS and the major ions (Table 2). The values of TDS relates well to SO_4^{2-}, Mg^{2+}, Ca^{2+}, Cl^- and Fe^{2+} with coefficients of determination R^2 = 0.79, 0.87, 0.80, 0.67, and 0.76, respectively. The relatively strong correlation between TDS, SO_4^2 and Ca^{2+} indicate that both ions are derived from the same source. The scatter plot of some positively correlated parameters are shown in Fig. 4.

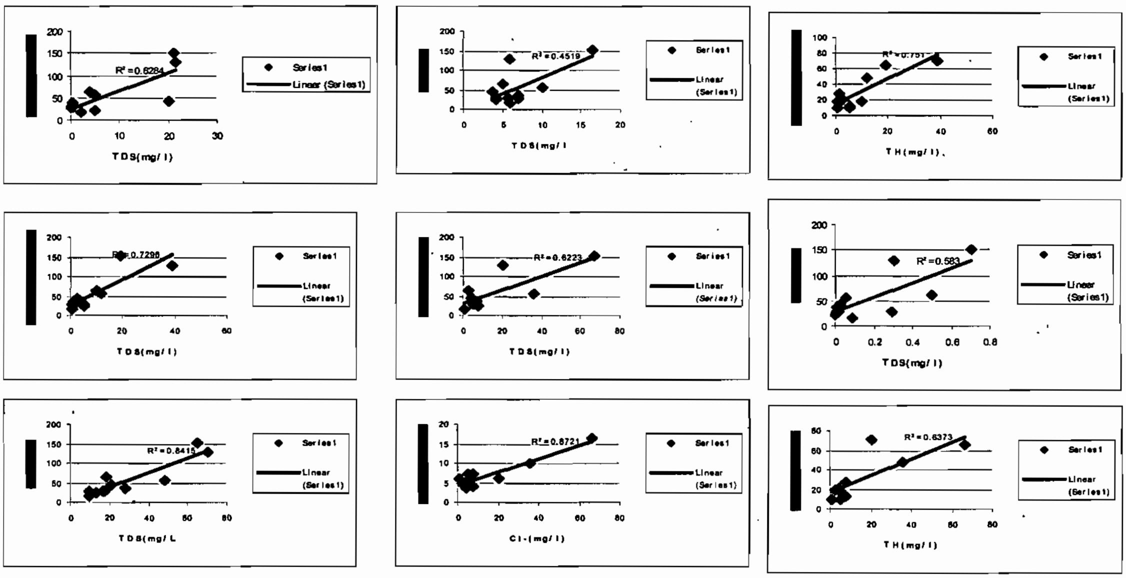

Fig. 4 Correlation analysis of the parameters.

Table 2 Correlation coefficient matrix of the parameters.

	pH	TH	TDS	HCO_3^-	SO_4^{2-}	Cl^-	NO_3^-	Mg^{2+}	Ca^{2+}	Fe^{2+}
pH	1.00									
TH	0.05	1.00								
TDS	−0.02	0.92	1.00							
HCO_3^-	0.01	0.13	0.16	1.00						
SO_4^{2-}	0.05	0.72	0.79	−0.02	1.00					
Cl^-	−0.18	0.66	0.67	0.50	0.35	1.00				
NO_3^-	0.57	0.54	0.30	−0.18	0.26	−0.04	1.00			
Mg^{2+}	0.26	0.87	0.85	−0.22	0.70	0.34	0.62	1.00		
Ca^{2+}	−0.59	0.80	0.79	0.40	0.55	0.93	0.03	0.52	1.00	
Fe^{2+}	−0.27	0.50	0.76	0.24	0.46	0.59	−0.18	0.54	0.60	1.00

TH = Total Hardness. All concentrations are in mg/l.

CONCLUSION

Water projects in most developing countries are often confronted with implementation barriers resulting in non-completion of some projects. Confidence in the success of proposed innovative technology for groundwater management is often low, and therefore conventional and more expensive technologies are often preferred (Prokop, 2003).

A proper understanding of the implications of access to water supply in the development of any economy can serve as a decision tool for resource managers. Efficient integrated management of valuable basin resources is essential for improved health standards and food security. This will ensure basin water resources protection in terms of quality and quantity to avoid water scarcity.

Information on hydrogeochemistry was employed in the characterization of the water resources of the Idemili and Odo River drainage basins for the integrated water management of the basins. The parameters analysed for water quality mainly fall within WHO acceptable standards. The depth to water table is very shallow and easily exploitable going by the geoelectric units of the geological formation. The results of the study show that the basins' water resources are moderately high in quantity and quality and can be exploited for domestic water projects. In view of that, it is recommended that integrated management of the water resources of the basins for the development of sustainable water schemes, especially for the rural dwellers, may be carried out. The exploration, exploitation and management of the huge water resources potential of the basins requires an integrated and systematic cost effective approach for efficient water supply to urban and rural areas for domestic, recreation and agricultural purposes. This will also enhance the socio-economic development of the country, especially as regards rural development and poverty alleviation. The outcome of the study is expected to provide an appropriate decision-support tool for sustainable development and a rational exploitation of the resources of the basins to achieve the millennium development goals.

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Hydrological characterization and groundwater resource studies in coastal areas: Sagar Island region, West Bengal, India

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Abstract Sagar Island is the largest island in the Ganga delta and is built up of unconsolidated alluvial sediments of Quaternary age. For water supply, villagers are totally dependent on the sweet groundwater tapped from deep confined aquifers through tube wells of depths more than 240 m. The deeper fresh aquifers lie between 180 m to 335 m below ground level (bgl). Surface water or dug well water is saline. The overlying shallow aquifer is also saline water bearing. The present study comprises an integrated geological, geoelectric and geochemical investigation to assess the prevailing surface and groundwater condition, viz. aquifer depth, chemical quality of groundwater and hydrological characteristics in parts of Sagar Island for groundwater resource studies. Vertical electric soundings (VES) were carried out with maximum electrode spacing of 1200 m and the resistivity layers parameters obtained from VES studies threw light on the facies change in the subsurface lithology and reveal the existence of a saline water bearing zone overlying a fresh-water bearing zone. The combination of the VES data with the borehole data provides useful information on subsurface hydrogeological conditions. The results show the presence of six prominent layers consisting of alluvial top soil, saline water, brackish water, impermeable clay layer, freshwater and lowermost clay with silt and sand lenses under the prevailing hydrodynamic condition. Such a fresh confined aquifer is typical of developed areas with overlying clay-rich silty formations which prohibit the infiltration of saline and brackish water. The average thickness of the freshwater bearing zone under confined condition is about 184 m at an average depth of about 178 m from the surface. A litho-resistivity relationship is also established for this area. Chemically the fresh groundwater is Na-HCO_3 type with TDS ranging from 495 to 740 mg/L. The groundwater is safe for drinking and domestic purposes with low to medium sodium adsorption ratio (SAR) values. The Na content is relatively higher than that of other elements present in the groundwater. The seawater contamination (SWC) values for these water samples are significantly low.

Key words vertical electric sounding; sodium adsorption ratio; seawater contamination; estuarine process; aquifer

INTRODUCTION

Sagar Island, the largest island in the Ganga Delta, is built up of unconsolidated sediments of Quaternary age. Across the 235 km^2 Island there are 46 villages with a total population of about 0.3 million. The reclamation of land for settlement and cultivation by deforestation has accelerated land erosion, especially in the east, west and southern sectors of the Island.

For drinking purposes the villagers are totally dependent on the fresh groundwater tapped from the confined aquifer through deep tube wells (depth more than 240 m). There are no dug wells as the overlying aquifers are saline/brackish. Peoples in the coastal areas are facing problems with the availability of fresh groundwaters. Geological, geoelectrical, geochemical and isotopic studies have been carried out in the Sagar Island region to assess the hydrological characteristics, chemical quality and depth to the aquifers containing freshwater.

Geographic location

Sagar Island is bounded by latitudes 21°37′N to 21°52′N and longitudes 88°2′35″E to 88°11′E and covered in Survey of India Toposheet no. 79C/2 (Fig. 1). It is longest in the north–south direction (~30 km) and is of varying width in the east–west direction, the maximum width being in the south (~12 km) of the Island. It is situated in the southern most part of West Bengal in the subdivision of Kakdwip of South 24-Parganas. It is bordered in the north, west and east by the River Hoogly and its two distributaries, the Gabtala and Muriganga rivers. The Bay of Bengal lies to the south.

Geological setting

Sagar Island lies at the southern-most part of the Indo-Gangetic Plain, which is the largest alluvial tract in the world, being mantled by Quaternary sediments carried and deposited by the river

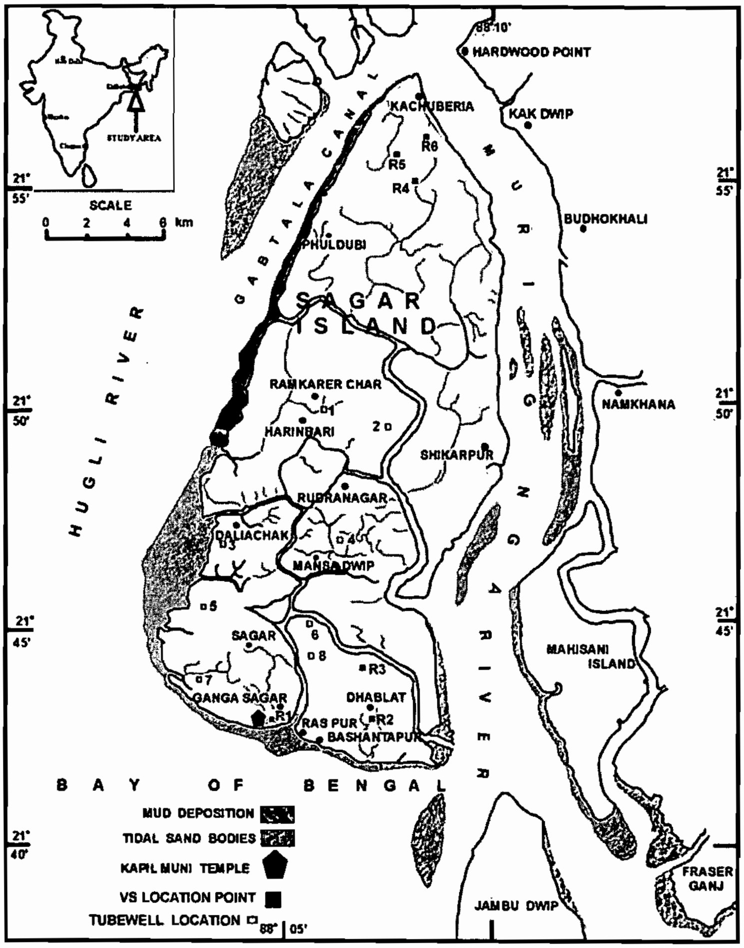

Fig. 1 Important places, VES and tubewell locations and coastal geology of Sagar Island (Deltaic-Western Sundarban).

Ganga and its distributaries. Here Quaternary sediments are underlain by Tertiary sediments (Pleistocene to upper Cretaceous), indicating an accumulation in a subsiding tectonic trough, i.e. all the sediments were deposited under shallow water, but were gradually lowered down due to tectonic movements. The thickness of the sediments varies from about 415 m near the western margin of West Bengal to about 8232 m below Kolkata (Hydrology Atlas-CGWB, 1987). The Quaternary sediments of the Bengal Plain are constituted of flood plain deposits as well as deltaic deposits, and may be subdivided into two major groups (Roychoudhuri, 1974): (a) newer alluvium: Recent to sub-recent; and (b) older alluvium of Pleistocene age.

The older alluviums are essentially flood plain deposits outcropping above the level of the present flood plain deposits. They comprise clay, silt, boulders and gravels which are locally cemented and red in colour. The newer alluvium is mostly confined to present day river beds, banks and associated flood plains being constituted of less cohesive sand, silts and clays with a higher water content and good amount of organic matter.

Sagar Island is criss-crossed by numerous creeks and man-made canals. Thirty-seven creeks cross the Island, subdividing it into a few large and several small sectors, and beside these, numerous man-made canals are also present. The creeks are actually fed by the water during high tides. During high tide the creeks are full of water, but during low tide there is very little water.

Coastal marshes, mangrove swamps, tidal flats, mudflats, sand dunes or ridges, marine terraces, and tidal inlets are all coastal features of this island (Paul & Bandyopadhyay, 1987).

Hydrogeological setting

Analysis of lithologs from several boreholes drilled by the Central Groundwater Board and Public Health Engineering have revealed two distinct groundwater bearing zones, viz. (a) upper aquifers of saline water comprising mostly fine to very fine grained sand layers; and (b) confined lower freshwater bearing aquifers of fine grained sand. The saline water bearing aquifers are separated from the freshwater aquifer by a clay layer of thickness of around 25 m. From the water level contour configuration it has been observed that the hydraulic gradient is low, ranging from 0.0005 to 0.000021 m per metre of travel. Regarding recharge of this fresh groundwater which is confined and at considerable depth, it is unlikely to be replenished by local rainfall, surface water and overlying saline water aquifers, as in the latter case the quality of the deep groundwater would have deteriorated. One possibility of recharge of the aquifers is from the porous and permeable near sediments of Hoogly, Burdwan, Nadia and Murshidabad districts.

GEOELECTRIC STUDIES

Geooelectric methods have become one of the most important tools for delineating upper subsurface layers. Systematic Schlumberger Vertical Electrical Sounding (VES) was carried out in six locations (Fig. 1) in the coastal part of Sagar Island, with a maximum depth of investigation of 1600 m, to ascertain the vertical distribution of the water bearing zones contributing the aquifer bodies. Field VES data were interpreted by the inversion technique using RESIST software (Van der Velpen, 1988). Preliminary values of model parameters obtained by manually matching the VES field curves with theoretical Master curves and Auxiliary point charts were used subsequently as input in the RESIST software for further refinement of the results of the 1-D inversion algorithm. The degree of uncertainty of the computed model parameters and goodness of fit in the curve fitting algorithms are expressed in terms of standard deviation and residual error.

Geoelectric resistivity methods have been extensively used for structural, hydrological, geothermal, contaminated site, water table and aquifer hydrological characteristic studies (Stewart *et al.*, 1983; Yadav & Abolfazli, 1998; Majumdar *et al.*, 2000, 2002; Pal & Majumdar, 2001; Majumdar & Pal, 2005; Majumdar & Das, 2007).

Data interpretation

The resistivity of different layers and corresponding thicknesses are reproduced by a number of iterations until the RESIST model parameters of all VES curves are totally resolved with a minimum RMS error (Fig. 2). The 1-D inversion technique reserves its importance and utility, as the interpreted parameters can serve as starting model for 2-D and 3-D approaches The interpreted resistivity layer parameters for these VES locations are shown in Fig. 3. The top layer is interpreted as alluvial clayey soil. The second and third layers represent saline and brackish water bearing saturated zones. Both layers are constituted of clay with silt and sand lenses. The fourth layer is an impermeable clay with average thickness of 24 m. This impervious layer separates the overlying brackish water zone from the underlying freshwater bearing formation. The fifth layer is the most important in the study area, it is interpreted as the freshwater bearing sandy zone. The fifth layer in the northern (R4, R5, R6) and southern (R2 and R3) parts is resolved due to increased electrode spacings down to 1600 m and its average thickness in Sagar Island is around 180 m with an average resistivity of 59 ohm m. The average depth of the upper surface of this aquifer is 170 m. The lower-most sixth layer is constituted of clay with fine sand lenses. This layer shows low water yield with an average resistivity of 18 ohm m. The borehole data near R1 corresponds significantly with the VES interpretation. The average depth of the upper surface of the sixth layer is at 354 m. Field studies confirm the presence of saline water within a few metres below ground level. A litho-resistivity relation was also established for the Island and is shown in Table 1.

VES No.	ρ_1 (ohm-m)	h_1 (m)	ρ_2 (ohm-m)	h_2 (m)	ρ_3 (ohm-m)	h_3 (m)	ρ_4 (ohm-m)	h_4 (m)	ρ_5 (ohm-m)	h_5 (m)	ρ_6 (ohm-m)	h_6 (m)	RMS error
R1	5.7	3.4	0.8	24.8	6.0	159.3	2.6	24.0	41.0	∞			2.0

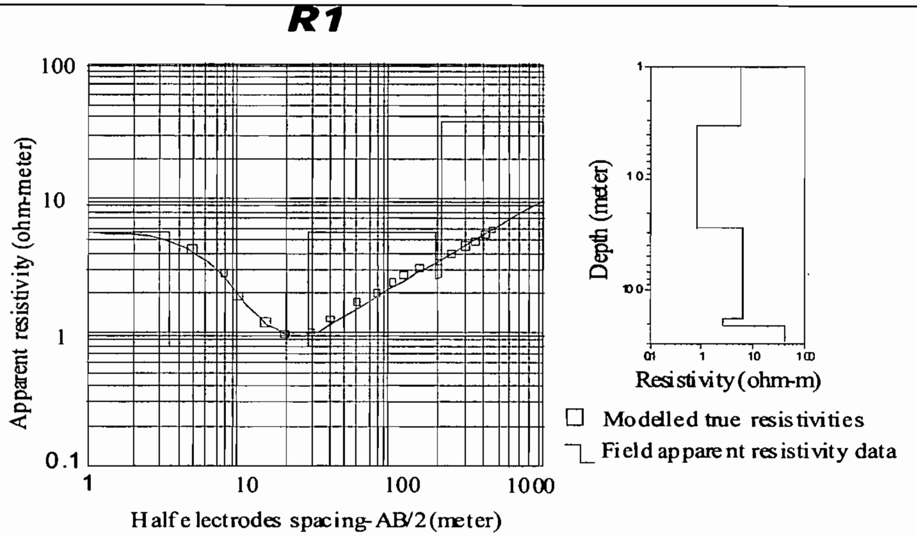

Fig. 2 Typical resistivity field curve with interpreted layers parameters for VES location R1 near the borehole in southern part of Sagar Island..

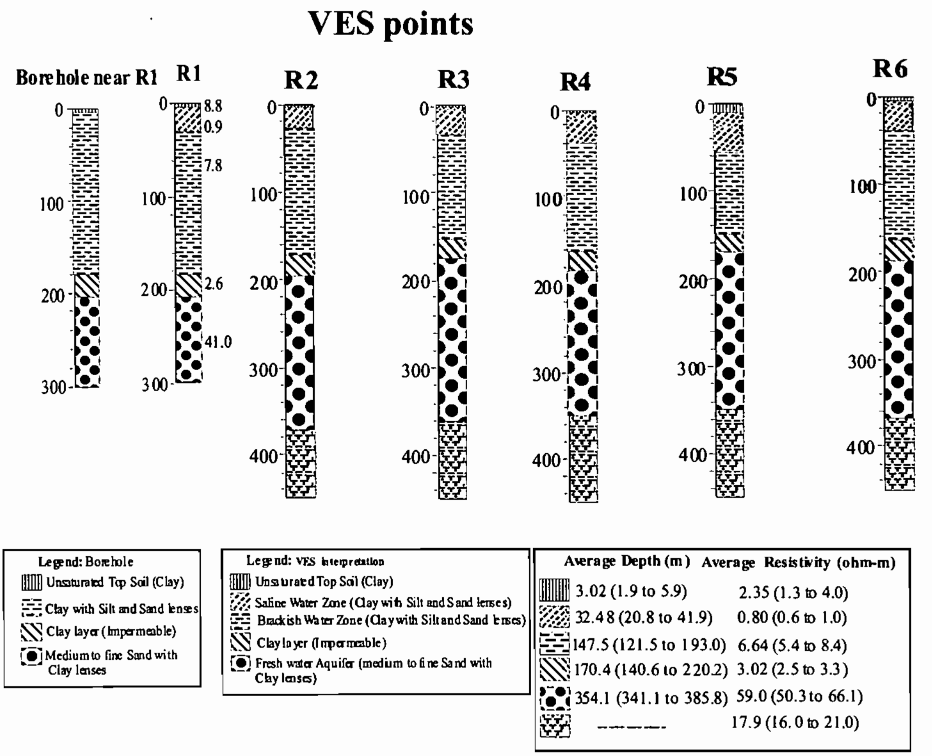

Fig. 3 Combined borehole litholog prescribed by PHED and CGWB near VES R1. Layer parameters are shown in VES interpretations for R1 to R6. Numbers in the left and right-hand side of the logs show depth from ground level (m) and true resistivity values (ohm m), respectively.

Table 1 Litho-resistivity relationships for Sagar Island.

Probable lithology	Resistivity range
Unsaturated top soil (clay)	1.3 to 5. 7 Ωm
Saline water zone (clay with silt and sand lenses)	0.5 to 1.0 Ωm
Brackish water zone (clay with silt and sand lenses)	5.4 to 8.4 Ωm
Clay, grey, sticky layer (impermeable)	2.5 to 5.0 Ωm
Fresh water aquifer (medium to fine sand with clay lenses)	50 to 66.0 Ωm
Clay with fine sand and silt lenses (poor yielding properties)	16.0 to 21.0 Ωm

HYDROGEOCHEMISTRY

A geochemical investigation was carried out to assess the suitability of the water for irrigation and drinking purposes and establish any seawater–groundwater interaction. Eight groundwater samples were collected from tubewells (Fig. 1) from ~180 to ~330 m depth between 14 and 18 March 2006 (three samples) and 16 and 20 September 2006 (five samples). The water samples were collected in pre-cleaned transparent plastic bottles. Before collection of water samples, the bottles were thoroughly rinsed with the same water sample. Concentrations of major cations, anions and some hazardous elements were measured in the chemical laboratory of the Center for Study of Man and Environment, Salt Lake, India.

RESULTS AND DISCUSSION

Table 2 details the analystical results for the water samples. Total dissolved solids (TDS) contents (495–740 mg/L) clearly show that all the samples are freshwater. TDS *vs* electrical conductivity (EC) plots show a linear trend with an equation Y = 1.616 × X – 1.624 (good correlation, R^2 = 0.992). The ratio of TDS and EC is 0.619, which is close to the ratio (0.627) for water from sands of the Gangetic alluvium and Tarai-Bhabar (Chaterji & Karanth, 1963). Main cations and anions in the water samples from the study area have been plotted in a Piper diagram (Fig. 4). All the samples represent the same facies which is Na-HCO_3 type.

Table 2 Chemical composition of groundwater in southern and central parts of Sagar Island.

Sample no.	Electrical conductivity at 25°C (μs/cm)	TDS (mg/L)	Carbonate (as $CaCO_3$, mg/L)	Bicarbonate (as $CaCO_3$, mg/L)	Chloride (Cl, mg/L)	Sulphate (SO_4, mg/L)	Sodium (Na, mg/L)	Potassium (K, mg/L)	Calcium (Ca, mg/L)
1	1020.0	620.0	0.0	350.0	95.0	74.2	180.0	13.0	24.05
2	960.0	590.0	0.0	362.5	65.0	21.0	190.0	18.0	16.03
3	1010.0	610.0	0.0	350.0	100.0	29.0	174.0	16.7	20.04
4	820.0	530.0	0.0	312.5	85.0	28.4	150.0	12.0	20.04
5	1210.0	740.0	0.0	245.0	143.9	110.9	142.5	05.5	40.10
6	900.0	580.0	0.0	280.0	155.0	19.5	198.0	22.0	20.8
7	810	495.0	0.0	336	52.0	3.6	180.0	16.0	9.6
8	950.0	595.0	0.0	410.0	75.0	1.3	180.0	20.0	4.80

Sample no.	Magnesium (Mg, mg/L)	Arsenic (As, mg/L)	Iron (Fe, mg/L)	Lead (Pb, mg/L)	Mercury (Hg, mg/L)	Total hardness (mg/L, $CaCO_3$)	SAR	Soluble sodium %	Residual sodium carbonate	Seawater contamination
1	17.1	<0.01	---	<0.01	<0.001	130.4	6.86	72.70	3.13	0.466
2	14.6	<0.01	---	<0.01	<0.001	100.1	8.26	76.98	3.94	0.308
3	07.3	<0.01	--	<0.01	<0.001	80.7	8.46	78.85	4.14	0.491
4	09.7	<0.01	--	<0.01	<0.001	89.9	6.88	75.55	3.32	0.468
5	22.4	<0.01	0.60	<0.01	<0.001	200.0	4.47	70.31	0.18	0.350
6	12.7	<0.01	0.70	<0.01	<0.001	136.0	8.43	86.77	2.50	0.553
7	07.8	<0.01	0.39	<0.01	<0.001	56.0	10.46	88.03	3.07	0.350
8	13.7	<0.01	0.72	<0.01	<0.001	68.0	9.47	91.53	5.35	0.182

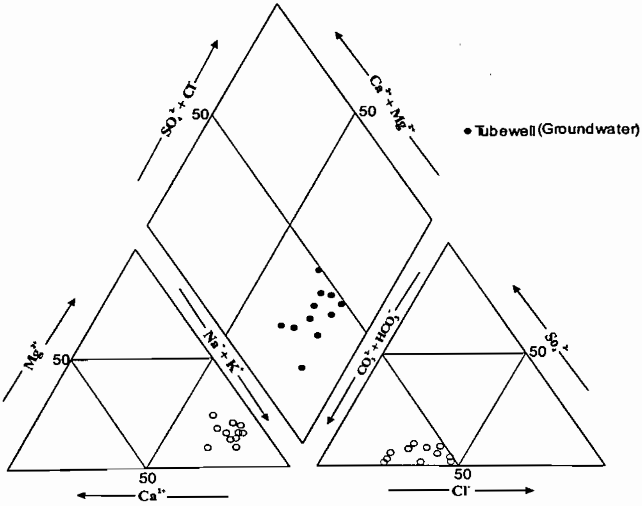

Fig. 4 Piper diagram for groundwater samples.

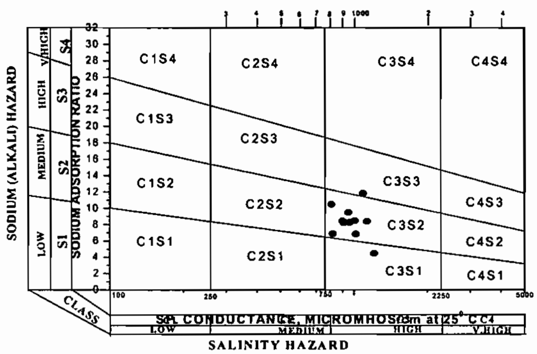

Fig. 5 Richards diagram for classification of irrigation waters (EC *vs* SAR).

Total hardness (2.497 Ca + 4.115 Mg) values indicate the waters are moderately soft (Sawyer & McCarty, 1967) and according to WHO guidelines (2004), suitable for domestic use. The Sodium adsorption ratios (SAR) for all samples were calculated $(Na^{+}/\{(Ca^{++}+Mg^{++})/2\}^{1/2}$, all values are in meq/L) and plotted against EC (Richards, 1954) (Fig. 5) to determine the suitability of the water for irrigation purposes. The plots suggest the groundwater in the southern parts of Sagar Island are moderately suitable for irrigation purpose in terms of the sodium (SAR) and salinity hazard (EC), The soluble sodium percentage $[\{(Na^{+})/(Na^{+}+K^{+}+Ca^{++}+Mg^{++})\} \times 100$, all values are in meq/L] values of 70.31–91.53 were also high, indicating the dominance of Na^{+} in the major cations. Residual sodium carbonate $[(HCO_3^{-}+CO_3^{--})-(Ca^{++}+Mg^{++})$, all values in epm] values of 0.18–4.14 suggest that the groundwater in the study area is suitable for irrigation purposes excepting the area near tubewell locations 2 and 3.

Concentrations of arsenic, iron, lead and mercury in the samples are found to be safe for drinking water purposes. It is important to note that the Na content is higher than that generally found (20 mg/L) in groundwater (WHO, 2004). It ranges from 130–198 mg/L where 200 mg/L is the acceptable limit for the drinking water recommended by the WHO (2004).

CONCLUSIONS

The following conclusions can be drawn from the above integrated studies:

1. Vertical electrical soundings (VES) have delineated the topsoil, the saline/brackish groundwater zones, an impermeable clay layer, freshwater aquifer and clay with fine sand lenses(low groundwater yield) in the subsurface geological formations. The freshwater zone is of appreciable average thickness(184 m)and at an average depth of 170 m.
2. VES findings suggest potable groundwater zones of appreciable thickness, which can be tapped for drinking water purposes. The potential groundwater-bearing zone is under confined conditions. The groundwater conditions of the region as interpreted from the VES study correlate well with the borehole data.
3. A litho-resistivity relationship is established that can be used for estimating lithologies in other unexplored areas under similar hydrodynamic conditions.
4. There is no evidence of seawater mixing with groundwater.
5. Groundwater of this area is Na-HCO_3 type. The chemical quality of groundwater is safe for drinking, domestic purposes, and suitable for irrigation purpose.
6. Concentrations of arsenic, iron, lead and mercury in the samples are below the recommended limit for drinking water of WHO (2004).

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Impact of land-use change on groundwater quality in the Indo-Gangetic Plains, India

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Abstract Rapid growth in population, urbanization, industrialization, and competition for economic development, and the associated land-use/cover change have led to an increased rate of groundwater depletion and degradation. Groundwater samples from Allahabad district, India were analysed for chemical parameters. Analysis of variance of all selected parameters of water quality showed significant differences with sources. Interaction between source and water quality was also significant. The groundwater in different parts of Allahabad district is severely affected and has also become very vulnerable to pollution by a wide range of contaminants. It is interesting to note that higher values of all the parameters were found in the agricultural areas, while lower values were recorded in settlements. This clearly shows the impact of land-use practices on the groundwater.

Key words Allahabad district, India; contamination; groundwater quality; anthropogenic; seasonal variation; land use/cover

INTRODUCTION

With rapid population growth, urbanization, industrialization, land-use/cover change, and competition for economic development, groundwater resources have become vulnerable to depletion and degradation. About 1% of the Earth's water is available for human consumption and more than 1.2 billion people still have no access to safe drinking water. Nearly 40% of global food production is attributed to irrigation abstraction, 85% of total human consumptive water use is for agriculture, and food production may soon be limited by water availability (Gleick, 2003; Datta, 2005). Declining water availability is one of the major global environmental challenges for the next century (Postel & Vickers, 2004; Skole, 2004). The increase in cropping intensity, and the replacement of less water consuming crops with more water requiring crops that yield better economic return, have resulted in greater water demand (Datta, 2000).

A considerable amount of work has been carried out on groundwater quality in different parts of India, but no-one has analysed the groundwater quality in respect of the land-use/cover change (Datta *et al.*, 1996; Rao & Devedas, 2003; Rao & Mamatha, 2004; Pathak *et al.*, 2008). Therefore, the present study has been designed to examine the relationship between land-use/cover change and the state of pollution in groundwater.

THE STUDY AREA

The Indo-Gangetic Plain is the most important region of India. The plain is a great alluvial crescent stretching from the Indus River system in Pakistan to the Punjab Plain and the delta of the Ganga. Allahabad district, located in the southern part of Uttar Pradesh, was selected as one of the representative districts for detailed study. It lies between 25°45′–25°00′N and 81°09′–82°45′E. Two important rivers, the Ganga and the Yamuna, divide the district into three physiographic units: Trans-Ganga, Doab and Trans-Yamuna. The normal annual rainfall in the district is 303.4 mm. May is usually the hottest month of the year, with a mean daily maximum temperature of 42°C and a mean daily minimum of 27°C.

MATERIALS AND METHODS

The land-use/cover data of 1990–2007 were collected from the Allahabad District Land Revenue Department. Water samples were collected from different sources in replicate and made a composite sample. Samples were collected in sterilized polyethylene 1-Litre bottles and brought to the laboratory for chemical analysis. Before collecting water samples, the hand pumps and tube wells were run continuously for some time to avoid any turbidity in the water due to the presence of fine mud and sample particles. The samples were collected in the rainy, winter and summer seasons. The water samples from dug wells representing the shallow unconfined aquifer were collected by grab sampling method.

Some parameters, such as pH and EC, were measured before filtering the samples. Other parameters were measured by adding an appropriate reagent, and samples were stored in clean polyethylene bottles fitted with screw caps before being analysed in the laboratory. The pH of the water was determined with a digital pH meter (Elico, Model L1-120) using a double-function reference electrode, and electrical conductivity (EC) was determined with a digital conductivity meter (Systronics, Model 306). Remaining samples were filtered and stored in a refrigerator. All analyses were completed within 10 days following standard methods (Eaton *et al.*, 1995).

RESULTS

Land-use pattern

The land-use pattern in the district as a whole showed about 2.66 and 3.58% of the total area under forest in 1990 and 2007, respectively. Current fallow and other fallow in the district was about 5.25 and 4.85% in 1990 and 6.51 and 4.05% in 2007. Barren land occupied about 4.24 and 1.92%, respectively, in both the years. The cultivable wasteland accounted about 3.21 and 2.50% of the total area and land put to non-agricultural use covered about 8.71 and 12% in 1990 and 2007, respectively. The total net sown area covered about 66.66 and 67.37% in 1990 and 2007, respectively. Area under pasture was 0.33 and 0.50% and that under miscellaneous trees and shrubs covered about 4.09 and 1.60% in 1990 and 2007, respectively (Table 1).

During the 17-year period, the area under forest has increased (36.36%), while barren and uncultivable land decreased by 53.73 in the district. About 20.76% increase was recorded in current fallow, but other fallow, cultivable waste and miscellaneous trees and groves showed a decreasing trend in the district as a whole. Land put to non-agricultural use and pasture land has been continuously increasing over the 17 years (Table 1). The net sown area has increased (28.62%) rapidly since 1990 in all parts of the district.

Table 1 Areas under different land-use/cover and change detection of the Allahabad district of Indo-Gangetic plains, 1990–2007.

Land use	Year 1990 (ha)	(%)	2007 (ha)	(%)	Variations (1990–2007) (%)
Forest	13 364	2.66	18 277	3.58	36.7
Barren and uncultivable land	21 261	4.24	9 836	1.92	–53.7
Current fallow land	26 345	5.25	33 249	6.51	20.7
Other fallow land	23 335	4.85	20 662	4.05	–15.0
Cultivable waste	16 089	3.21	12 767	2.50	–20.6
Land put to non-agricultural use	43 683	8.71	61 258	12.00	40.2
Pasture	1 676	0.33	2 581	0.50	54.0
Misc. trees	20 531	4.09	8 168	1.60	–60.2
Net sown area	333 724	66.66	343 275	67.37	28.6

Groundwater quality

State of pollution The descriptive statistics of the analysed water quality parameters are depicted in Table 2. Analysis of variance of all selected parameters showed significant differences

Table 2 Groundwater quality at different locations in the Allahabad district.

Water quality	Source: Dug Well	Hand Pump	Diesel Pump Set	Electric Pump Set
EC ($\mu S \cdot cm^{-1}$)	1616	1520	1377	1163
pH (-)	8.09	8.08	8.02	8.05
HCO_3 (mg/L)	485	388	376	329
Cl (mg/L)	242	247	231	227
NO_3 (mg/L)	53	43	32	34
SO_4 (mg/L)	110	108	96	91
Ca (mg/L)	29	28	26	23
Mg (mg/L)	61	55	55	54
$CaCO_3$ (mg/L)	241	252	269	268
Na (mg/L)	270	207	194	225

ANOVA: Source $F_{(3, 107)} = 8.167$ ($P < 0.05$); Water quality $F_{(9, 107)} = 673.832$ ($P < 0.05$); Source × Water quality $F_{(27, 80)} = 76.223$ ($P < 0.05$).

with source, water quality and source × water quality (Table 2). Higher values of all the selected parameters were recorded in samples from the dug well, except $CaCO_3$, in comparison to other sources. The groundwater in different parts of Allahabad district is severely affected and has also become considerably vulnerable to pollution with a wide range of contaminants at concentrations ranging as follows (mg/L): HCO_3 (329–485); Cl (227–247) NO_3; (32–53); SO_4 (91–110); Ca (23–29); Mg (54–61); $CaCO_3$ (241–268), and Na (194–270) (Table 2). It is interesting to note that higher values of all the parameters were found in the agricultural areas, while lower values were observed in the settlements. This shows the clear impact of land use on groundwater.

Seasonal variation of groundwater quality Seasonal variation in groundwater quality is presented in Table 3. All the selected parameters reflected the seasonal pattern showing higher values in the rainy season. The highest electrical conductivity (EC) was recorded in the rainy season and the lowest in summer, in all sources (Table 3). The pH of the dug well, hand pump, diesel pump and electric pump waters was alkaline, with the highest value of 8.15 at the dug well. The pH was highest in the rainy season and lowest in summer (Table 3). The highest total hardness (Ca and Mg concentration) was also recorded in the dug well in all seasons. The chloride concentration was highest in the rainy season, decreasing in winter and was lowest in summer for all the sources.

The concentration of NO_3 varied significantly between sources and seasons, and showed a strong negative correlation with only $CaCO_3$. The most important source of nitrogen is the ammonification of organic matter. The dug well source in the rainy season showed high nitrate from farm land. The SO_4 of the dug well, hand pump, diesel pump and electric pump waters varied significantly among sources and seasons (Table 3). It ranged from 84 to 120 mg/L showing high values at the dug well in all the seasons. Comparison between seasons showed that it was lower in the summer and higher in the rainy season. Sodium was highest (279 mg/L) in the rainy season and lowest (191 mg/L) in summer, showing significant variation between sources and seasons (Table 3). The recorded sodium concentration was highest in the dug well and lowest in the diesel pump data set. The higher values of HCO_3 were recorded in the dug well and the lowest in the electric pump set for all seasons. The concentrations of both HCO_3 and calcium carbonate ($CaCO_3$) varied significantly between sources and seasons (Table 3).

Correlation analysis Table 4 shows the correlation matrix and significance levels for all the observed data. A high and significant positive correlation was observed between all selected parameters except $CaCO_3$ (Table 4). Calcium carbonate was negatively correlated with all parameters. The observed pH value, ranging from 8.02 to 8.09, shows that waters were alkaline in nature (Table 2) and were not suitable for drinking. Total hardness of all water samples is near to the highest desirable limit (300 mg/L), but was within the maximum permissible limit (600 mg/L)

Table 3 Seasonal variation in electrical conductivity and groundwater quality at different sources in the Allahabad district.

Source/Season	EC ($\mu S \cdot cm^{-1}$)	pH	HCO_3 (mg/L)	Cl (mg/L)	NO_3 (mg/L)	SO_4 (mg/L)	Ca (mg/L)	Mg (mg/L)	$CaCO_3$ (mg/L)	Na (mg/L)
Dug well										
Rainy	1629	8.18	485	253	61	120	31	64	255	279
Winter	1621	8.09	491	242	28	112	33	62	236	269
Summer	1599	8.00	480	231	69	98	24	56	233	262
Hand pump										
Rainy	1533	8.15	394	261	58	116	33	61	267	216
Winter	1518	8.10	388	244	28	105	25	55	242	207
Summer	1509	8.00	382	235	43	102	24	50	248	198
Diesel pump										
Rainy	1409	8.09	386	260	49	103	31	61	270	201
Winter	1365	8.03	371	219	26	93	26	52	264	191
Summer	1356	7.93	372	213	22	91	21	53	274	191
Electric pump										
Rainy	1198	8.11	342	256	52	98	30	60	279	228
Winter	1153	8.06	328	214	27	90	23	54	267	219
Summer	1138	7.98	318	211	24	84	17	49	260	229

ANOVA:
Electrical conductivity (EC) – Source $F_{(3, 6)} = 978.96$ ($P < 0.05$); Season $F_{(2, 6)} = 15.22$ ($P < 0.05$); pH – Source $F_{(3, 6)} = 23.25$ ($P < 0.05$); Season $F_{(2, 6)} = 0.024$ ($P < 0.05$); HCO_3 – Source $F_{(3, 6)} = 416.26$ ($P < 0.05$); Season $F_{(2, 6)} = 6.12$ ($P < 0.05$); Cl – Source $F_{(3, 6)} = 3.55$ ($P < 0.05$); Season $F_{(2, 6)} = 18.82$ ($P < 0.05$); NO_3 – Source $F_{(3, 6)} = 2.086$ ($P < 0.05$); Season $F_{(2, 6)} = 6.23$ ($P < 0.05$); SO_4 – Source $F_{(3, 6)} = 30.37$ ($P < 0.05$); Season $F_{(2, 6)} = 28.34$ ($P < 0.05$); Ca – Source $F_{(3, 6)} = 3.04$ ($P < 0.05$); Season $F_{(2, 6)} = 15.24$ ($P < 0.05$); Mg – Source $F_{(3, 6)} = 6.91$ ($P < 0.05$); Season $F_{(2, 6)} = 25.56$ ($P < 0.05$); $CaCO_3$ – Source $F_{(3, 6)} = 12.002$ ($P < 0.05$); Season $F_{(2, 6)} = 6.35$ ($P<0.05$); Na – Source $F_{(3, 6)} = 130.89$ ($P < 0.05$); Season $F_{(2, 6)} = 5.67$ ($P < 0.05$).

Table 4 Correlation co-efficient matrix of hydro-chemical data of groundwater in Allahabad district.

Parameters	EC	pH	HCO_3	Cl	NO_3	SO_4	Ca	Mg	$CaCO_3$	Na
EC	-	0.652	0.897	0.873	0.834	0.964*	0.999**	0.760	–0.876	0.426
pH		-	0.641	0.780	0.910	0.812	0.637	0.626	–0.915	0.708
HCO_3			-	0.630	0.899	0.838	0.875	0.967*	–0.891	0.730
Cl				-	0.757	0.952*	0.885	0.461	–0.826	0.248
NO_3					-	0.894	0.813	0.883	–0.993**	0.816
SO_4						-	0.964*	0.709	–0.938	0.474
Ca							-	0.728	–0.860	0.383
Mg								-	–0.846	0.862
$CaCO_3$									-	–0.745
Na										-

* Significant at 0.05 level.
** Significant at 0.01 level.
EC: electrical conductivity.

as per the Bureau of Indian Standards (BIS). The amount of Ca is moderate and within the permissible limit but Mg is slightly high and exceeds the maximum permissible limit (30 mg/L). All samples under observation are rich in chloride concentration and are very close to the permissible limit (250 mg/L) according to Indian standard specifications for drinking water. The same trend is followed in the context of sulphate concentration. Nitrate concentration ranged from 32 to 53 mg/L and exceeded the highest desirable limit (45 mg/L). No prescribed standards are suggested by BIS for the parameters electrical conductivity and HCO_3.

DISCUSSION

The intensive cultivation practices in the Indo-Gangatic plains can be mainly attributed to population growth. The use of excessive doses of fertilizers and pesticides/insecticides on agricultural land to improve soil fertility and for crop protection, and the haphazard disposal of untreated domestic and industrial wastes, have led to groundwater quality deterioration/pollution. Although the study years were limited to the years 1990–2007, it should be noted that significant groundwater depletion and contamination occurred after the "Green Revolution" (1966/67). The decreasing rate of water quality and water availability decline between 1966 and 2007 can be attributed to the interaction of a number of factors, including improvements in irrigation technology, government programmes and high-yielding varieties of seeds (Kettle *et al.*, 2007).

The groundwater is alkaline and has a very hard character, but is within the permissible limit of potable water (WHO, 1984). The higher pH was attributed to the presence of carbonates (Jain *et al.*, 1999). The total hardness of all the water samples slightly exceeds the highest desirable limit (269 mg/L). The Ca was at moderate levels, but Mg was slightly high and exceeded the maximum permissible limit (30 mg/L). The high level of Mg ions was related to the weathering of ferro-magnesium mineral and anthropogenic sources, as also suggested by Hem (1991), Zhang *et al.* (1995), Satyanarayan & Periakali (2003) and Rao *et al.* (2006). All water samples were rich in chloride, with concentrations ranging from 231 to 247 mg/L, and very close to the desirable permissible limit (250 mg/L) as per Indian Standard specification for drinking water. The same trend was followed by sulphate. Nitrate concentration in all sampled water exceeded the highest desirable limit. The enrichment of Na and Cl ions in groundwater was due to the interaction of water with rocks and, secondly, the association of TDS with higher concentrations of Na and Cl ions. This indicates anthropogenic activities, such as discharge of sewage in open pits, which allow the sewage to percolate and finally mix with groundwater (Pathak *et al.*, 2008).

CONCLUSIONS

Few studies have documented the detailed spatial relationship between groundwater and land-use/cover change. Results indicate that there is an identifiable relationship between groundwater deterioration and intensive agricultural practices in the Ganga-Yamuna plains. The detailed findings of this research, based on an intensive local study, are consistent with other studies that have shown that intensive use of chemical fertilizers and pesticides to meet the food demand have accelerated the contamination level of groundwater in the study area. In general, the groundwater of the selected villages is alkaline in nature and hard, and meets the prescribed WHO and Indian standards.

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Impacts of land use on groundwater quality in Western Australia

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Abstract To determine the effects of land use on groundwater quality in Western Australia, a quantitative analysis is carried out using groundwater quality data supplied by the Department of Water from over 500 groundwater wells across the Perth metropolitan area. We analysed four main groundwater quality indicators; nutrients, physical parameters, inorganic non metals and trace metals. We found that groundwater beneath agricultural land was found to be particularly susceptible to nutrient loading due to the application of fertilisers. Nutrient levels were found to be rising over time due to increasing agriculture and urban developments. Industrial areas were also found to have numerous contamination plumes that continue to migrate with the groundwater flow. According to Australian and New Zealand Environment and Conservation Council (ANZECC) guidelines and the Australian Drinking Water Guidelines (ADWG), several areas are identified as vulnerable for groundwater quality, including: rural areas (Carabooda Lake, Gnangara and Jandakot Mounds, Cockburn Sound, Forrestdale, Joondalup and Ellenbrook); high-density urban areas (Balcatta and Neerabup); and industrial areas (North Fremantle, Welshpool and Kwinana).

Key words groundwater quality; pollution; land use; Western Australia

INTRODUCTION

Freshwater is fast becoming a scarce resource in Australia. The largest available source of freshwater is currently groundwater. As groundwater has a huge potential to ensure future demand for water, it is important that human activities on the surface do not negatively affect this precious resource. It is important to Western Australia as it supplies approximately 70% of Perth's freshwater for industrial, agricultural and municipal use (National Water Commission, 2007). Reduced rainfall in recent years and diminishing inflow has led to depleted dam levels in Perth (National Water Commission, 2005). With Western Australia's population growth rate of 2.3%, coupled with concerns about climate change, Perth is facing significant water shortages and demand for groundwater continues to rise (Australian Bureau of Statistics, 2008). It is becoming increasingly important that the quality of Perth's groundwater system be monitored and sustained for years to come.

Pollution of groundwater occurs when waste products or any foreign substance alters the biological or chemical characteristic of water and degrades the quality so that animals, plants or humans are affected (Waters and Rivers Commission, 1998). Perth groundwater is particularly susceptible to groundwater pollution as the area has predominantly sandy soils that are generally correlated with low adsorption potentials, allowing for easy leaching of foreign chemicals through the soil profile (Davidson, 1995). Although the progression of underground contaminants depends on numerous microbiological, physical and chemical processes, the most significant factor controlling contamination of groundwater is the source of contamination on the surface, including its type, strength and location relative to the water source (Erckhardt & Stackelberg, 1995). By studying the relationship between groundwater contamination and land use, issues of sustainability can be addressed and integrated with better land-use practices and water protection strategies.

STUDY AREA: PERTH BASIN

The Perth basin, Western Australia is selected as the study area. The geological formations of the Perth basin have been grouped into six distinct aquifers: the Superficial, Rockingham, Kings Park, Mirrabooka, Leederville and Yarragadee. These aquifers are locally, hydraulically connected or separated by confining beds or geological formations (Davidson, 1995). Drainage patterns and hydraulic characteristics are categorised into distinct groundwater flow systems known as groundwater mounds. The Gnangara Mound is Perth's most important water source, supplying 380 GL of freshwater per year. Recent studies have found that extractions from the mound are close to its sustainable limits (Department of Water, 2008).

In the past, Perth's groundwater has been described as generally good in quality and predominantly pollutant free. However, in recent years, investigations have shown increased levels of dispersed contaminants, as well as a significant number of localised contamination plumes (Hirschberg, 1991; Barber *et al.*, 2005). Land use has developed considerably since the development of the Swan River Colony in 1829. As Perth continues to grow and expand, there are increasing agricultural (primarily horticulture) and aquaculture developments over the Perth basin. Furthermore, urban developments are intruding on the boundaries of the Gnangara and Jandakot reserves (Department of Environment, 2008). It is becoming increasingly important to have more stringent protection and management of Perth's groundwater sources to limit the effects of urbanisation on groundwater quality.

MATERIALS AND METHODOLOGY

The Department of Water (DoW) supplied qualitative data on 500 groundwater wells located within the Perth metropolitan area. Available data are based on samples tested from 1984 to 2008. Based on available data, water quality data were grouped into historic (1984–1994), recent (1995–1999) and current (2005–2007) periods. For each period, data were analysed across the Perth metropolitan area to determine if there are any correlations between groundwater quality parameters and location. The data were assessed for spatial and temporal trends. A geostatistical method of analysis was utilised that uses mathematical models to determine temporal and spatial trends in the data. The SURFER mapping system by Golden Software Inc. (SURFER, 1994) was used to find trends in concentration distributions of groundwater quality parameters. The following groundwater quality indicators were analysed:

- nutrients: total nitrogen (TN), nitrate (NO_3), ammonia, total phosphorus (TP)
- physical: dissolved oxygen (DO), acidity (pH), alkalinity ($CaCO_3$)
- inorganic non metals: arsenic (As), chloride (Cl), boron (Bo)
- trace metals: zinc (Zn), lead (Pb), nickel (Ni)

Spatial distributions were examined and compared with assessment levels for groundwater quality adopted from the Department of Environment and Conservation (DEC). The DEC evaluates contaminated groundwater sites according to the Australian and New Zealand Environment and Conservation Council (ANZECC) guidelines (ANZECC, 1992) and the Australian Drinking Water Guidelines (ADWG) (Department of Environment, 2003). Results were discussed, comparing them to the land use for the study area. A map of land use for the Perth Metropolitan Region Scheme (MRS) was supplied by the Department of Planning and Infrastructure through their Mapping and Geospatial Data (Department of Planning and Infrastructure, 2008). The maps specify land use across the Perth metropolitan area, including: urban, industrial, state forests, parks and private recreational areas; as well as rural and agricultural lands.

RESULTS AND DISCUSSION

Comparison of spatial distribution of groundwater quality and land-use pattern

Several water quality parameters were analysed for historic, recent and current periods and their spatial distribution mapped using the SURFER software. Figure 1 shows the spatial distribution of TN, alkalinity, As and Zn to represent the distribution of nutrients, physical parameters, and inorganic non metals and trace metals in the current period (2005–2007). Results for TN showed

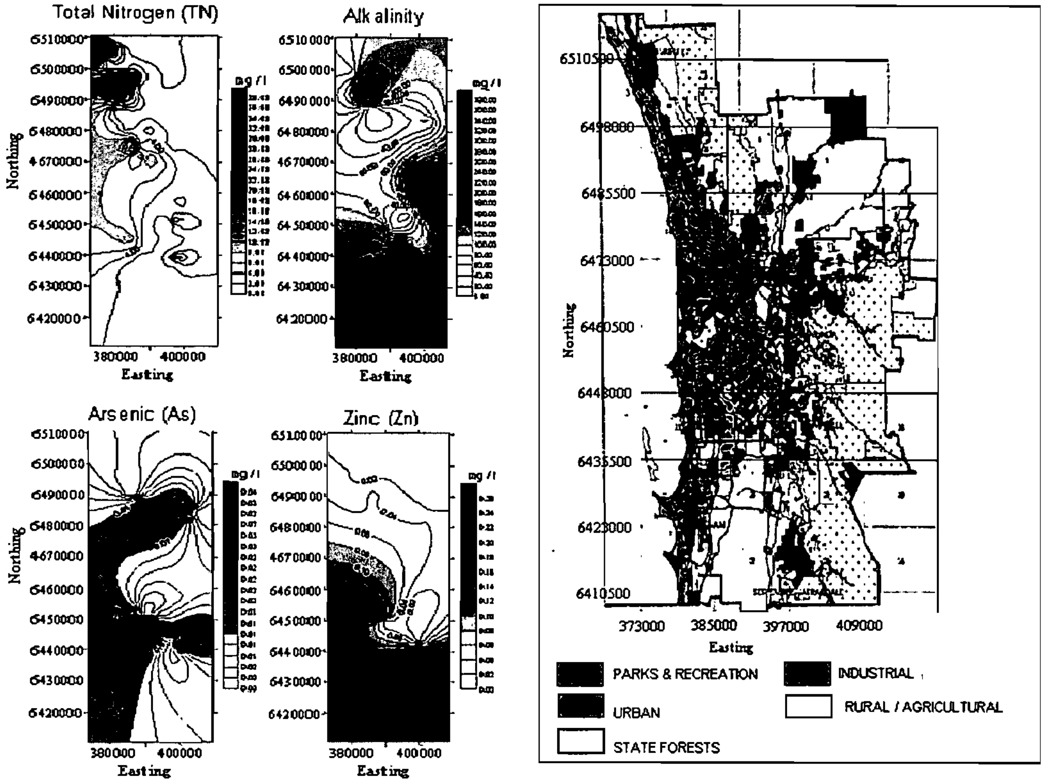

Fig. 1 Spatial distribution of TN, alkalinity, arsenic and zinc, and land-use pattern in the Perth region (2005–2007).

that levels in Perth are significantly high. Historic samples had an average total nitrogen content of 3.2 mg/L. This amount exceeds the ANZECC trigger limit for freshwater of 700 µg/L. Furthermore, the TN average increased to 3.6 mg/L for the 2005–2007 period. The spatial distribution indicates high levels of nitrogen in northern rural areas around Carabooda Lake. This result is a cause for concern, as this area lies over the Gnangara Mound.

Figure 1 further shows that low alkalinity is evident in the northern suburbs of Joondalup and Ellenbrook. These areas are less likely to resist changes in acidity levels. Studies have found that groundwater with alkalinity ($CaCO_3$) as low as 30 mg/L are vulnerable to rapid drops in acidity and increases in iron, aluminium and heavy metal concentrations (DEC, 2008). Low alkalinity can be problematic during dewatering in preparation for new urban developments. Low alkalinity can support severe changes in pH levels. This analysis also confirms that observed pH levels are generally declining and there are areas with elevated levels of inorganic metals and non metals.

The concentration of Zn in Australian drinking water supplies is generally around 0.05 mg/L, but can be as high as 0.26 mg/L. There is no health guideline for zinc; however, for aesthetic purposes, a limit of 3 mg/L is set by the ADWG. Water exceeding this amount can develop a metallic taste and can develop a "greasy" film when boiled (NHMRC, 2004). In 2007, 48 groundwater wells were tested and 301 samples tested for zinc between 0.001 and 1.5 mg/L. Average concentrations for each site were calculated; values ranged from 0.003 to 0.419 mg/L (Fig. 1). Findings for zinc were found with hot spots located over Forrestdale Lake. Further investigation is required to decipher the source of the contamination.

Recent data (2005–2007) tell that 44 groundwater wells across Perth were tested for arsenic and 283 samples were taken. Average arsenic concentrations ranged from 0.005 to 0.358 mg/L, with average and median concentrations of 0.027 and 0.003 mg/L, respectively. In this period, 15 groundwater wells registered an average concentration in excess of the Australian Drinking Water Guideline of 0.007 mg/L (Fig. 1). Arsenic can enter the groundwater by leakage of industrial effluents, waste incineration and burning of fossil fuels, etc. External contaminants can also react with naturally occurring arsenic in the soil and cause it to discharge into groundwater. The ADWG sets a limit of 0.007 mg/L for drinking water based on health reasons and possible carcinogenicity. According to the NHMRC (2004), arsenic concentrations in Australian waters tend to be less than 0.005 mg/L. From the spatial distribution map, Forrestdale Lake and North Lake show elevated levels of arsenic. North Lake is close to urban areas and Forrestdale is in a rural-residential setting. Further analysis is required to determine the cause of the contamination.

The current groundwater quality of the other parameters was also verified by analysing the 2005–2007 data. The study found that many areas in Perth have groundwater quality outside the limits given in the ADWG guidelines. High nitrate levels are correlated with TN levels, with critical levels being recorded in some rural locations over the Gnangara and Jandakot Mounds. Ammonia was also registered in areas at critical levels exceeding ADWG guidelines of 1.5 mg/L for all testing periods. Furthermore, 70% of samples tested for TN were over the ANZECC trigger limit. Ammonia samples measured over the ANZECC trigger limit of 900 μg/L in 16% of samples. Arsenic quantities are also high with 15 sites found to be in excess of the ADWG limit. Chloride and Ni levels are also extreme in 21 and 10 sites, respectively. The pH levels of 33 sites are outside the neutral limits of 6.5–8.5, with the majority showing acidic conditions. Interestingly, the majority of the samples for 2005–2007 are taken in close proximity to water catchment areas. It is presumed that these areas should have higher quality groundwater than other areas of Perth. The results suggest that groundwater conditions in other areas are likely to be of moderate quality.

The other testing periods (historic and recent) also found critical areas of TN in rural land east of Kwinana and in high-density urban areas near Balcatta, with levels reaching as high as 89.6 mg/L. Also many rural areas have nutrient levels in excess of the ADWG guidelines. Urban areas also show elevated nutrient levels owing to residential activities and a history of agriculture and fertiliser use. Testing for the three periods indicates that nutrient levels in Perth are typically higher in rural areas, where fertilizer is the primary pollution source. Perth sandy soils do not readily adsorb nutrients, so fertiliser can easily leach into superficial aquifers, especially when over watered. Urban areas have also shown higher levels of nitrogen, but further investigations are required to determine whether the pollution source is due to municipal or historical agricultural use. Recent sampling data are limited to predominantly rural land close to water-protected sites. Inorganic non metals and trace metals in groundwater in Western Australia include site-specific spills or leaks from industrial and waste disposal sites. A large number of contaminated sites have been recorded in industrial areas, including: North Fremantle, Welshpool and Kwinana. Restrictions have been placed on these sites that limit the land zoning to either industrial or commercial. Groundwater extractions are prohibited in the majority of these sites based on health concerns of the contaminants.

Temporal change of nutrient content

Nutrient content was examined in more detail as it is considered the best indicator of anthropogenic processes and impact from land-use activities. Figure 2 illustrates the change in TN and TP between historical and current periods. The temporal changes also show similar results to the spatial distributions for TN, with a general rise in content towards the outer suburbs where rural land is predominant. These distributions indicate critical areas near Carabooda and Neerabup, north of Perth. Both areas are in close proximity to rural land and Neerabup also lies next to industrial developments. The distributions show a reduction in nitrogen in the inner urban suburbs of Perth towards the mouth of the Swan River. Variations in TP appear less severe than nitrogen levels. Increased TP was only found in a few dispersed areas across Perth. Rural areas inland from the Cockburn Sound, Forrestdale and Southern River (south of Perth) were found to have critical changes in phosphorus levels.

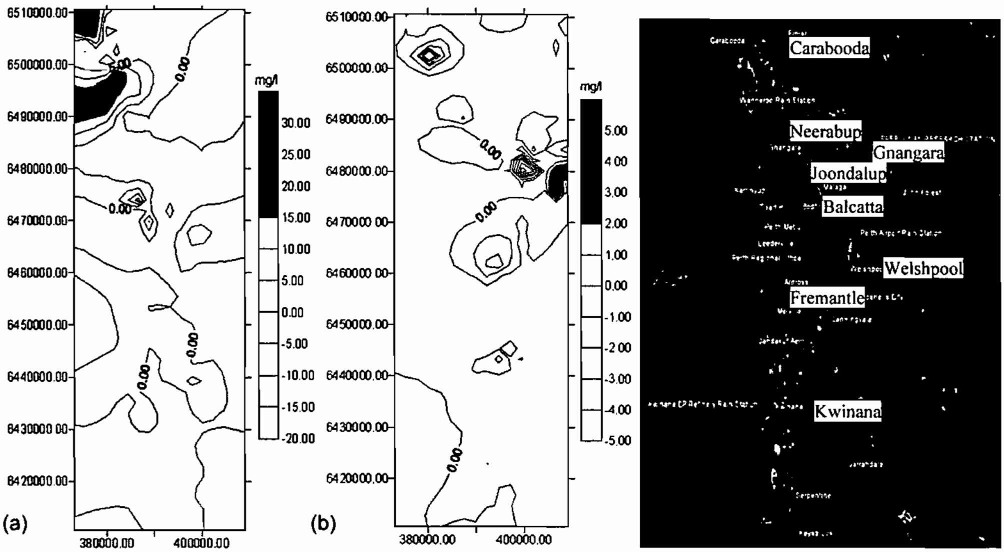

Fig. 2 Variations in: (a) total nitrogen and (b) total phosphorous between 1984 and 2007; and an aerial map showing sampling locations.

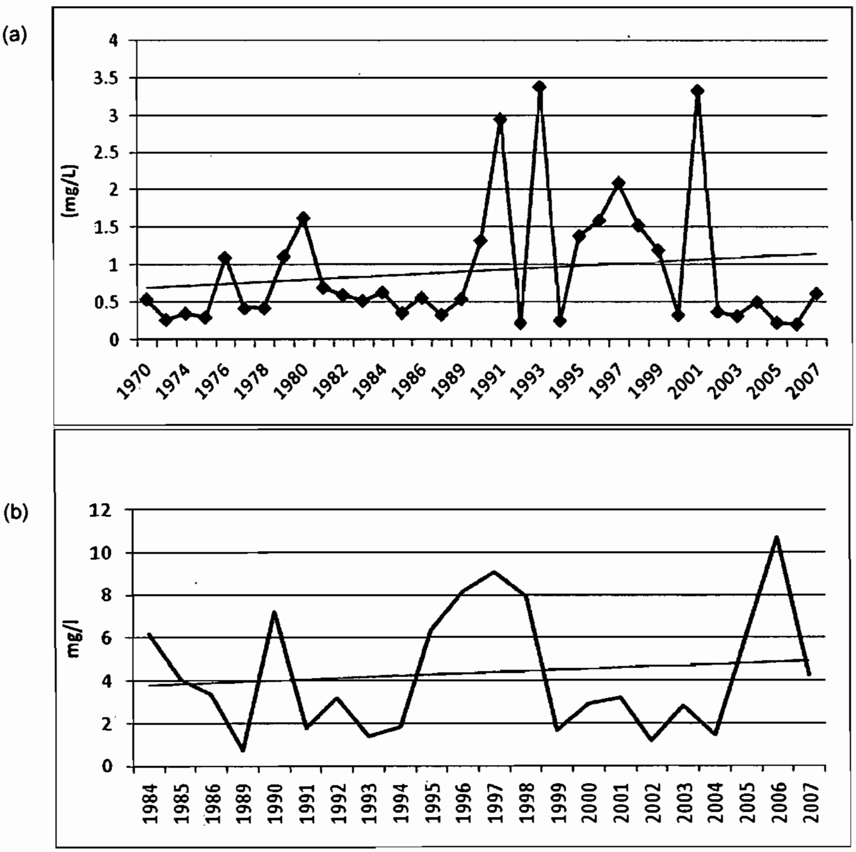

Fig. 3 Variation of: (a) annual average TN (1970–2007); and (b) annual average TP (1984–2007).

The fertiliser content and characteristics of the soil in these areas should be analysed further to determine more conclusive outcomes.

Variation of nutrient content over time

To establish a general understanding of temporal trends, changes in annual average TN and TP were considered. The study found that there is a general rise in nutrient concentration in groundwater over time for the Perth metropolitan area. Figure 3 shows the average increasing TN and TP trend for the 500 groundwater wells. Similar increasing trends were found for all nutrient parameters, including total nitrogen, nitrate, ammonia and total phosphorus.

Contrary to this trend, groundwater wells with elevated levels of total nitrogen were found to show decreasing levels in water-protected areas. These findings indicate that nutrient content is affected by land use, but can be controlled with proper management and monitoring processes. This result, combined with the large number of groundwater wells exceeding trigger limits, demonstrates the need for the establishment of long-term monitoring and management programmes over a broader region.

CONCLUSIONS

Perth water sources have reached their sustainable limits, so it is increasingly important to protect Perth's groundwater to ensure its quality is maintained. This study found that groundwater beneath agricultural land was found to be particularly susceptible to nutrient loading due to the application of fertilisers. Nutrient levels were found to be rising over time due to increasing agriculture and urban developments. Industrial areas were also found to have numerous contamination plumes that continue to migrate with the groundwater flow. Therefore this study concludes that there are many areas in Perth, including rural areas such as Carabooda Lake, Gnangara and Jandakot mounds, Cockburn Sound, Forrestdale, Joondalup and Ellenbrook, high density urban areas such as Balcatta and Neerabup, and industrial areas such as North Fremantle, Welshpool and Kwinana, that are showing critical levels of nutrients, inorganic metals and heavy metals. Furthermore, temporal analysis indicates increasing trends in nutrient level. These findings indicate that nutrient content is affected by land use but can be controlled with proper management and monitoring processes. This result, combined with the large number of groundwater wells exceeding trigger limits, demonstrates the need for the establishment of long-term monitoring and management programmes over larger target areas.

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Dissolved nitrogen removal in the ponded streams of an alluvial, suburban basin with a developing city, western Japan

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Abstract This study aims to clarify the nitrate removal in a ponded stream with 12 pools, in western Japan. In the summer season, denitrification coincides with nitrate assimilation by phytoplankton in the pool. In addition, organic matter mineralization in sediment also occurs in summer. DN fluxes estimated on the main stream by the chloride ion balance method suggest nitrate has been removed within the pool. However, the estimated amount of nitrogen removed in each pool was quite different. The estimated amount removed was divided into denitrification and assimilation by the Redfield ratio. Consequently, it is suggested that assimilation by phytoplankton contributes significantly to removing dissolved nitrogen in the summer season, while denitrification contributes in autumn. Furthermore, the relationship between the amount of removed dissolved nitrogen, except for assimilation, and characteristics of pools, indicates that the removable amount of DN is larger in the pools characterized by large area, shallow depth or slow flow velocity.

Key words small reservoir; river basin scale; denitrification

INTRODUCTION

The ability for nitrogen removal in freshwater systems is important to decrease the impact of nitrogen loading by human activities (Saunders *et al.*, 2001).

As human activities continue to alter the global nitrogen cycle, the ability to remove the impact of increased nitrogen loading to freshwater systems is becoming more and more important (Saunders *et al.*, 2001). Mass balance studies have shown that reservoirs may retain a large portion of N inputs to the catchment (Andersen, 1971; Kennedy & Walker 1990). Josette *et al.* (1999) suggest that sediment denitrification could be an important retention mechanism. Ochoa *et al.* (2006) has confirmed the ability of denitrification in coastal area, estuaries, lakes and rivers. Garnier *et al.* (1999) has confirmed a clear loss or retention of nitrogen in the large reservoirs of the Seine Basin and estimated that it is equal to about 40% of the incoming flux of nitrate. However, most of the previous studies show a nitrate removal effect only in large reservoirs. It is important to manage nitrate emission control in freshwater systems.

The objective of this research is to estimate the amounts of dissolved nitrogen removed in the pools based on mass balance and to evaluate the nitrate removal effect of ponded streams in a suburban basin with a developing city in Japan.

STUDY AREA AND METHOD

The study area is located in a watershed of Takaya River, which is a tributary of the Ashida River in western Japan (Fig. 1). The area of the Takaya River watershed is about 130 km^2. The study area is characterized by temperate climate with annual precipitation of 1100 mm and mean temperature of 15.0°C. The average monthly rainfall is 25–105 mm during the dry season and 113–221 mm during the rainy season. The catchment population has grown dramatically, from 87 000 to 94 650 during the 1970s and in 2006 was approx. 101 425 people. The rapid conversion of agricultural area to suburban residential areas has occurred with only immature domestic wastewater treatment systems in the middle to downstream area of the catchment. Accordingly, domestic wastewater drains directly into the river.

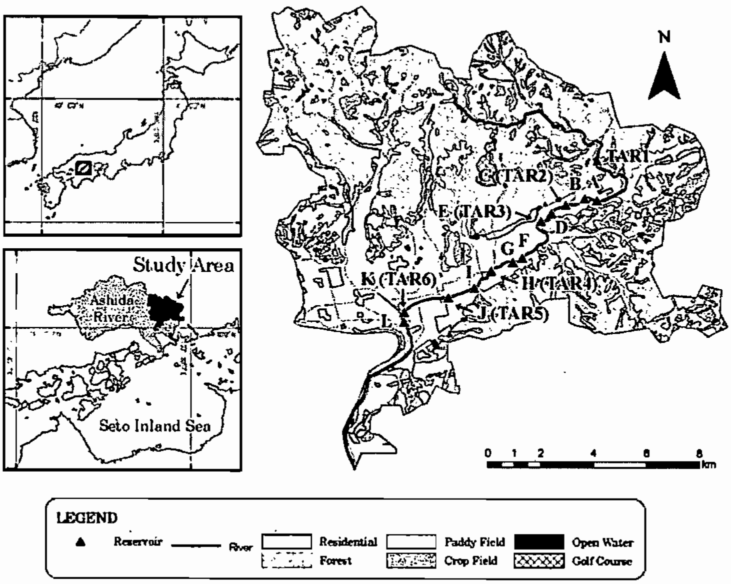

Fig. 1 Location of study area and intake weirs with pools in the Takaya River. From TAR1 to TAR3 are the stream is lotic, in contrast, from TAR4 to TAR5 are lentic stream (pools).

Table 1 Characteristics and size of pools behind the intake weirs.

Pool	Mean Depth (m)	Length (m)	Mean width (m)	Area (m^2)	Mean flow velocity ($m\ s^{-1}$)
A	0.9	361	12	1 905	0.21
B	1.1	195	18	6 332	0.18
C	0.9	288	13	8 254	0.19
D	0.7	1084	17	7 111	0.16
E	1.2	846	27	20 384	0.05
F	0.2	328	29	8 015	0.12
G	0.5	823	25	16 401	0.09
H	0.7	389	17	6 624	0.06
I	0.7	257	11	4 537	0.05
J	0.6	866	39	23 322	0.02
K	1.2	1043	35	34 392	0.01
L	0.7	331	37	14 213	0.03

Paddy field covers the alluvial area around the river. For the irrigation, there are a lot of intake weirs and intermittent small pools behind the weirs in the middle part. Table 1 shows the characteristics and size of pools of the ponded stream on this catchment. The locations are shown in Fig. 1. The size of pools downstream is relatively large compared with those in the upstream. Water depth increases with the size of pools, but flow velocity decreases.

To confirm the nitrogen budget and removal amount in pools of the ponded stream, water sampling of the river, domestic wastewater and groundwater, and river runoff observations were conducted at seven points on the main stream and another four on the tributary. Field measurements were conducted twice during both the irrigation period (July and September 2007) and non-irrigation period (October 2007 and April 2008). The EC, pH and DO were measured during the sampling in the field. Water samples were collected into polyethylene bottles after filtering with 0.2 μm cellulose ester membrane filters, and were analysed for chemical and isotopic components. The Cl^-, NO_3^- and SO_4^{2-} concentrations were analysed by ion chromatography (HPLC

LC-VP, SHIMADZU). Nutrient concentrations (NO_3^-, NO_2^-, NH_4^+, total dissolved nitrogen, PO_4^{3-}, and total dissolved P) were measured colorimetrically with an autoanalyser (BL-tec, swAAt). Nitrogen stable isotope ($\delta^{15}N$) was measured by mass spectrometer (Delta-S, Thermo Finnigan).

RESULTS AND DISCUSSION

Spatial variation in dissolved nitrogen

Figure 2 shows the variation of NH_4^+-N, NO_2^--N, NO_3^--N and dissolved organic nitrogen (DON) concentrations in river water from the upper stream (TAR1) to downstream (TAR6) in July 2007. Concentrations of nitrogen and the other components change rapidly between TAR3 and TAR4. The NO_3^--N concentration decreases from 1.79 to 0.68 mg L^{-1} from TAR3 to TAR4. In contrast, NH_4^+-N, NO_2^--N, and dissolved organic nitrogen (DON) concentrations increase. The pH in the surface water of the pool at TAR4 is 9.4 and is higher than 6.9 at TAR3. In addition, the saturation percentage of DO rises from 135% to 178%. These results suggest two principal nitrogen cycle processes in the ponded streams. The first possibility is reduction processes in the anoxic layer of the pool bottom. Another is nitrate assimilation by growing algae and phytoplankton. However, nitrogen mineralization and phosphorus dissolution are also suggested.

Table 2 shows the average concentration of NO_3^--N and $\delta^{15}N$ in the upstream (TAR1 to 3) and pools (TAR4 to 5), respectively. The river water in the upstream is characterized by high NO_3^--N concentration (1.9 mg L^{-1}) and low $\delta^{15}N$ (10.0 ‰), compared with those in the river water at the irrigation area (Fig. 1). The $\delta^{15}N$ in the downstream through the ponded stream rose to 12.5 ‰ and the NO_3^--N concentration fell 0.7 mg L^{-1}. This isotopic property indicates that the nitrogen source is domestic wastewater. In addition, the inverse variation in NO_3^--N and $\delta^{15}N$ suggests that nitrate is attenuated by denitrification under the reductive conditions in the ponded stream, rather than by dilution effects. Natural denitrification must occur at the bottom of the pools, because there are oxygenated conditions in the surface water due to photosynthesis.

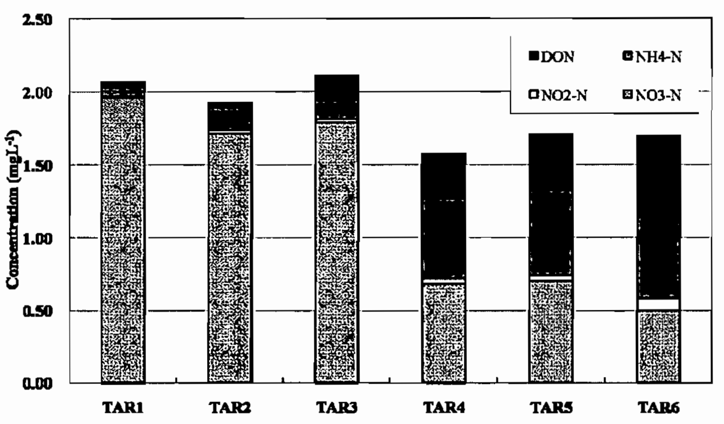

Fig. 2 Variation in nitrogen species concentrations from upstream to downstream in July, 2007.

Table 2 Average concentration of NO_3^--N and $\delta^{15}N$ (July, 2007).

	NO_3^--N conc. (mg L^{-1})	$\delta^{15}N$ (‰)
Stream	1.8	10.0
Pool	0.7	12.5

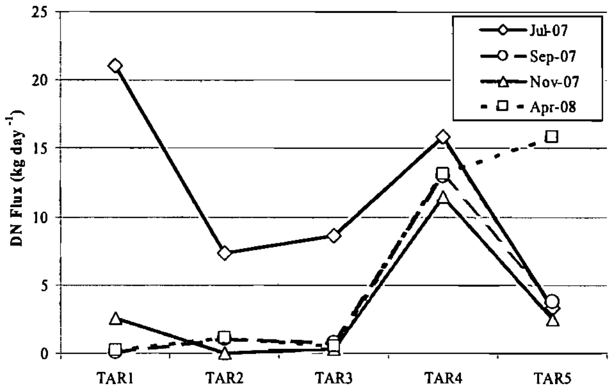

Fig. 3 Variation in dissolved nitrogen flux on the main stream.

Figure 3 shows variation in the dissolved nitrogen flux from the upstream to downstream in the four seasons of 2007 to 2008. The flux of dissolved nitrogen increases in July 2007 more than in other seasons due to increase of river discharge. The flux of dissolved nitrogen rises at TAR4 in all seasons. Because the dissolved nitrogen concentration decreased at TAR4 as described above, this must reflect the influence of the inflow of the big tributary rather than production of dissolved nitrogen by mineralization. On the other hand, the flux has decreased suddenly in TAR5. It has implied that dissolved nitrogen was removed in the pool behind a weir.

Nitrogen budget in a pool of a ponded stream

In order to evaluate the magnitude of the removal effect of dissolved nitrogen in the pool of the ponded stream, it is necessary to estimate the dissolved nitrogen budget. We assumed the water balance composed of inflows from the main stream, tributary and domestic wastewater to the pool, outflow from the pool to the downstream, and river water leakage to groundwater, as in equation (1). Therefore, the dissolved nitrogen removal effect was based on water and material fluxes. We estimated the unknown inflow flux and outflow flux between one observation point and the next one by simultaneous equations shown as below:

$$Q_{i+1} = Q_i + Q_t + Q_x - Q_y \tag{1}$$

where Q_{i+1} is river discharge from the pool to the downstream after confluence with the tributary and unknown inflow, Q_i is the inflow from the main stream to the pool, Q_t is the inflow from the tributary, Q_x is the inflow from the domestic wastewater, and Q_y is the leakage from the river to the groundwater. However, Q_x and Q_y are the unknowns. Therefore, we need the next mass balance equation for the non-reactive species. We used the Cl^- budget:

$$C_{i+1} = C_i + C_t + Q_xC_x - Q_yC_y \tag{2}$$

where C_{i+1} is Cl^- flux after the confluence with the tributary and unknown inflow, C_i is Cl^- flux of the main stream before confluence, C_t is Cl^- flux of tributary, C_x is Cl^- concentration of domestic wastewater, C_y is Cl^- concentration after confluence with tributary. Dissolved nitrogen attenuation was estimated from the difference between the calculated nitrogen flux based on the Cl^- budget and the observed flux as follows:

$$R = N_o - N_c \tag{3}$$

$$R = N_o - N_i + N_t + Q_xN_x - Q_yN_y \tag{4}$$

where R is removed dissolved nitrogen, N_o is dissolved nitrogen flux measured by observation, N_c is calculated dissolved nitrogen flux with unknown flow, N_i is dissolved nitrogen flux of main stream, N_t is dissolved nitrogen flux of tributary, N_x is dissolved nitrogen concentration of domestic wastewater, N_y is dissolved nitrogen concentration after confluence with the tributary.

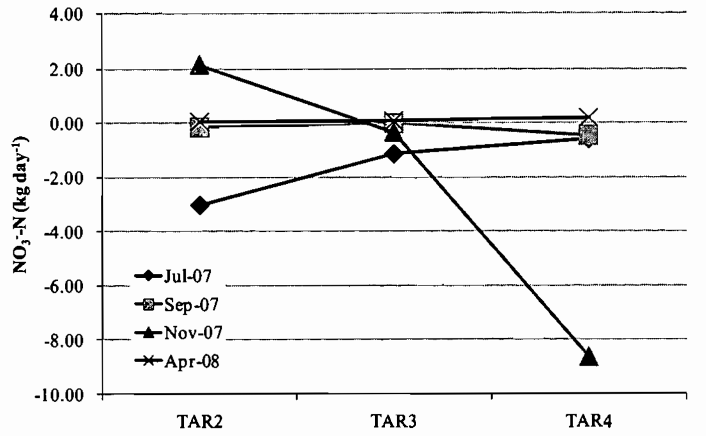

Fig. 4 Nitrate retention between TAR2 and TAR4 based on mass balance with unknown sources.

Figure 4 shows nitrate retention between TAR2 and TAR4 based on the mass balance with unknown sources. The removal potential of dissolved nitrogen was high at TAR4 in all seasons, and it varied from –0.1 kg day^{-1} to –2.5 kg day^{-1}. The amount of removed nitrate at TAR4 has varied from +0.1 kg day^{-1} to –5.4 kg day^{-1}. In contrast, the ammonium flux has increased at TAR4 in the summer season due to mineralization from the bottom of pool. Removed dissolved nitrogen increased at all stations in July 2007 and it varied from –0.7 kg day^{-1} to 4.0 kg day^{-1}. However, the removal effect of dissolved nitrogen was negligible in other seasons except at TAR4. Denitrification and assimilation coincide with mineralization in the pool during the summer season.

Removal effect of dissolved nitrogen

The removal amount calculated by the mass balance includes the assimilation in the pool. For separation of the dissolved nitrogen removal effect by denitrification and assimilation, we estimated the assimilation amount in the pool, based on the Redfield ratio which is the N/P ratio of the phytoplankton. Removed dissolved nitrogen and phosphorus is assumed to be assimilated by the phytoplankton. After we calculated the P budget based on the equation (4), we estimated the removed phosphorus and nitrogen by assimilation. Table 3 shows the result of estimation of the amount of dissolved nitrogen removed due to assimilation at TAR4. This result indicates that 92% of dissolved nitrogen was removed by assimilation in July 2007. In contrast, the contribution ratio of denitrification was high during the autumn season. In November 2007 especially, 100% of dissolved nitrogen was removed by denitrification.

It is necessary to clarify the relation between the dissolved nitrogen removal effect and physical characteristics, such as water depth, flow velocity and pool bottom area for estimating the dissolved nitrogen removal effect at the all pools. The removal coefficient of dissolved nitrogen was calculated by multiple regression analysis:

Table 3 Estimation of removed dissolved nitrogen except assimilation at TAR4.

	Removed DN (kg day^{-1})	Removed DP (kg day^{-1})	DN removed without assimilation (kg day^{-1})	(%)*
Jul-07	0.746	0.095	0.060	8
Sep-07	0.989	0.040	0.697	71
Nov-07	2.531	–0.062	2.531	100
Apr-08	0.103	0.007	0.051	49

* The contribution ratio of denitrification to removed DN.

$$Y = 22.789D + 79.56V - 0.00044A - 11.393 \quad (5)$$

where Y is the amount of removed dissolved nitrogen per day, D is mean depth of pool, V is mean flow velocity, A is the bottom area of pool. Adjusted R^2 of this formula is 0.968, and the number of samples (n) is 12.

Figure 5 shows removal potential for dissolved nitrogen at each pool on the main stream. This result indicates that not all pools become removers for dissolved nitrogen. In particular, pools in the upper stream have increased dissolved nitrogen. In contrast, pools downstream have decreased amounts. Equation (5) indicates that dissolved nitrogen removal efficiency is related to the bottom area of the pools: the smaller the bottom area, the smaller the dissolved nitrogen removal efficiency.

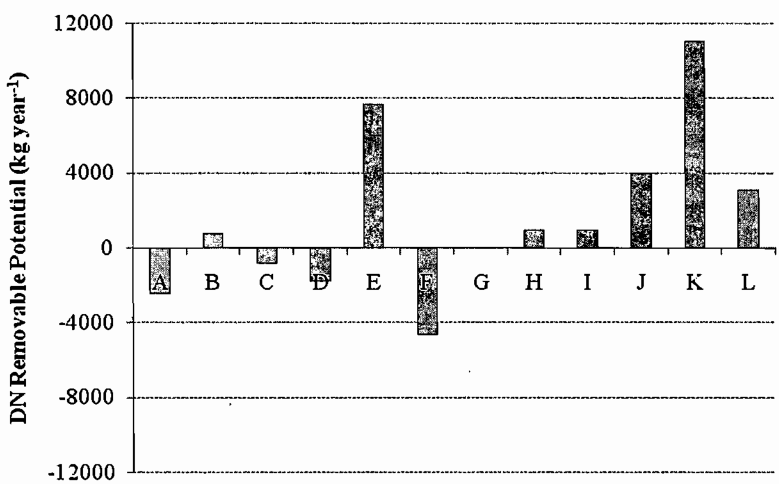

Fig. 5 Removal potential for dissolved nitrogen at all pools on the main stream. The alphabet increases toward the downstream.

CONCLUSIONS

The variations in nitrogen concentration and flux were confirmed and removed DN flux was estimated for the ponded stream, in western Japan, to clarify the effects of dissolved nitrogen removal at 12 pools. Decrease of nitrate concentration and the enrichment of $\delta^{15}N$ indicate that denitrification occurred in the pool. In addition, nitrate removal by assimilation by growing algae or phytoplankton has also occurred. The flux of dissolved nitrogen at the main stream estimated by mass balance suggests that it is removed in the pool. The nitrogen removal process was divided into denitrification and assimilation by the Redfield ratio. The result demonstrates that assimilation significantly contributes to remove dissolved nitrogen in the summer season. In contrast denitrification contributes in the autumn season. The relationship between the amount of removed dissolved nitrogen except for assimilation and pool characteristics indicate removal of dissolved nitrogen is larger in the pools with a large bottom area, shallow depth and slow flow velocity. The magnitude of removal of dissolved nitrogen depends on characteristics of the pools.

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River-mouth dam effect on integrated management of a coastal water resource in a Seto Inland Sea watershed, Western Japan

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Abstract To evaluate the groundwater resource in the coastal area of the Ashida River, which is one of the catchment of the Seto Inland Sea, we tried to estimate the variation of the volume of "fresh" groundwater (no seawater intrusion) before and after the construction of the river-mouth dam. The result suggests that it has increased since the dam construction in 1978, apparently by the shift of the brackish–freshwater boundary from inland to the coastal line. The volume of brackish water was estimated to be approximately twice that before 1978. The result suggests that the coastal groundwater is a valuable freshwater resource in the study area for the future.

Key words coastal groundwater; water use; qualitative problem; city water supply; river-mouth dam

INTRODUCTION

Coastal groundwater is important as a water resource for coastal city water supply, in addition to reservoirs in the upstream area. However, excessive pumping of groundwater can cause saltwater intrusion in confined and unconfined coastal aquifers. In many coastal megacities particularly, such as Bangkok and Jakarta, land subsidence and groundwater salinization are serious environmental problems (Phien-wej *et al.*, 2006; Abidin *et al.*, 2007; Onodera *et al.*, 2008). It is important to evaluate the available coastal groundwater resource and use it sustainably for the future.

In the case of the watershed of the Seto Inland Sea in western Japan, despite the high demands on the freshwater resource, little coastal groundwater has been used. Large amounts of city water and industrial water are supplied from the river, and the river-mouth dam was constructed to reserve the industrial water and prevent saltwater encroachment. However, the river water near the dam is contaminated and has recently become eutrophic. The objective of this study is to evaluate the groundwater resource in the coastal area of the Ashida River which is one of the catchments of the Seto Inland Sea. We focused especially on the effect of construction of the river-mouth dam on the coastal groundwater resource.

STUDY AREA

The study area is located on the Ashida River catchment in the Hiroshima prefecture of western Japan (Fig. 1). The main river flows from the Chugoku mountains into the Seto Inland Sea and has a length of 86 km and a catchment area of 860 km^2. Seto Inland Sea is the biggest inland sea in Japan, and eutrophication of the seawater is an unsolved environmental problem. Levels of contamination in this river are the most serious in western Japan, and the downstream BOD concentration is more than twice the environmental standard value. Maximum altitude in the catchment is about 700 m, and this summit is in a midstream area. In this catchment, the landscape is generally hilly in the upstream area, mountainous in the middle part of the catchment, and a mixed hilly and flat terrain in the downstream area. The catchment is mainly underlain by granite, and 10% of the area is covered by alluvial sediment with a thickness of more than 10 m, which is located mainly on the flat lowland around the river downstream from the middle reaches.

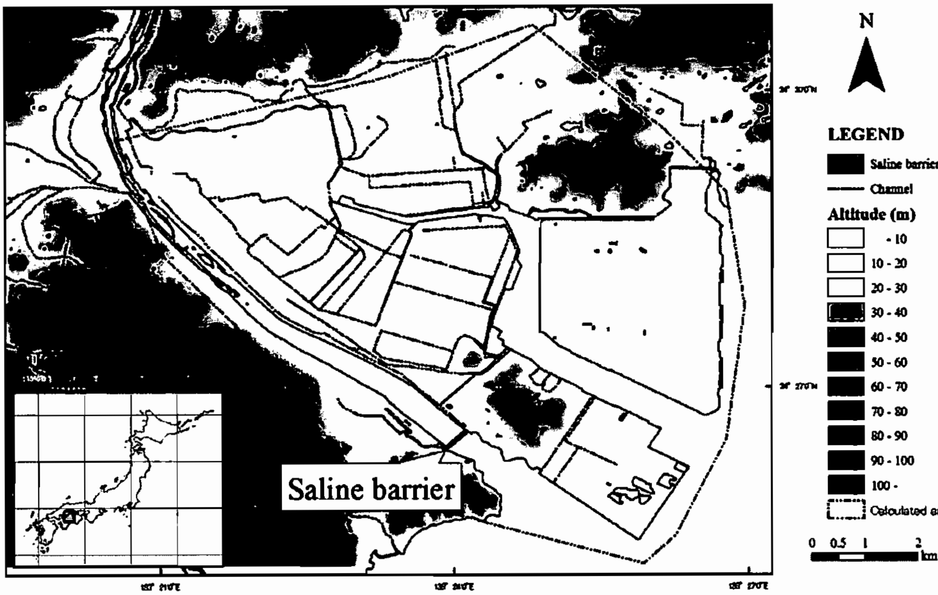

Fig. 1 Study area.

The study area is the river delta (Fig. 1), where the alluvial sediment reaches a maximum thickness of 40 m with an average value of 20 m (Onodera *et al.*, 2007). The average annual rainfall is about 1100 mm, which is two-thirds of the average for the whole of Japan. Monthly rainfall is highest in the rainy season during June and July. To prevent the inflow of the seawater to the river water and reserve the industrial water supply, the river-mouth dam was constructed in 1978. This freshwater pool has stopped the recharge of brackish water into the delta groundwater, whereas it has kept the recharge of freshwater into the brackish and marine sediment. There are many small channels in the delta area connected to the main river and its tributaries.

METHOD

Estimation of the current groundwater level in the coastal area

The current distribution of groundwater level elevation was estimated in the study area (calculated area in Fig. 1). The calculated area was separated with a 30 m × 30 m grid. Firstly, the relational equation was derived between the observed elevation of ground surface and groundwater level at the 22 dug wells located on the delta of the Ashida River (Fig. 1) in July 2006. Based on the equation, the elevation of groundwater level was calculated in each grid square.

Estimation of the coastal groundwater volume available for water resource

We tried to estimate the volume of groundwater which is available for the freshwater resource (no seawater intrusion) in the study area before and after construction of the river-mouth dam. The height of the freshwater was calculated by the Ghyben-Herzberg relation (Ghyben, 1889; Herzberg, 1901):

$$z = 40h_f$$

$$h = h_f + z$$

where z is the height of the saltwater column, or the depth below sea level to a point on the freshwater–seawater interface, h_f is the hydraulic head above sea level, and h is the height of the freshwater column. The h is calculated in each square of the 30-m mesh. Here, the area of possible seawater intrusion is defined by the mean seawater elevation at the high tide (3.87 m) and the low tide (0.33 m) of the study area. It was assumed that the seawater could intrude the area below 3.87 m before construction of the river-mouth dam and below 0.33 m after construction,.

However, approx. 30% of the study area is characterized by the outcrop area with more than 10 m above sea level. The effective porosity is defined as 0.01 in the outcrop area (>10 m), and 0.2 in the river plain (<10 m).

RESULTS AND DISCUSSION

Distribution of groundwater level

The estimated present groundwater level distribution is shown in Fig. 2. The mean groundwater level is approx. 2 to 5 m in the river plain area, and more than 10 m in the outcrop area. Near the river-mouth dam, it is relatively high, approx. 6 to 8 m. It suggests that the groundwater around the dam is affected by recharge from outcrop area with high groundwater level, and river water.

Variation in the groundwater resource by the construction of river-mouth dam

Figure 3 shows the distribution of "fresh" groundwater volume in the study area estimated in each 30 m grid before (a) or after (b) the river-mouth dam construction in 1978. The result suggests that the groundwater volume has increased after 1978 apparently by the shift of the brackish–freshwater boundary from the inland to the coastal line.

The estimated total volume of groundwater in the study area for the freshwater resource is shown in Table 1. The saline volume was estimated to be approx. twice that of before 1978. Onodera *et al.* (2007) estimated that the groundwater flux in the delta area is approx. 0.5 ton/year based on the hydrogeological condition and simulated groundwater flow. It is equal to about 5% of annual precipitation in the Ashida River basin. If the river discharge is assumed to be close to zero during the low-water season (July–September), groundwater with relatively constant discharge could supply 10 to 20% of the city's water supply. These results suggest that the coastal ground-

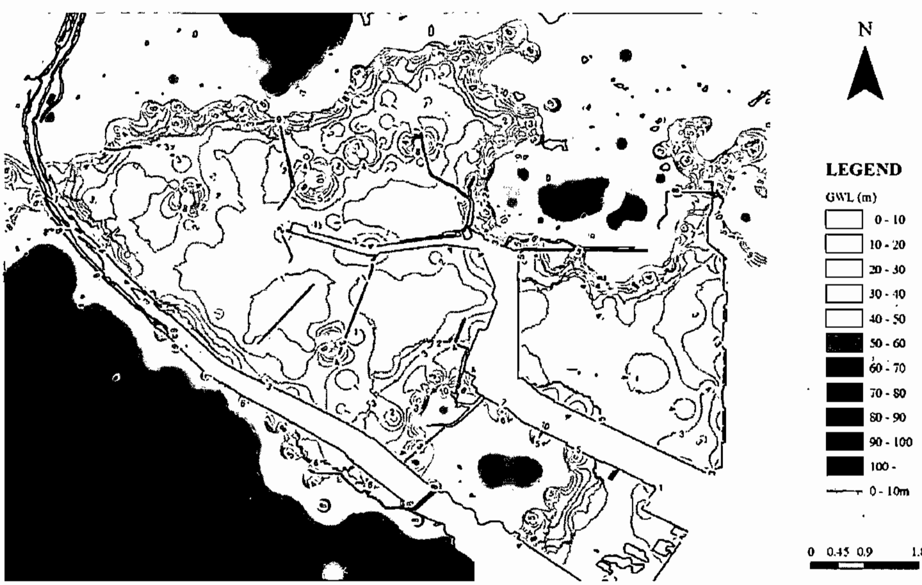

Fig. 2 Distribution of groundwater level in the study area.

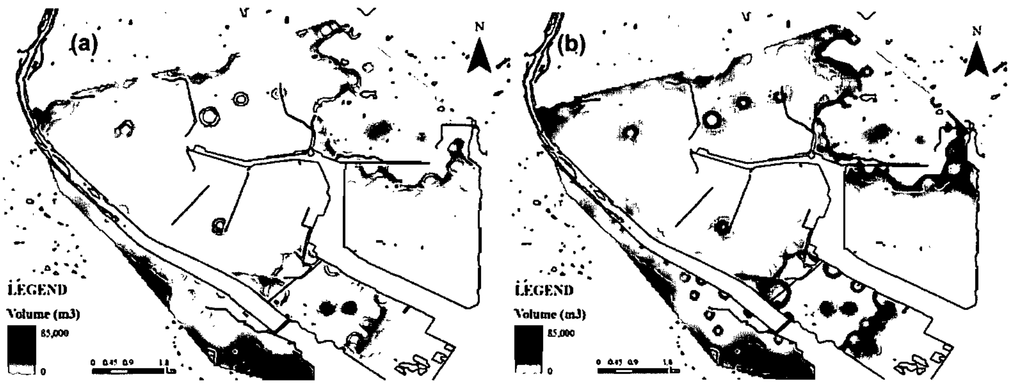

Fig. 3 Estimated distribution of the volume of "fresh" groundwater in the study area: (a) before 1978, (b) after 1978.

Table 1 The total volume of groundwater for freshwater resource in the study area before and after the construction of the river-mouth dam.

	Before 1978	After 1978
Total groundwater volume (10^8 m^3)	3.0	5.9

water is a valuable freshwater resource in the study area. However, we should mention that there is contamination of groundwater around the dam. As already stated, the river water behind the dam is polluted and it contains high concentrations of nutrients. The potential distribution suggests that river water recharges to groundwater around the dam. We should evaluate the quality as well as quantity of groundwater in the future.

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A laboratory model of the evolution of an island freshwater lens

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Abstract In this paper, a laboratory physical model was used to examine freshwater lens evolution. Groundwater recharge by rainfall was taken into account. The result shows that there is a direct relationship between freshwater lens thickness and cumulative rainfall infiltration. In the model, the lens thickness finally stabilized at 22 cm. The upconing movement process was also analysed for the case of groundwater exploitation. When pumping freshwater, the lower surface of the freshwater lens would rise gradually to form an upconing shape. Finally, the upconing moved into the pumping well which led to a breakdown of the freshwater lens. The result shows that the greater the exploitation rate the easier it is for the lens to break down.

Key words freshwater lens; laboratory simulation experiment; transition zone; up-coning

INTRODUCTION

The geology of the study area, the Xisha islands, is limestone formations of the Pleistocene period of the Quaternary. The permeability of the limestones, which have highly developed dissolution pores and solution cracks, is very high. Seawater intrusion can easily take place in the areas occupied by these formations. The age of the limestone formation is relatively younger on the upper areas of the island. The permeability of these upper parts, with inadequately developed dissolution pores and solution cracks, is poor. The conditions favour rainwater percolation and storage here and seawater intrusion is less problematic. The freshwater lens is formed because of the different densities between the underlying seawater and floating freshwater, on these islands. The freshwater lens is a precious freshwater resource available in these islands. For the purpose of judicious utilization of this precious resource, it is absolutely necessary to study its formation characteristics and factors influencing its safe utilization.

In order to exploit the freshwater lens safely and to meet the water demands of the island residents, Presley (2005) analysed the influence of drought on freshwater lens in depth on Marshall Islands in 1998. He pointed out that overexploitation will lead to the thickness of the freshwater lens decreasing quickly in the situation of recharge shortage (Presley, 2005). Schneider & Kruse (2005) used SEAWAT to simulate the influence of natural factors and human activities on the shape of freshwater lenses. They pointed out that factors activities such as the tide, human exploitation and human pollution will lead to the shape of the freshwater becoming gradually smaller (Schneider & Kruse, 2005). Jocson *et al.* used SWIG2D combining rainfall events information and water level information to simulate the influence of rainfall on freshwater lens in Mariana Island (Jocson *et al.*, 2002). Hocking & Forbes used the Green function to simulate the variation of upconing during exploitation (Hocking & Forbes, 2004). Numerical simulation software is used frequently to study freshwater lenses, but research studying the behaviour of freshwater lenses using physical experiments are few. Numerical simulation is usually based on field experiment data. Numerous conditions limit the use of field experiments for better understanding, and warrant laboratory simulation experiments which are used to efficiently complement field experiments. Here, a self-made physical model in the laboratory is used to simulate the evolution of a freshwater lens, and not only provides a theoretical foundation for exploiting the island's fresh groundwater resource, but also offers a reference for similar domestic and overseas research.

RESEARCH EQUIPMENT AND METHODS

Research equipment

Model tank The model tank is 2.0 m in length, 1.0 m in width, and 1.5 m in height. It is filled with sand to simulate the island conditions and 260 water quality monitoring tubes are installed in the front of the tank. These tubes are set in 13 rows, with 20 tubes for every row. All these tubes are used to measure water electrical conductivity value and to monitor the formation process of the freshwater lens. An extraction well is set in the centre of the model tank; it is 1.0 m in height and 0.02 m in diameter. The distance between the well bottom and model floor is 0.1 m. In order to exploit freshwater, a filter of 0.05 m in height is set on top of the extraction well. Ten water level observation wells 0.2 m length and 0.01 m diameter are set on top of the sand layer. To observe the position of the freshwater lens water table, they are all installed at a depth of 0.13 m in the sand.

Seawater tanks Two seawater tanks are created on both sides of the model tank; they are about 0.5 m in length, 1.0 m in width and 1.5 m in height. They are filled with "seawater" which is manufactured in the laboratory. The seawater tanks and the model tank are separated by a baffle perforated by holes of 3 mm diameter. Seawater flows through these holes into the model tank. A horizontal baffle which separates the seawater from the external environment is set 1.2 m high in the seawater tank. The interflow and surface runoff will discharge along the horizontal baffle to avoid changing the seawater concentration.

Rainfall equipment Rainfall equipment is located at the top of the model tank. Water is pumped by water-pump, and records the amount of water used for rainfall simulation through a water gauge.

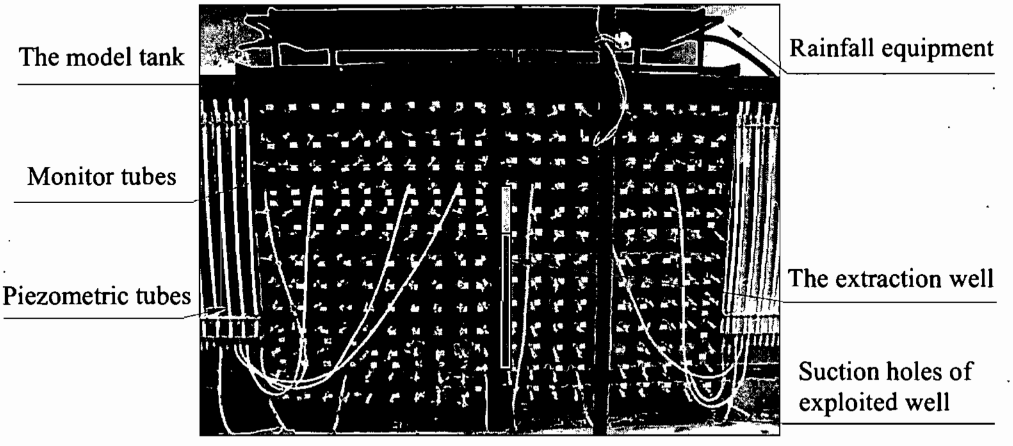

Fig. 1 Schematic diagram of the experimental apparatus.

Research methods

As the groundwater movement is expected to be faster in coarse sand and slower in fine sand, formation of a freshwater lens is complex; medium sand was used as the experimental medium to simulate freshwater lens evolution. In order to make the medium sand similar to the practical island situation, the sand sample was tamped every 5–10 cm as the tank was filled. Finally, the height of the sand sample maintained for the experiment reached 1.30 m. The hydrogeology parameters evaluated for the sand box model experiment are listed in Table 1. All these hydrogeology parameters provide basic data for the relevant numerical simulations.

The seawater used in this experiment is sodium chloride solution (NaCl) prepared in the laboratory. The solution density is 1.025 g/cm^3, and the chloride concentration is 18 g/L. Seawater flows through perforated baffle from the seawater tanks into the sand tank. A seawater level of 1.2 m is maintained throughout the experimentation process. Zhou *et al.* (1999) defined the a chloride concentration of 600 mg/L as the critical limit value between seawater and freshwater, which is the determining condition for a freshwater lens. The electrical conductivity value of critical chloride concentration is used as a parameter to judge the boundary of the freshwater lens during this experiment because of the linear relationship between the chloride concentration and electrical conductivity value (Chen & Li, 1996). The temperature varies at different times everyday, and the variation influences the conductivity values of seawater and freshwater heavily, so it is not accurate to use the absolute value of conductivity to characterize the shape of the freshwater lens. Because the relative difference between the conductivity values of the freshwater lens lower interface and the seawater always stabilizes at 0.2, so the relative difference used to characterize the lower interface of the freshwater lens. The contour value of 0.2 characterizes the shape of freshwater lens lower interface, and the contour value 1.0 characterizes the seawater surface, and the area between is a transitional zone.

When the seawater surface stays constant, rainfall events occur to simulate freshwater lens evolution. The rainfall intensity increases regularly, and the relationship between freshwater lens thickness and cumulative rainfall infiltration is analysed. Controlled exploitation of freshwater when the freshwater lens was stable was simulated and the up-coning formation observed.

Table 1 Hydrogeology parameters of the porous medium.

Parameters	d_{50} (mm)	n	K (m/d)	μ	D_L (cm^2/s)
Value	0.36	0.40	21.7	0.21	0.16

d_{50} :average particle size; n: porosity; k: hydraulic conductivity; μ: specific yield;
D_l: longitudinal dispersion coefficient

EXPERIMENT RESULTS ANALYSIS

Freshwater lens formation process analysis

The freshwater lens thicknesses were calculated under conditions of different cumulative rainfall infiltration. The freshwater lens thickness significantly increased in accordance with increasing rainfall infiltration at the beginning of the experiment, but the increase tendency became flatter after the lens thickness reached a certain value, and finally the thickness remained stable (Fig. 2). The freshwater lens thickness reached 22 cm and the freshwater resource quantity reached 323.4 L as the ultimate values in this experiment.

The samples were analysed after each rainfall events using the Surfer software to depict the contour of conductivity value, and the formation process of freshwater lens is observed (Fig. 3). It can be seen from Fig. 3 that the contour value 0.2 characterizes the freshwater lens lower interface,

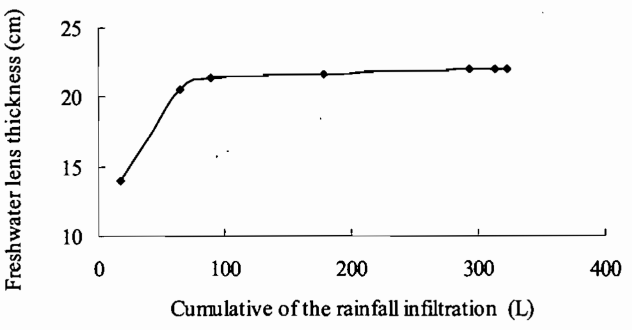

Fig. 2 The relationship between the freshwater lens thickness and cumulative of rainfall infiltration.

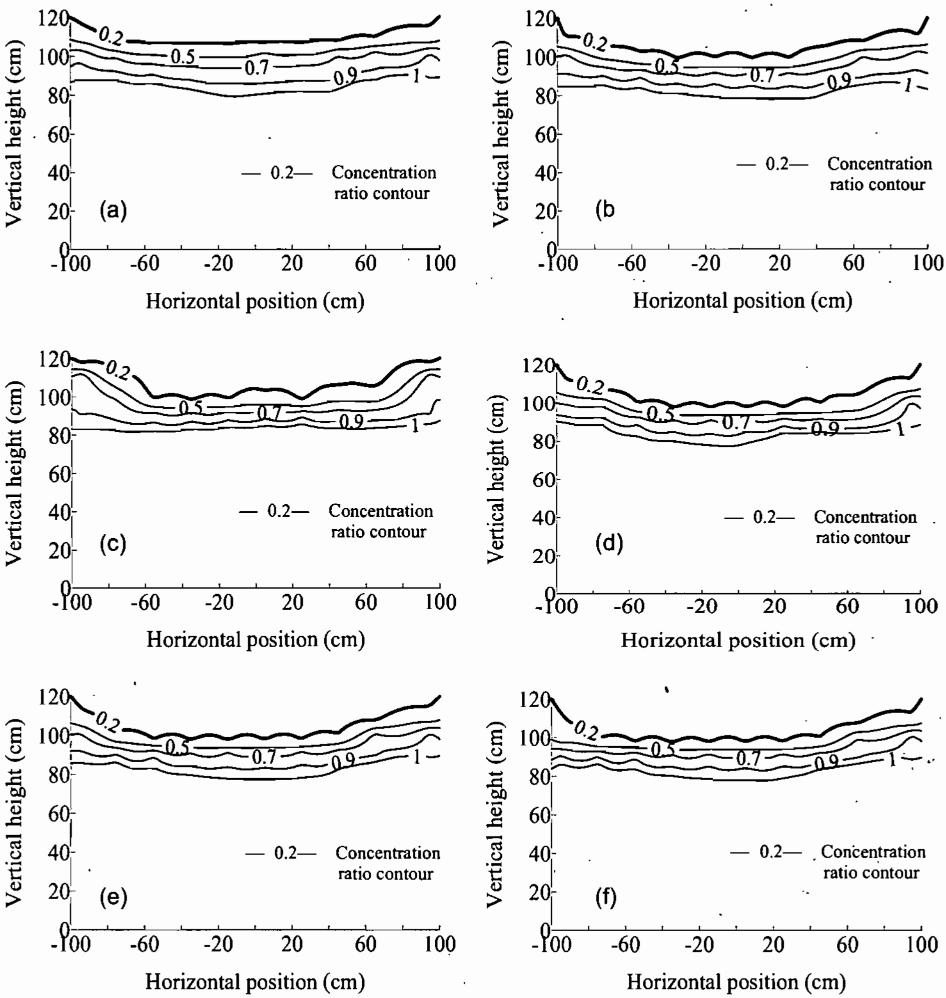

Fig. 3 The development process of the freshwater lens: (a) the freshwater lens lower interface after the first rainfall event; (b) after the second rainfall event; (c) after the third rainfall event; (d) after the fourth rainfall event; (e) after the fifth rainfall event; and (f) after the sixth rainfall event.

the contour value 1.0 characterizes the seawater surface, and the middle area is a transitional zone. In ideal conditions, the shape of the freshwater lens is axisymmetric relative to the island's centre because of the homogeneous and isotropic aquifer medium and consistent underlying surface situation (Schneider & Kruse, 2003). But the shape of freshwater lens in this experiment was a little asymmetric and irregular because of the heterogeneous aquifer medium and observation error.

It can be seen from Fig. 4 that the height of the freshwater lens saturated surface increases slightly through the whole rainfall process. The final height of the saturated surface was 0.13 cm above the seawater surface according to the limitation experiment scales (Liang *et al.*, 2006).

Upconing evolution law analysis

The density of seawater is larger than the density of fresh water, the transitional zone will rise gradually and form upconing when exploiting freshwater. There were two tests applied in the experiment. Each test included several exploitation methods. The exploitation intensity,

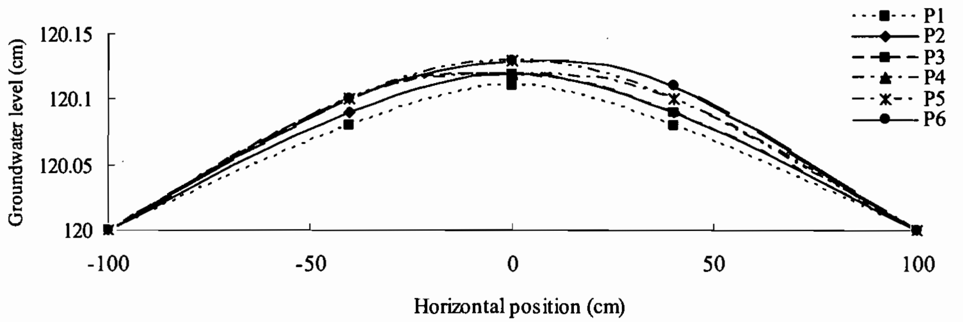

Fig. 4 Variation of the water table after each rainfall event.

Table 2 The statistics of pumping data.

Test	P (L/s)	T (s)	Q (L)	I (%)
1	0.012	600	7.20	2.23
2	0.030	300	9.04	2.80

P: exploitation intensity; T: exploitation duration; Q: total exploitation quantity; I: ratio between exploitation quantity and total freshwater resource quantity.

exploitation duration, and total exploitation quantity were different from each other. In the process of exploitation, the freshwater lens is considered to be broken when the electrical conductivity values of exploited water samples were higher than the critical electrical conductivity values of the chloride concentration of 600 mg/L (Zhen *et al.*, 2008), and the exploitation was stopped as soon as the electrical conductivity value reached the critical value to allow the freshwater lens to recover in about 16 hours by natural processes. After the natural recovery, rainfall continued until the freshwater lens was back to the original state, and the next exploitation begin. The two tests data are shown in Table 2.

The variations of the freshwater lens with different exploitation intensities are shown in Fig. 5; the greater exploitation intensity is, the faster the lower interface of the freshwater lens rises, and the easier freshwater lens is broken.

The recovery state of freshwater lens is observed 16 hours later. It can be seen from Fig. 6 that although the freshwater lens lower interface obviously recovers, the whole shape does not recover to the original state.

The decreasing values of freshwater lens thickness are analysed when it is broken. When the exploiting intensity reaches 1.20×10^{-5} m^3/s, the lens thickness declines by 6 cm, accounting for 27.3% of the initial thickness. While the exploiting intensity reaches 2.80×10^{-5} m^3/s, the lens

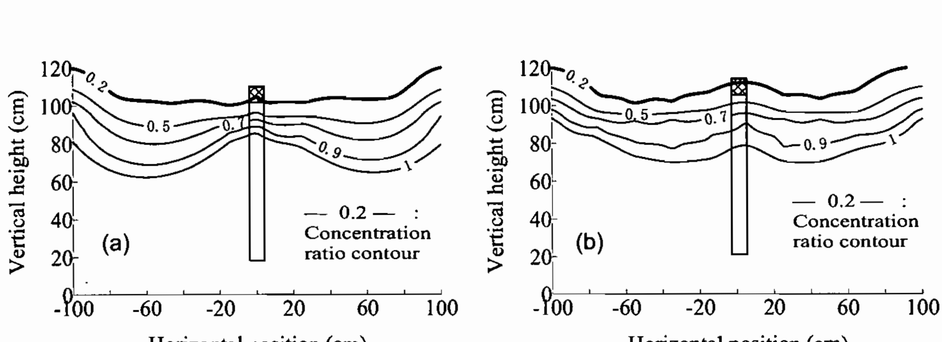

Fig. 5 The development of upconing under different pumping conditions: (a) K = 0.012 L/s, (b) K = 0.030 L/s.

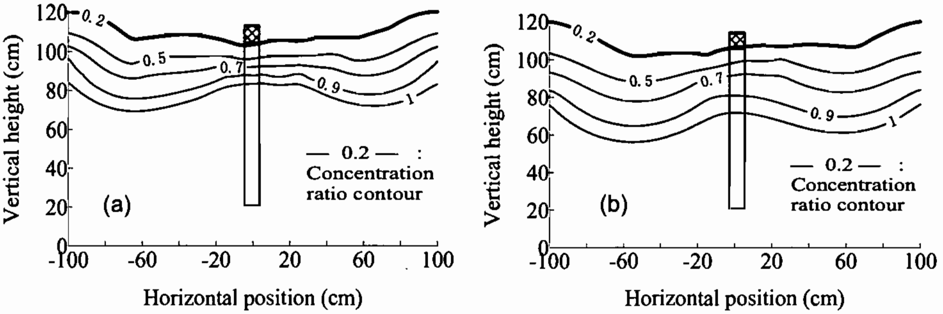

Fig. 6 The shape of freshwater lens after 16 hours natural recovery: (a) K = 0.012 L/s (b) K = 0.030 L/s.

Table 3 The variation of freshwater lens thickness under different pumping rates.

P	L_1(cm)	I (%)	L_2(cm)	I (%)
0.012	6.0	27.3	4.5	20.5
0.030	12.5	56.8	8.0	36.4

P: exploitation intensity; L_1: the decrease thickness of lens; L_2: the decrease thickness of lens after natural recovery; I: the ratio between the decrease thickness and the total lens thickness.

thickness declines by 12.5 cm, accounting for 56.8% of the initial thickness. The result shows that when the exploitation intensity is low the decrease values are smaller compared to the decrease when the exploitation intensity is heavy. After a period of natural recovery, the decrease values respectively reach 20.5% and 36.4% of the total freshwater lens thickness. It can be seen from Table 3 and Fig. 7 that the smaller the exploitation intensity is, the lower the ratio between the decrease thickness and the total lens thickness is, and the easier the lens recovers naturally. So in practical terms, on the precondition of satisfying the daily water demand, it is better to reduce the exploitation intensity for the freshwater sustainable utilization.

The decrease rate of the saturated surface centre is smaller than the upconing rise rate when the freshwater lens is exploited, but the variation range is not obvious. When the freshwater lens is broken, the decrease rate of the saturated surface centre is 1.0×10^{-4} cm/s, and the decrease value of the saturated surface is 0.045 cm under an exploitation intensity of 0.012 L/s; The decrease rate of the saturated surface centre is 1.7×10^{-4} cm/s, and the decrease value of saturated surface is

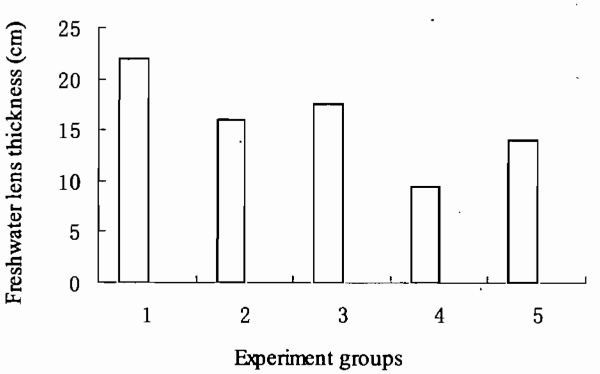

Fig. 7 The thickness of freshwater lens under different pumping conditions. Experiment 1: Unexploited; Experiment 2: Exploitation intensity 0.012 L/s, measured immediately after exploitation; Experiment 3: exploitation intensity 0.012 L/s, measured after 16 h considered as the natural recovery process; Experiment 4: exploitation intensity 0.030 L/s, measured immediately after exploitation; Experiment 5: exploitation intensity 0.030 L/s, measured after 16 h.

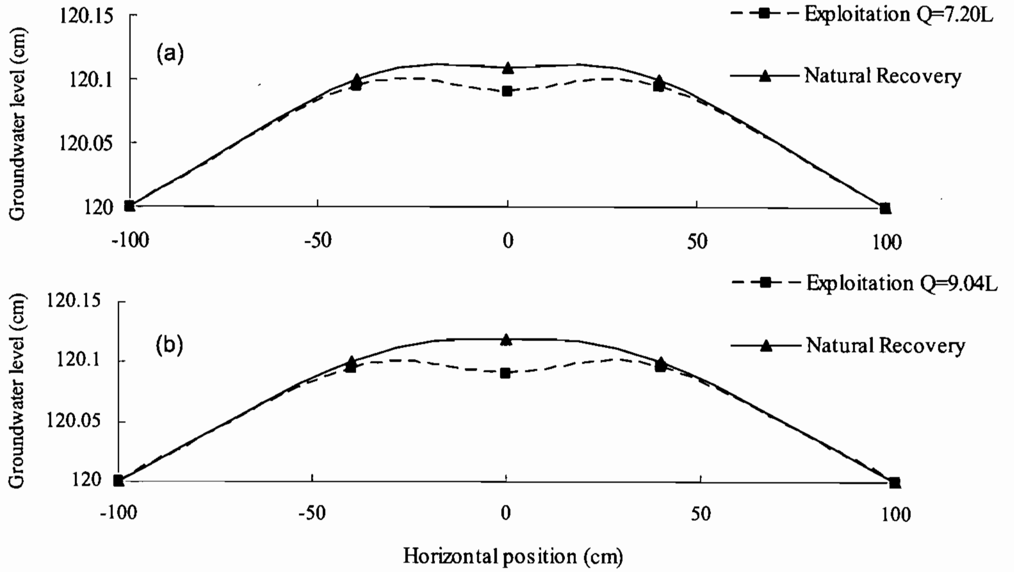

Fig. 8 The variation of water table under different pumping rates: (a) K = 0.012 L/s; (b) K = 0.030 L/s.

0.040 cm under the exploitation intensity of 0.030 L/s. It can be seen from Fig. 8 that the larger the exploitation intensity, the more easily the freshwater lens is broken.

CONCLUSION

1. The freshwater lens thickness increases significantly according to the increasing cumulative rainfall infiltration at the beginning of the experiment, and the increase tendency flattens after the thickness reaches a certain value. Finally the freshwater lens thickness attained stability at 22 cm thickness, and the freshwater resource quantity reached 323.4 L in this experiment.
2. During the process of exploitation, the lower surface of freshwater lens rises gradually, and finally forms upconing. The lens would eventually be broken with uninterrupted exploitation. When the exploiting intensity reaches 1.20×10^{-5} m^3/s, the lens thickness declines by 6 cm, 27.3% of the initial thickness. When the exploiting intensity reaches 3.00×10^{-5} m^3/s, the lens thickness declines 12.5 cm, 56.8% of the initial thickness. It shows that the larger the exploitation intensity is, the more easily the freshwater lens will be broken.
3. The water table varies gently. The highest point is 0.13 cm over the seawater level. When the freshwater lens is broken, the decrease value of the saturated surface is 0.045 cm under the exploitation intensity of 0.012 L/s, and 0.040 cm under the exploitation intensity of 0.030 L/s. It shows that the variation of saturated surface is not distinct under the exploitation intensity.

The results of this experiment not only provide basic data for numerical models, but also provide a scientific basis for rational exploitation and utilization of groundwater in an island region. But in this experiment some natural factors are not considered, e.g. evaporation, tides, which need further study.

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4 Floods and Droughts

Groundwater flooding in fractured permeable aquifers

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Abstract Current interest in groundwater flooding follows the events of 2000/01 when extreme long duration rainfall caused the water tables in Chalk catchments in South East England and Northern France to rise to unusually high levels. Individual rainfall events, and even monthly totals were unexceptional. By contrast, in July 2007, an extreme short-duration rainfall event (>90 mm in 12 hours) resulted in a rapid (<1 day) and marked rise in the water table. This behaviour is attributed to the dual permeability nature of the Chalk unsaturated zone (CUZ). The potential for individual extreme rainfall events to cause groundwater flooding is explored in this paper. Physical observations from the CUZ are combined with an analysis of the temporal water table response to rainfall. The rainfall characteristics (intensity and duration) appear to be a better predictor of rapid recharge than the activation of the fractures as determined by matric potential measurements.

Key words Chalk; groundwater flooding; groundwater recharge; unsaturated flow

INTRODUCTION

Groundwater flooding can be defined as the emergence of groundwater at or near the surface in areas where such water is rarely observed, or in much larger volumes than normally experienced (Morris *et al.*, 2007). Emergence may be relatively localised, via the activation of normally inactive springs, or more distributed when the water table intersects the ground surface over a wider area. In both cases the cause is the water table rising up to unusually high levels. The topic of groundwater flooding had not received much attention prior to the major flooding that occurred in the winter of 2000/01 in Chalk catchments in South East England (Finch *et al.*, 2004; Morris *et al.*, 2007) and Northern France (Pinault *et al.*, 2005). Total rainfall in England and Wales over the period September 2000 to April 2001 was 166% of the long-term average (Marsh & Dale, 2002), and in the Thames region (in South East England) the period October 2000 to April 2001 received the highest total seven month rainfall in Environment Agency records, which begin in 1885 (Morris *et al.*, 2007). However, rainfalls within this period over shorter time scales (daily to monthly) were unexceptional. Therefore, the 2000/01 flood is attributed to exceptional long duration rainfall during the winter recharge period which led to exceptional groundwater levels in the Chalk aquifers.

In this research we focus on understanding of the flooding mechanisms, and specifically try to address the question of how the water table responds to rainfall. Key to this question is: how is water transmitted through the Chalk unsaturated zone (CUZ), which is often very deep (in excess of 100 m)? Chalk is a dual porosity/dual permeability micro-porous, fractured limestone aquifer (Price *et al.*, 1993) that constitutes a major water supply in England. Extensive past research has looked at the flow processes in the CUZ in the context of quantifying recharge and contaminant transport (particularly nitrates) to the aquifers. A review of this research is given by Ireson *et al.* (2009). Key contributions are from Smith *et al.* (1970) who looked at the first tracer study, Wellings & Bell (1980) who first applied soil physics techniques, and Price *et al.* (2000) who proposed the now widely adopted conceptual model for flow and storage in the CUZ. The challenge has been to quantify the role of the matrix and fractures, and some progress has been made by the application of numerical models by Mathias *et al.* (2006), Brouyère *et al.* (2006), van den Daele *et al.* (2007) and Ireson *et al.* (2009). It is understood that recharge is predominantly through the Chalk matrix, which is normally saturated by capillarity, but that under certain rainfall conditions (though these conditions have not been well defined) fast recharge pathways through the fractures may be activated (Gallagher *et al.*, 2009; Ireson *et al.*, 2009). Fractures are typically assumed to become active when the matric potential exceeds –0.5 m, an air entry pressure related to a fracture aperture of 30 μm (Wellings, 1984). In reality, a distribution of fracture apertures exists, hence smaller, less conductive fractures will start to transmit water at lower matric potentials than the larger, more conductive fractures. Furthermore, it has been postulated that storage and flow of water may occur in and along surface discontinuities of the walls of partially saturated fractures (Price *et al.*, 2000). Therefore, the threshold of –0.5 m should be considered as indicative of fracture activation, and as the matric potential increases above this value, hydraulic conductivity and flow through the fractures is likely to increase dramatically. Furthermore, flow in the matrix will continue even when the fractures are activated, and for potentials close to the activation threshold may still be significant or even dominant.

Another important way to gain insights into the flow processes in the CUZ is by examining the water table response to rainfall. Headworth *et al.* (1972) looked at lag times between rainfall events and the associated rise in the water table in chalk aquifers in Hampshire, UK. Lags of the order of tens of days were identified, and recharge was attributed to the piston displacement mechanism, via the Chalk matrix. More recently, Lee *et al.* (2006) looked at the cross-correlation between daily rainfall and water table response (i.e. the rate of change of the water table with time) at various locations in South East England. This analysis identified two modes of response: a very rapid response (within 1 day) associated with fracture flow, and a more delayed response (up to 4 weeks) associated with piston displacement. The rapid recharge events were found to occur during or after the winter of 2000/01, and in one case involved a water table at around 60 m below ground level (b.g.l.). However, this conclusion was based solely on correlation (which does not necessarily imply causation), and no physical evidence of fracture activation was available.

In this study, we combine physical observations from the CUZ with an analysis of water table response to rainfall events, to identify different modes of recharge, with a particular focus on responses likely to cause flooding.

FIELD SITE AND INSTRUMENTATION

During the winter of 2000/01 extensive groundwater flooding occurred in the upper part of the Pang valley (Berkshire, South East England), the spatial distribution of which was mapped by Finch *et al.* (2004). In this paper we make use of an instrumented borehole at East Ilsley (EI) (SU 4996 8114, Fig. 1), described in Gallagher *et al.* (2009) and located in the Pang valley where

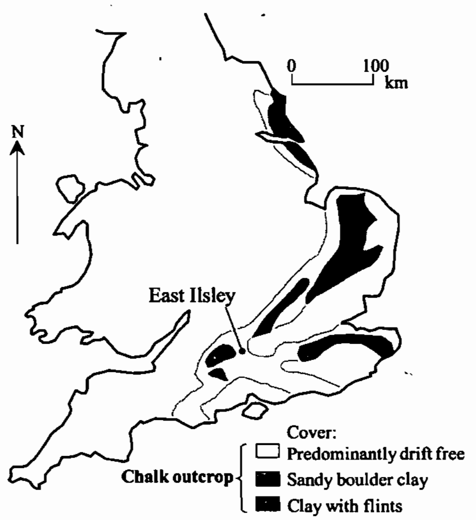

Fig. 1 The Chalk outcrop (adapted from Gardner *et al.*, 1990) showing the location of the East Ilsley field site in the Berkshire Downs.

flooding occurred. The borehole was installed in 2005, monitoring began in October 2005, and here we look at the record up to September 2007. The borehole is uncased between 5 and 35 m b.g.l., and the water table fluctuates within the range of 14–28 m b.g.l.. As well as a borehole piezometer, 8 deep jacking tensiometers (Cooper, 2006) were installed at depths of 13, 17, 18, 19, 21, 22, 23 and 24 m b.g.l.. Clay-rich marl horizons and brittle, fractured hard-chalks punctuate the Chalk at East Ilsley. In their study of borehole geophysics and matric potential data from the site, Gallagher *et al.* (2009) suggest that both the Chalk Rock (a highly fractured hard-chalk) and a number of marl horizons, such as that present at 18 m b.g.l., may influence recharge processes in the CUZ during wet periods. Although it is considered that their effect is relatively minor, particularly when compared with another similarly instrumented site (in the South Downs) where the marls are more frequent and prominent.

Rainfall and actual evaporation (calculated as the residual of the surface energy balance using measurements of sensible heat flux by eddy correlation, net radiation and soil heat flux over grass) data measured at Warren Farm (14 km to the west) are used in this analysis.

RECHARGE PROCESSES

Figure 2 shows a comparison of the cumulative effective rainfall (CER, the summation of daily rainfall minus daily actual evaporation) with the water table at EI. During the two year record, there are three periods when the water table rises significantly (i.e. by more than 1 m), each with a distinct response. The time delay between changes in slope of CER and water table was used to identify these different responses. This is shown in Fig. 2 for peak to peak and trough to trough delay times. For the recharge period in 2005/06 these lags are greater than 100 days. This was a drought period, characterised by low CER and a low antecedent water table. For the main recharge period in 2006/07 the amount of effective rainfall markedly increased, and the lag times reduced. The onset of recharge lagged by 72 days, but this was reduced to 22 days by the time the water table reached its peak – a figure consistent with those of Headworth *et al.* (1972) and Lee *et al.* (2006) for the piston displacement mechanism. The third period of interest follows the extreme rainfall event in the summer of 2007, when a large water table response occurred within one day. In the subsequent analysis we treat each of these responses separately.

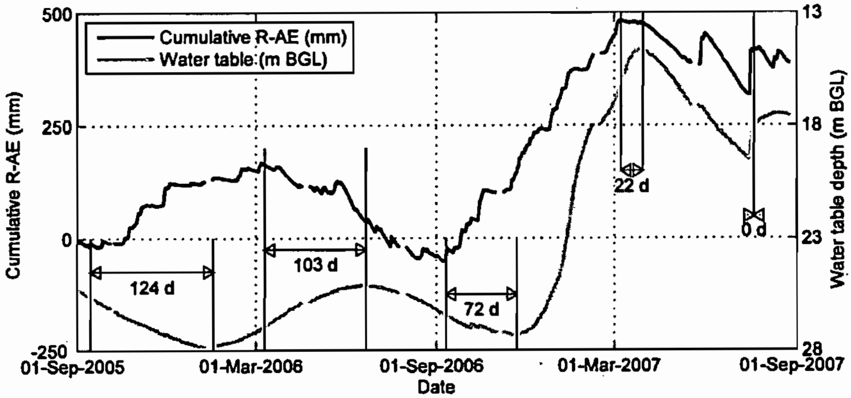

Fig. 2 Water table response to cumulative effective rainfall at East Ilsley.

Physical observations of the CUZ

The antecedent conditions for the recharge periods in 2005/06 and 2006/07 appear similar, as seen from the water table elevation in September 2005 and September 2006, and from the earliest head profiles in Figs 3 and 4. During the first recharge period, the water table remained below all of the deep jacking tensiometers, which consequently gave consistently negative pressure readings. The wetting of the profile was gradual and uniform, as shown in Fig. 3. There is some evidence of the marl band at 18 m depth perturbing the head profile at certain points in time, but on the whole, there is a steady downward head gradient, which increases over time, and flow is exclusively via the matrix.

In 2006/07, as a result of the greater CER, the water table rises up more rapidly, eventually going above most of the tensiometers (which give positive hydraulic heads consistent with that of the piezometer). Figures 2, 3 and 4 suggest that in January 2007 and June 2006 the CUZ was in an approximately identical state. However, in June 2006 this was the wettest point that year, whilst in January 2007 the onset of recharge had only just begun. In 2007, as the water table progressively moved up, passing each of the tensiometers in turn, the vertical hydraulic head gradient in the saturated zone was essentially zero (below 18 m b.g.l.) or small (above 18 m b.g.l.), implying that flow here is predominantly lateral, as would be expected given the high transmissivity of the Chalk aquifer. The discontinuity at 18 m b.g.l. may be associated with the marl band, but we can only speculate about this with the data currently available. From January until the end of the entire recharge period, matric potential readings suggest that fracture flow was initiated throughout the

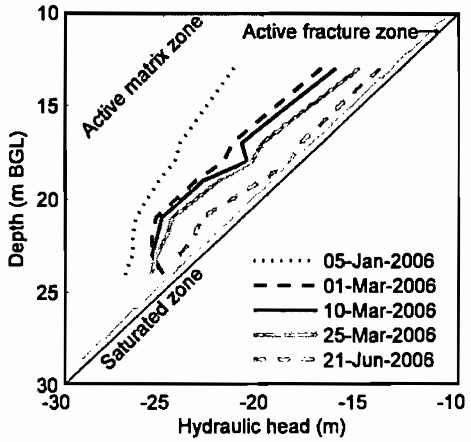

Fig. 3 Hydraulic head profiles derived from tensiometer data during the recharge period in 2005/06.

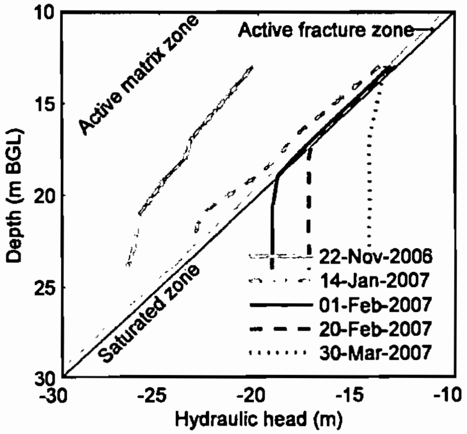

Fig. 4 Hydraulic head profiles derived from tensiometer data during the recharge period in 2006/07.

entire measured profile, as seen clearly on 1 February 2007 in Fig. 4. This was unexpected, since the water table response was on a timescale that has been associated with recharge through the matrix.

For the final recharge event, in the summer of 2007, the tensiometer at 13 m b.g.l. was not functioning correctly, so we only have the water table response (as given by the piezometer and tensiometers below 19 m b.g.l.), and matric potentials at 17 and 18 m b.g.l. The water table response is shown in Fig. 5. There was a collection of moderately large rainfall events in late May that caused a small, rapid perturbation at the water table. There then followed a gap of about 50 days when little or no rainfall was recorded. On 20 July, a total of 90 mm of rain fell within about 12 hours, and the water table rose dramatically on the same day. Therefore we can be confident that the water table rise was caused by this rainfall event.

In Fig. 6 the recharge event is shown in finer detail, where all data have a 15 min resolution. The water table starts to respond 13 hours after the very start of the rainfall event, and just 75 min after the centroid of the rainfall event. The tensiometers at 17 and 18 m b.g.l. respond at approximately the same time as each other, but not until 8 hours after the water table response. Matric potentials measured at 17 and 18 m b.g.l. indicate that the fractures were inactive before the

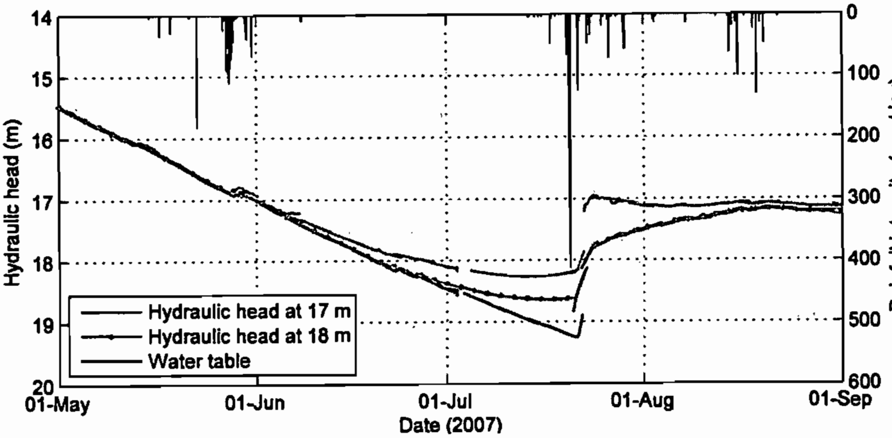

Fig. 5 The summer 2007 recharge event.

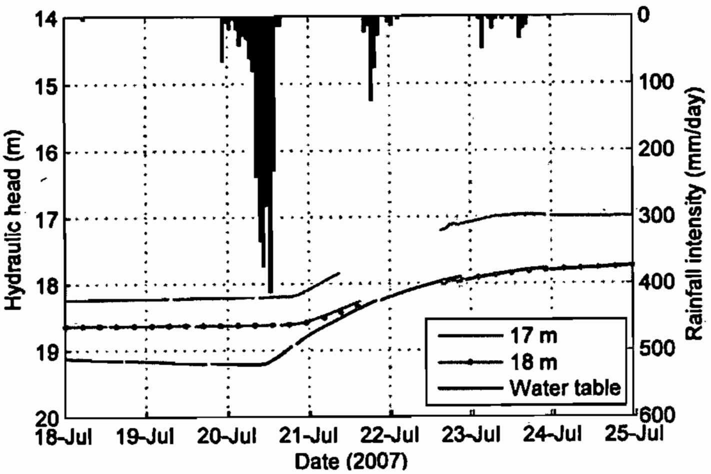

Fig. 6 The summer 2007 recharge event in fine detail.

response to the event. This suggests that rapid recharge via the fractures bypassed the tensiometers. This could be in the form of localised recharge via a discrete number of large fractures which are normally inactive, or it could be via the ubiquitous fracture network, if the fractures and matrix are not in pressure equilibrium. We also cannot rule out the possibility that the borehole itself acts as a pathway through the CUZ.

Rainfall characteristics causing rapid fracture recharge

The above analysis suggests that we have three distinct types of recharge evident in the EI data set: slow matrix recharge during a drought; combined matrix-fracture recharge, at a rate consistent with previous findings; very rapid (<1 day) recharge through the fractures. The 2000/01 flood was probably caused by the second type of response. However, it is clear from the third response that there is another potential mechanism for groundwater flooding, which should not be overlooked: water table response to an extreme rainfall event. Fracture activation may not be the most useful indicator of this rapid recharge (and is anyway difficult to measure), so we now focus on rainfall characteristics. Figure 5 shows that some rainfall events cause an obvious, visible perturbation of the water table within 1 day, whilst others do not. We define a rainfall event as a cluster of non-zero rainfall measurements (on an hourly time step) containing no gaps of longer than 6 hours. In total 334 such events were identified in the two year record, with durations from 1 hour to >2 days and mean intensities from 1.2 to 180 mm/d. By looking carefully through the data set, it was possible to identify 11 rainfall events that caused an obvious, rapid perturbation of the water table (two prominent examples are shown in Fig. 5). In Fig. 7 the mean event intensity is plotted against event duration for all events, and the 11 events which caused a perturbation are also shown. For these 11 events the antecedent depth to the water table covered the entire range of water table fluctuation, hence the rapid response was not caused by the proximity of the water table to the ground surface, but was due to the rainfall intensity/duration characteristics. Six of the 11 events are clear outliers in Fig. 7, whereas the remaining five are close to the edge of the overall cluster. This allows us to postulate three distinct regions on this plot, labelled A, B and C. Rapid recharge is very unlikely to be caused by rainfall intensity/duration combinations from region A; it may or may not be caused by those is region B; and it is very likely to be caused by those in region C. Whilst the event on 20 July 2007 caused the most severe response, it is neither the most intensive, nor the longest duration event in the record. However, it is the furthest point from the

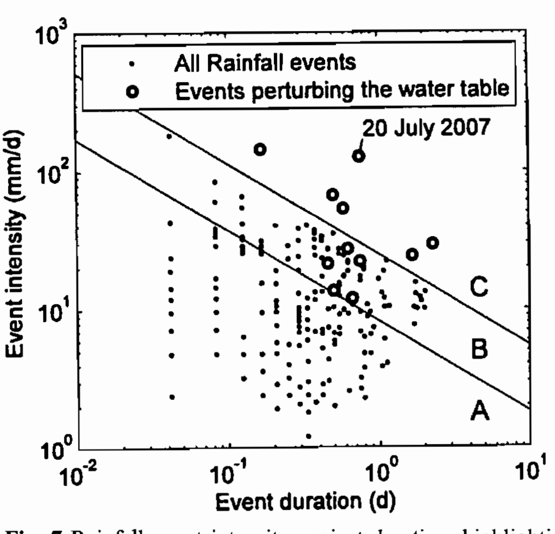

Fig. 7 Rainfall event intensity against duration, highlighting the events which caused a rapid perturbation of the water table.

threshold of region C, and in that respect can be identified as the most extreme rainfall event. These observations suggest that this is a potentially useful method for identifying the likelihood of rapid recharge from rainfall characteristics, although more data would be required to quantify a reliable threshold with wider application than this one location.

CONCLUSIONS

In this paper physical observations of the CUZ were combined with detailed analysis of the rainfall–water table response to gain insights into recharge processes. Whilst no groundwater flooding occurred at East Ilsley during the period studied, important insights can be made and three distinct types of recharge are postulated.

Ireson *et al.* (2006) looked at matric potential measurements in the near surface (0.2–1.2 m b.g.l.) at a nearby field site (West Ilsley), and observed significant fracture activation (i.e. matric potentials > –0.5 m) during the recharge period of 2003/04. In this study deeper measurements were available (13–24 m b.g.l.), and there was evidence of significant fracture activation in the CUZ during the recharge period in 2006/07. Similar recharge lags (24 days in 2003/04 reported by Ireson *et al.*, 2006; 22 days in 2006/07 reported in this study) were noted for these recharge periods, albeit for different boreholes. Neither of these periods resulted in an exceptional water table response. This suggests that fracture activation (as indicated by the matric potential exceeding –0.5 m) may not necessarily imply a rapid recharge response.

Rapid recharge events are reasonably common (11 events in 2 years), but normally result in minor changes in the water table. However, this analysis suggests that extreme rainfall events may cause a significant rise in the water table, irrespective of the antecedent conditions. If the rainfall event of 20 July 2007 had occurred a few months earlier, it is highly likely that groundwater flooding would have occurred.

Early warning systems have been proposed to forecast the risk of groundwater flooding a number of months into the future, assuming a linear response (Adams *et al.*, 2009). There is a need for groundwater flood risk assessment tools which are able to capture the different types of rainfall response that have been observed in this study, in particular those associated with extreme rainfall events. Numerical models of flow in the CUZ are improving, but are probably not going to be feasible for use within large-scale groundwater models being developed for risk assessment purposes, since they are very computationally expensive, and have a large number of parameters that need to be identified. It will therefore be necessary to develop simple recharge models, capable of transmitting these different types of recharge. The classification of different recharge responses by rainfall intensity/duration characteristics may therefore be an attractive way forward in addressing this need.

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Assessing the influence of drought on vegetation vigour within the Laohahe catchment, China, by NDVI and PDSI indices

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Abstract The objective of this study is to assess the influence of drought on vegetation vigour. The correlation analysis based on different vegetation type was conducted between monthly NDVI and Palmer Drought Severity Index (PDSI) during the growing season within the Laohahe catchment. It was found that NDVI had good correlation with the PDSI, especially for shrubs and grasses. The correlation between NDVI and PDSI varies significantly from one month to another. The influence of drought on vegetation vigour is stronger in the first half of the growing season before the vegetation reaches its peak greenness. In order to take the seasonal effect into consideration, a regression model with seasonal dummy variables was used to simulate the relationship between the NDVI and PDSI. The results showed that the NDVI–PDSI relationship was significant ($\alpha = 0.05$), and that NDVI was an effective indicator to monitor and detect droughts if seasonal timing was taken into account.

Key words drought; PDSI; NDVI; correlation analysis; regression analysis

INTRODUCTION

Drought is a complex natural hazard that has economic, environmental and social consequences worldwide. In the agricultural sector, drought is one of the dominant causes of crop loss in China. According to statistics, the mean annual area of drought was about 21.6 million ha during the period 1949–1999, accounting for 60% of the total area of meteorological disasters, and crop loss was about 10 billion kg in each year (Yuan & Zhou, 2004). The complex drought phenomenon can be represented by a drought index, which allows scientists to quantify climate anomalies in terms of intensity, duration, and spatial extent (Wilhite *et al.*, 2000).

Palmer (1965) developed the PDSI (Palmer Drought Severity Index) based on the concept of climatically appropriate for existing condition (CAFEC). The effects of water deficit and duration on drought severity are considered by the PDSI comprehensively. The main items of the water budget in the PDSI include precipitation, evapotranspiration, runoff and soil water storage, which can be calculated by the water balance method or with a hydrological model. In addition, the influence of water deficit in the prior period on the later period is also considered by the PDSI. All the above characteristics mean the PDSI has explicit physical meaning and spatial and temporal comparability. Since the theory, advantages, and calculation methods of the PDSI were introduced by Fan & Zheng (1984), many Chinese meteorologists have modified the PDSI model using the meteorological data and established the meteorological drought model of China (Liu *et al.*, 2004). Nowadays, the PDSI is widely used to represent the genesis, severity, onset and end, and duration of drought in China.

In addition to drought indices, satellite sensor data have been playing an increasingly important role in monitoring drought-related vegetation condition. Because the NDVI is highly correlated to green-leaf density and vigour, it is considered to be a proxy for the status of above-ground biomass at the landscape level, and is widely used to monitor and evaluate terrestrial vegetation vigour (Bannari *et al.*, 1995; Lei & Albert, 2003). Because of the close relationship between vegetation vigour and available soil moisture, especially in arid and semi-arid areas, the AVHRR-derived NDVI has been used to evaluate drought condition by directly comparing it to precipitation or drought indices (Tucker, 1989; Gutman, 1990). Although a number of previous studies have indicated a strong relationship between NDVI and precipitation, air temperature or SPI (Li *et al.*, 2002; Lei & Albert, 2003; Wang *et al.*, 2003; Liu & Xu, 2007), this relationship is far more complex than what can be represented by simple linear correlation between two variables. It has been noted that the impact of water availability on vegetation changes considerably within different phenological periods of vegetation growth cycles.

The objective of this paper is to assess the influence of drought on vegetation vigour within the Laohahe catchment by analysis and simulation of monthly NDVI and PDSI relationships based on forest, shrub, grass, and crop vegetation types.

STUDY AREA AND DATA PREPARATION

Study area

The Laohahe catchment, with a total drainage area of 18 599 km^2, is controlled by Xiaoheyan hydrological station (42°26′N, 119°34′E) and is situated in a semi-arid region in northern China. Elevation within the catchment ranges from 444 m to 1836 m a.m.s.l., with elevation declining from southwest toward northeast. Forest land, shrub land, grassland and crop land make up most of the vegetation cover in the catchment. There are 52 raingauges, and four meteorological stations (Fig. 1) and the recorded data are available from 1960 to 2005. Annual average maximum (minimum) temperature is 14°C (2°C) ranging from –4°C (–16°C) in January to 29°C (18°C) in July. The average annual precipitation is ~451 mm, and the spatial and temporal distribution of precipitation is uneven.

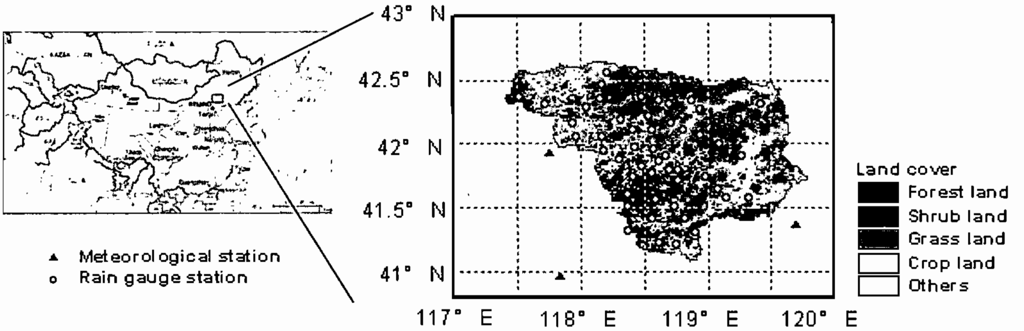

Fig. 1 Location of the Laohahe catchment and distribution of land cover, meteorological stations and rain gauge stations within the catchment.

Data preparation

Topography The digital elevation model (DEM) data within 40.9°–42.9°N and 117.2°–120°E at a spatial resolution of 30 s × 30 s were obtained from the Global Land One-kilometer Base Elevation database.

Land cover data The University of Maryland's 1-km global land cover data was used to represent the vegetation cover over the study area. The main vegetation types in the study area include forest, shrub, grass and crop, the area percentages of which are 18.1%, 5.6%, 40.2% and 35.6%, respectively (Fig. 1).

NDVI The NOAA-AVHRR NDVI data set is available monthly for the globe at an interval of 8 km, covering the period from July 1981 to September 2001. The NDVI data within the catchment were clipped according to the basin boundary and transferred into the geographical projection at 30 s resolution.

Meteorological data The required meteorological data include daily mean, maximum and minimum air temperature; air vapour pressure; wind velocity; daylight duration; and precipitation. In this study, daily precipitation data were derived from 52 raingauge stations spread across the catchment, and other meteorological data were derived from four meteorological stations around the catchment. All meteorological data were interpolated over the whole study area using the inverse distance square method. Based on the DEM, meteorological variables such as air temperature, air vapour pressure and wind velocity were topographically corrected with the empirical relationships (Ren *et al.*, 2009) during the spatial interpolation.

METHODOLOGY

Computation of the PDSI

Monthly PDSI values have been calculated for each grid cell at a spatial resolution of 30 s over the Laohahe catchment, using the above meteorological data, for the period 1960–2005. The basic concepts and steps of computation are as follows (Szep *et al.*, 2005).

Step 1: Potential evapotranspiration In this study, the Penman-Monteith equation is used to calculate the potential evapotranspiration, a key variable of the water balance and also of the PDSI computation procedure (Liu *et al.*, 2004).

$$ET_0 = \left[0.408\Delta(R_n - G) + \gamma\frac{900}{T+273}U_2(e_s - e_a)\right] \Big/ [\Delta + \gamma(1 + 0.34U_2)] \quad (1)$$

where ET_0 is the reference evapotranspiration (mm d^{-1}); R_n is the net radiation absorbed by land surface (MJ m^{-2} d^{-1}); G is the soil heat flux (MJ m^{-2} d^{-1}); T is daily mean temperature(°C); e_s and e_a are the saturation and actual vapour pressures (kPa), respectively; U_2 is the wind speed (m s^{-1}) at 2 m height; γ is the psychrometric constant (kPa °C^{-1}); and Δ is the first-order derivative of saturation vapour pressure with temperature (kPa °C^{-1}).

Step 2: Hydrological accounting A simple water balance model was used to calculate the hydrological variables in the PDSI model. The soil is divided into two arbitrary layers. The upper layer is assumed to contain 40 mm of available moisture at field capacity, and the lower layer is 200 mm in the study area. The loss from the underlying layer depends on the initial moisture content, as well as on the computed Potential Evapotranspiration (PE) and the Available Water Capacity (AWC) of the soil system. Runoff is assumed to occur, if and only if, both layers reach their combined moisture capacity, AWC.

Step 3: Climatic coefficients This is accomplished by simulating the water balance for the period of available weather records. Monthly coefficients are computed as proportions between climatic averages of actual *vs* potential values of evaporation, recharge, runoff and loss, respectively.

Step 4: CAFEC values The derived coefficients are used to determine the amount of precipitation *(I)* required for the Climatically Appropriate For Existing Conditions (CAFEC), i.e. "normal" weather during each individual month.

Step 5: Moisture anomaly index Difference between the actual and CAFEC precipitation is an indicator of water deficiency or surplus in that month, expressed as $D = P\text{-}I$. These departures are converted into indices of moisture anomaly as $Z = K{\cdot}D$, where K is a weighting factor, also accounting for spatial variability of the departures (D).

Step 6: Drought severity In the final step the Z-index time series are analyzed to develop criteria for the beginning and ending of drought periods and an empirical formula for determining drought severity. In this study, the empirical formula is (Liu *et al.*, 2004):

$$x_i = 0.9331x_{i-1} + z_i / 125.99 \quad (2)$$

where z_i is the moisture anomaly index and x_i is the PDSI for the i-th month. The equation indicates that PDSI of a given month strongly depends on its value in the previous months and on the moisture anomaly of the actual month.

Time series of monthly NDVI and PDSI

Based on the grid cell at a spatial resolution of 30 s, the monthly NDVI and PDSI of all pixels in each vegetation cover type within the study area were determined. The time series of monthly

NDVI and PDSI for each vegetation type were calculated by averaging the values of the pixels with the same vegetation type. Only the data during the growing season (April–October) were used for the analysis.

Correlation and regression analysis

Pearson correlation analyses were conducted for the NDVI and PDSI. Because the relationship between vegetation vigour and water availability varies within a growing season, each month was analysed separately. The correlation coefficients and their p-values were obtained. The p-value of the correlation coefficient is the probability of rejecting the null hypothesis that there is no correlation between two variables.

Because the linear relationships between the NDVI and PDSI are significantly different for different seasonal periods, seasonal dummy variables were employed in the regression model (Doran, 1989). In this study, the dummy variables were a set of seven levels assigned to the seven months of the growing season and used to account for the effect of the "month" on NDVI. The regression model containing seasonal dummy variables are expressed as:

$$\begin{aligned} NDVI = &\beta_0 + \beta_1(PDSI) + \beta_2 D_1 + \beta_3 D_2 + \beta_4 D_3 + \beta_5 D_4 + \beta_6 D_5 + \beta_7 D_6 + \beta_8 D_1(PDSI) \\ &+ \beta_9 D_2(PDSI) + \beta_{10} D_3(PDSI) + \beta_{11} D_4(PDSI) + \beta_{12} D_5(PDSI) + \beta_{13} D_6(PDSI) + \varepsilon \end{aligned} \tag{3}$$

where D_1–D_6 are the dummy variables, β_0, β_1, ... β_{13}, are the regression coefficients, and ε is random error. Table 1 shows the assigned binary values of dummy variables D_1–D_6.

Table 1 Binary value of dummy variables.

Condition	D_1	D_2	D_3	D_4	D_5	D_6
If the month is April	0	0	0	0	0	0
If the month is May	1	0	0	0	0	0
If the month is June	0	1	0	0	0	0
If the month is July	0	0	1	0	0	0
If the month is August	0	0	0	1	0	0
If the month is September	0	0	0	0	1	0
If the month is October	0	0	0	0	0	1

RESULTS AND DISCUSSION

Covariation of NDVI and PDSI time series

The relationship between vegetation vigour and moisture availability was clarified by analysing the covariation of NDVI and PDSI time series with the scatter plots and Pearson correlation analysis. Correlation coefficients and p-values of the NDVI and PDSI for different vegetation type during the growing season months are shown in Table 2.

The correlation coefficients vary by month and vegetation type. For example, in September and October there are no significant correlations between the NDVI and PDSI, while the significant correlations are found in the other months of the growing season for every vegetation type. The correlation coefficients between the NDVI and PDSI for shrub and grass are higher during the growing season, while the correlation coefficient for forest is lowest, and the correlation coefficient for crop is moderate among the main vegetation types. This demonstrates that the influence of drought on shrub and grass is stronger while forest has less response to the drought in the Laohahe catchment. The different influences of drought on vegetation depends on two factors: (a) The plant growing environment: the forest is mostly distributed in the upstream high elevation region within the Laohahe catchment, where precipitation is relatively ample. Precipitation is not the dominant factor restricting the growth of plants in such region, thus the influence of drought on forest is slighter. (b) The plant's own characteristics: the root depths of shrubs and grass are shallow, and their root systems are powerful and sensitive to precipitation, thus they have an active response to the drought.

Table 2 Correlation coefficients (r) and p-values of the NDVI and PDSI for different vegetation types within the Laohahe catchment.

Month	Forest		Shrub		Grass		Crop	
	r	p-value	r	p-value	r	p-value	r	p-value
Apr	0.497	0.022	0.554	0.009	0.573	0.007	0.551	0.010
May	0.455	0.038	0.774	0.000	0.749	0.000	0.678	0.001
Jun	0.681	0.001	0.807	0.000	0.800	0.000	0.770	0.000
Jul	0.485	0.026	0.724	0.000	0.696	0.000	0.621	0.003
Aug	0.555	0.009	0.677	0.001	0.670	0.001	0.669	0.001
Sep	0.250	0.274	0.368	0.101	0.346	0.125	0.372	0.096
Oct	−0.001	0.997	0.235	0.306	0.204	0.375	0.148	0.521

Seasonal timing of vegetation and moisture relationship

The correlation analyses demonstrate that the relationship between the NDVI and PDSI varies markedly during the growing season. The examples discussed above indicate that there is a stronger relationship in the first half of the growing season (April–August) than at the end (September–October). The correlations between the NDVI and PDSI for all vegetation types are shown in Fig. 2. The seasonal vegetation growth characteristics, e.g. onset, peak, and senescence, can be easily seen on these graphs. When comparing the correlation coefficients with the NDVI phenological cycle, it is clear that vegetation response to moisture availability varies significantly between months.

The seasonal patterns of NDVI for all vegetation types is unimodal, the maximum values of which appearing in August. Very high positive correlations between the NDVI and PDSI for all vegetation types appear in June when the vegetation is growing. And the lower correlations are noted at the end of the growing season for all vegetation types. This implies that the influence of drought on vegetation vigour is stronger in the first half of the growing season before the vegetation reaches its peak greenness, after which the influence of drought on vegetation vigour is quickly weakened. This analysis of the correlation between the NDVI and PDSI also indicates that vegetation response to drought depends on the plant growth stage. In general, the moisture

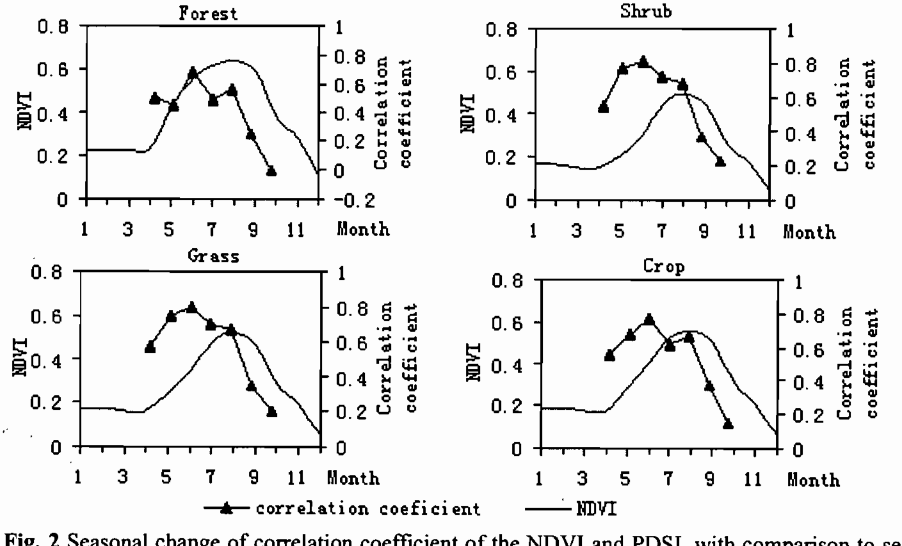

Fig. 2 Seasonal change of correlation coefficient of the NDVI and PDSI, with comparison to seasonal NDVI variations for all vegetation types. The NDVI curves were obtained by averaging monthly values over all 21 years.

sensitive period occurs during the development of reproductive organs, when plants obtain maximum beneficial effect from the available water supply (Salter & Goode, 1967). In this study, the highest correlations appear in June when the vegetation is growing, which is the moisture-sensitive period for the vegetation in the Laohahe catchment.

Relationship between the NDVI and PDSI

The goodness-of-fit of the NDVI and PDSI correlation was tested and measured using analysis of variance and R^2 from the regression model with dummy variables (equation (3)). Based on the regression models with dummy variables, the correlations between the NDVI and PDSI for all vegetation types are very significant, with p-value < 0.001, and $R^2 > 0.9$. For example, Table 3 shows the result of the regression model for grassland.

Table 3 Regression analysis on the NDVI and PDSI for grassland.

Variable	Coefficient	Standard error	t-value	p-value
Intercept	0.1632	0.008	20.612	<0.0001
PDSI	0.0095	0.004	2.552	0.0118
D_1	0.0755	0.011	6.736	<0.0001
D_2	0.1734	0.011	15.481	<0.0001
D_3	0.3116	0.011	27.827	<0.0001
D_4	0.3572	0.011	31.899	<0.0001
D_5	0.298	0.011	26.606	<0.0001
D_6	0.1214	0.011	10.833	<0.0001
D_1(PDSI)	0.0279	0.009	3.118	0.0022
D_2(PDSI)	0.0331	0.009	3.881	0.0002
D_3(PDSI)	0.0154	0.007	2.367	0.0193
D_4(PDSI)	0.0154	0.007	2.313	0.0221

$F_{11,135} = 165.64$, p-value < 0.001, $R^2 = 0.931$

The following regression functions representing the linear relationship between the NDVI and PDSI for grassland in each month of the growing season was obtained by inserting the estimated coefficients (Table 3) and the values of dummy variables into equation (3):

$$
\begin{array}{llll}
\text{April} & \text{NDVI} = 0.1632 + 0.0095(\text{PDSI}) & \text{August} & \text{NDVI} = 0.5204 + 0.0249(\text{PDSI}) \\
\text{May} & \text{NDVI} = 0.2387 + 0.0374(\text{PDSI}) & \text{September} & \text{NDVI} = 0.4612 + 0.0095(\text{PDSI}) \\
\text{June} & \text{NDVI} = 0.3366 + 0.0426(\text{PDSI}) & \text{October} & \text{NDVI} = 0.2846 + 0.0095(\text{PDSI}) \\
\text{July} & \text{NDVI} = 0.4748 + 0.0249(\text{PDSI}) & &
\end{array} \quad (4)
$$

The different intercepts and slopes of these regression equations indicate the change of the NDVI–PDSI relationship from one month to another. The maximum intercepts values occur in August for all vegetation types as a result of the maximum values of NDVI appearing in August. The maximum values of the slope for all vegetation types appear in June, which reflect that vegetation is most sensitive to moisture conditions in June.

In order to demonstrate that the relationship between the NDVI and PDSI can be simulated by the regression model with dummy variables, taking grassland as an example, the monthly NDVI of grass during the period of 1996–2001 was simulated (Fig. 3). The simulated NDVI using the regression with dummy variables fits the observed NDVI quite well, and the regression model with dummy variables has a very high goodness-of-fit.

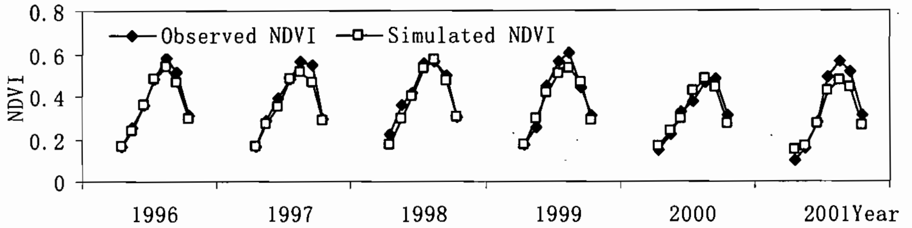

Fig. 3 Comparison between the simulated and observed NDVI for the grassland in the Laohahe catchment.

CONCLUSIONS

The purpose of this paper is to assess the influence of drought on vegetation vigour within the Laohahe catchment by analysis of monthly AVHRR-NDVI and PDSI relationships based on forest, shrub, grass, and crop vegetation types. The correlation coefficients between NDVI and PDSI vary by month. Very high positive correlations between the NDVI and PDSI for all vegetation types appear in June when the vegetation is growing, and lower correlations are noted at the end of the growing season. This implies that the influence of drought on vegetation vigour is stronger in the first half of the growing season before the vegetation reached its peak greenness, after which the influence of drought on vegetation vigour is quickly weakened. The relationship between the NDVI and PDSI is also affected by the vegetation type. When comparing the correlation coefficients with the NDVI phenological cycle, it is clear that vegetation response to moisture availability varies significantly between months. This analysis of the correlation between the NDVI and PDSI indicates that vegetation response to drought depends on the plant growth stage. In the Laohahe catchment, June is the moisture-sensitive period for vegetation. During this period, drought has the greatest effect on the vegetation growth.

When regression techniques are used to quantify the relationship between the NDVI and PDSI, the seasonal effect was taken into account. The good goodness-of-fit of the NDVI and PDSI correlation using a regression model with dummy variables indicated that the NDVI is a good indicator of moisture condition and can be an important data source when used for detecting and monitoring drought in this area, and seasonality should be an important factor when NDVI and NDVI-derived vegetation indices are used for drought detection.

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Application of a fuzzy comprehensive model for drought assessment: Jinan City, China, case study

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Abstract Drought is different from other natural hazards and it is difficult to recognize the seriousness of drought at its beginning. Economic, environmental, and social drought impacts can be enormous. The purpose of this paper is to comprehensively evaluate drought degree. Based on the single-objective method of drought assessment, a fuzzy assessment model for drought assessment is introduced; several influencing factors, such as surface water, groundwater, soil moisture and precipitation are considered in evolving a new system of drought assessment and realizing drought assessment quantitatively. This model was applied to some areas for drought assessment and was found to be uniform in evaluating the degree of drought.

Key words fuzzy theory; drought assessment; drought indexes; comprehensive assessment; Jinan City, China

INTRODUCTION

Water scarcity has been a major issue in China since the turn of the 21st century, and this has influenced economic development and environmental improvements. Along with the population increase, economic development and changing natural conditions, the water-deficit situation has becomes increasingly serious and the frequency, scope and impacting area of drought occurrence has expanded progressively. Some structural and systematic obstacles existed in the long-term drought fighting work, which faces unprecedented challenges (He, 2005). Due to historical reasons and conceptual differences, serious drought disasters are usually covered up by sudden flood accidents, resulting in a lack of research on drought events. So it is very important to assess drought grade correctly and take corresponding drought-resistance measures to minimize the drought losses and safeguard social economic stability (Liu, 2005).

In drought research, agreeing on a drought index is the most important problem. At present, a number of scholars particularly emphasize agricultural droughts and meteorological droughts. The Palmer drought severity index (PDSI) method is widely applied to real time drought survey, which introduces the concepts of water balance and supply–demand relationships to describe the drought process from formation, development, reduction, to end (Ramachandra *et al.*, 1984; Gengshan *et al.*, 2004; Heim *et al.*, 2006; Liangfang, 2007; Yang, 2007; Yue *et al.*, 2008), on the assumption that annual precipitation obeys the normal distribution. Xu (1950) used precipitation standard deviation to divide drought ranks; Changming (2002) analysed and processed the drought degree of the Haihe River basin by the departure index; Ping (2000) applied artificial neural networks to assess drought degree. The research shows that the indices used in describing hydrological drought and meteorological drought are different. Therefore, establishing a comprehensive drought index is a new study area.

This paper presents an assessment of drought associated with a fuzzy assessment and forecast model, based on the single-objective methods of drought assessment, building a new system of drought assessment and forecast, and realizing drought assessment quantitatively.

METHODOLOGY OF THE FUZZY COMPREHENSIVE MODEL

Determining assessment indexes

Drought disaster not only means agricultural drought or meteorological drought, but also refers to many influencing factors, such as meteorology (precipitation, evaporation, temperature), hydrology (upstream water, project impounding, groundwater), soil (texture, moisture content), crops (different varieties and development stages), irrigation conditions, etc. (Shengle, 2006). This paper determines four assessment indices, viz. surface water drought index, groundwater drought index, soil moisture and precipitation drought indices, taking hydrologically-induced water resources drought disasters as a basis, and considering the influence of artificial factors, based on the single-objective methods of drought assessment.

Determining grade set of assessment indices

According to Floods And Droughts In China and drought fighting practices, drought degree can be divided into four grades, viz. light drought, moderate drought, severe drought and extreme drought. This division method includes fuzziness, so it is essential to use fuzzy set theories to make scientifically rational assessment on drought disaster.

Assessing drought grade with single-objective methods, building fuzzy membership matrix

Surface water and precipitation are divided by using the anomaly percentage. For groundwater depth and soil moisture, taking the frequency analysis and finding the values corresponding with 1%, 50% and 99%, then interpolating 14 points similarly, we get m_1(1%), …, m_{17}(99%), and so they are divided. Four single-factor indexes are divided into nine grades, as shown in Table 1. The detail of building the fuzzy membership matrix, and the detailed derivation process are given in Chen (1998) with results as below.

Table 1 Standard characteristic value of drought assessment.

Grade	Surface water	Groundwater	Soil moisture	Precipitation
Extreme wetness	>80%	$<m_{16}$	$>m_{16}$	>28%
Severe wetness	50% – 80%	$m_{16} - m_{14}$	$m_{14} - m_{16}$	28% – 20%
Moderate wetness	30% – 50%	$m_{14} - m_{12}$	$m_{12} - m_{14}$	20% – 12%
Light wetness	10% – 30%	$m_{12} - m_{10}$	$m_{10} - m_{12}$	12% – 4%
Normal	–10% – 10%	$m_{10} - m_8$	$m_8 - m_{10}$	–4% – 4%
Light drought	–10% – 30%	$m_8 - m_6$	$m_6 - m_8$	–4% – 12%
Moderate drought	–30% – 50%	$m_6 - m_4$	$m_4 - m_6$	–12% – 20%
Severe drought	–50% – 80%	$m_4 - m_2$	$m_2 - m_4$	–20% – 28%
Extreme drought	<–80%	$>m_2$	$<m_2$	<–28%

The characteristic value matrix of factor i regarding sample j is denoted as $X = \{x_{ij}\}$, where $i = 1,2, \ldots, m; j = 1,2, \ldots, n$; m, n representing the number of factors and samples, respectively.

The standard characteristic value matrix of factor i regarding sample j is denoted as $Y = \{y_{ij}\}$, where $i = 1,2, \ldots, m$; $h = 1,2, \ldots, c$; c representing number of grade.

According to the definition of relative membership degree (RMD) (Shouyu, 1994, 1996), it varies linearly; when its characteristic value lies between 1 and c, the formula is as follows:

$$r_{ij} = \begin{cases} 0, & x_{ij} \le y_{ic} \text{ or } x_{ij} \ge y_{ic} \\ \dfrac{x_{ij} - y_{ic}}{y_{i1} - y_{ic}}, & y_{i1} > x_{ij} > y_{ic} \text{ or } y_{i1} < x_{ij} < y_{ic} \sqrt{b^2 - 4ac} \\ 1, & x_{ij} \le y_{i1} \text{ or } x_{ij} \ge y_{i1} \end{cases} \tag{1}$$

Similarly,

$$s_{ih} = \begin{cases} 0, & y_{ih} = y_{ic} \\ \dfrac{y_{ih} - y_{ic}}{y_{i1} - y_{ic}}, & y_{i1} > y_{ih} > y_{ic} \text{ or } y_{i1} < y_{ih} < y_{ic} \\ 1, & y_{ih} = y_{i1} \end{cases} \tag{2}$$

where r_{ij} is RMD of factor i regarding sample j, and s_{ij} is RMD of standard characteristic value of factor i regarding sample j.

Determining fuzzy weights of assessment indexes

This paper adopts the theory of relative membership degree (Chen, 2002) to determine the weight of index *i*, for which the basic principle is to find the relative membership degree of index *i* regarding fuzzy concept "importance". Though "importance" and "unimportance" are two completely opposite concepts, there is no absolute distinction but an intermediary transition between the two concepts, which are objective fuzzy characteristics taking on in the process of recognition (Chen, 2001, 1998).

Suppose the RMD regarding superiority is:

$$R = \begin{bmatrix} r_{11} & r_{12} & \cdots & r_{1n} \\ r_{21} & r_{22} & \cdots & r_{2n} \\ \vdots & \vdots & & \vdots \\ r_{m1} & r_{m2} & \cdots & r_{mn} \end{bmatrix} = (\omega_{ij}) \tag{3}$$

Then there are two cases:

Case 1, the RMD regarding superiority is bigger, the weight is bigger, i.e. importance is proportional to superiority. So take matrix R as the RMD regarding superiority;

Case 2, the RMD regarding superiority is bigger, the weight is smaller, i.e., importance is inversely proportional to superiority. The RMD regarding superiority is:

$$R = \begin{bmatrix} 1\text{-}r_{11} & 1\text{-}r_{12} & \cdots & 1\text{-}r_{1n} \\ 1\text{-}r_{21} & 1\text{-}r_{22} & \cdots & 1\text{-}r_{2n} \\ \vdots & \vdots & & \vdots \\ 1\text{-}r_{m1} & 1\text{-}r_{m2} & \cdots & 1\text{-}r_{mn} \end{bmatrix} = (\omega_{ij}) \tag{4}$$

since each alternative is in a state of fair competition and has the same weight. Suppose the RMD regarding important is $\omega_{(i)}$, regarding unimportant is $\omega_{(i)}^{c}$, there must be $\omega_{(i)}^{c} = 1 - \omega_{(i)}$. So $\omega_{(i)}$ and $\omega_{(i)}^{c}$ can be considered as weights to calculate the generalized distances, respectively.

$$\begin{aligned} D_{Ii} &= \omega_{(i)} d_{Ii} = \omega_{(i)} \left\{ \sum_{j=1}^{n} (1 - \omega_{ij})^{p} \right\}^{\frac{1}{p}} \\ D_{Oi} &= \omega_{(i)}^{c} d_{Oi} = (1 - \omega_{(i)}) \left\{ \sum_{j=1}^{n} \omega_{ij}^{p} \right\}^{\frac{1}{p}} \end{aligned} \tag{5}$$

The objective function is established as follows:

$$\min \left\{ F(\omega_{(i)}) = D_{Ii}{}^{2} + D_{Oi}{}^{2} = \omega_{(i)}^{2} \left[\left(\sum_{j=1}^{n} (1 - \omega_{ij})^{p} \right)^{\frac{1}{p}} \right]^{\delta} + (1 - \omega_{(i)})^{2} \left[\left(\sum_{j=1}^{n} \omega_{ij}^{p} \right)^{\frac{1}{p}} \right]^{\delta} \right\} \tag{6}$$

To set the partial derivatives of the above function with respect to each $\omega_{(i)}$ equal to zero, $dF(\omega_{(i)})/d\omega_{(i)} = 0$, we can obtain:

$$\omega_{(i)} = \left[1 + \left(\frac{\sum_{j=1}^{n} (1 - \omega_{ij})^{p}}{\sum_{j=1}^{n} \omega_{ij}{}^{p}} \right)^{\frac{\delta}{p}} \right]^{-1} \tag{7}$$

By normalizing the weight vector:

$$\omega = \left(\frac{\tilde{\omega}_1}{\sum_{i=1}^{m}\tilde{\omega}_i}, \frac{\tilde{\omega}_2}{\sum_{i=1}^{m}\tilde{\omega}_i}, \cdots, \frac{\tilde{\omega}_m}{\sum_{i=1}^{m}\tilde{\omega}_i}\right) \tag{8}$$

where δ and p are the optimization criterion parameter and the distance parameter, respectively.

Applying all the above to the fuzzy comprehensive model

The difference between sample j and grade h is represented by weighted distance vector,

$$d_j = (d_{1j}, \cdots, d_{a_j j}, d_{(a_j+1)j}, \cdots, d_{b_j j}, \cdots, d_{cj}),\ \text{with}\ d_{hj} = \left\{\sum_{i=1}^{m}\left[\omega_i(r_{ij} - s_{ih})\right]^p\right\}^{\frac{1}{p}} \tag{9}$$

where $h = a_j, \ldots b_j$, and d_{cj} is the generalized weighted distance between sample j and grade h.

Establish the following objective function:

$$\min\left\{F(u_{hj}) = \sum_{h=a_j}^{b_j} u_{hj}^2 d_{hj}^{\alpha}\right\},\ \text{with}\ \sum_{h=a_j}^{b_j} u_{hj} = 1 \tag{10}$$

Construct the Lagrange function, assuming λ_j as the Lagrange multiplier:

$$L(u_{hj}, \lambda_j) = \sum_{h=a_j}^{b_j} u_{hj}^2 d_{hj}^{\alpha} - \lambda_j\left(\sum_{h=a_j}^{b_j} u_{hj} - 1\right) \tag{11}$$

Set the partial derivatives of the above function with respect to λ_j and u_{hj} equal to zero: $\frac{\partial L(u_{hj}, \lambda_j)}{\partial u_{hj}} = 0$ and $\frac{\partial L(u_{hj}, \lambda_j)}{\partial \lambda_j} = 0$ we can obtain:

$$u_{hj} = \left\{d_{hj}^{\alpha}\sum_{k=a_j}^{b_j} d_{kj}^{-\alpha}\right\}^{-1}\ \text{with}\ d_{hj} \neq 0, a_j \le h \le b_j \tag{12}$$

Generally, we can get the complete form as follows:

$$u_{hj} = \begin{cases} 0, & h < a_j \text{ or } h > b_j \\ (d_{hj}^{\alpha}\sum_{k=a_j}^{b_j} d_{kj}^{-\alpha})^{-1}, & d_{hj} \neq 0, a_j \le h \le b_j \\ 1, & d_{hj} = 0 \text{ or } r_{ij} = s_{ih} \end{cases} \tag{13}$$

CASE STUDY

Area description and analysis of drought hazard

Shandong province is situated in eastern China, between 34–38°N and 114–122°E, and includes 17 cities with the province capital of Jinan City. Jinan City is in the warm temperate semi-humid and monsoon-controlled climatic zone with hot, rainy summers and cold, dry winters. The water resources mainly consist of two parts: precipitation, and rivers flowing through the city, such as the Yellow River. The multi-year average precipitation is about 654 mm, occurring in mostly summer. The water deficit situation in Jinan City is uniform intra-annually and in regional distribution, with obvious annual variations.

Using the surface water, groundwater, soil moisture and precipitation data in 1989, the fuzzy comprehensive model is applied to evaluate the drought degree for Jinan City, Shandong provincial capital, and compare the results with the actual situation.

Assessing drought degree with single-objective methods

According to the above method, we can obtain nine standard characteristic values of four indexes, as shown in Table 2, and the characteristic values for Jinan City in 1989, as shown in Table 3.

Table 2 Standard characteristic value of drought assessment.

Rank	Surface water (10^4 m^3)	Groundwater (m)	Soil moisture (%)	Precipitation (mm)
Extreme wetness	>4610	<0.808	>24.23	>790
Severe wetness	3842–4610	0.808–1.445	21.39–24.23	741–790
Moderate wetness	3329–3842	1.445–2.082	18.54–21.39	691–741
Light wetness	2817–3329	2.082–2.718	15.70–18.54	642–691
Normal	2305–2817	2.718–3.355	13.24–15.70	593–642
Light drought	1793–2305	3.355–3.992	11.17–13.24	543–593
Moderate drought	1281–1793	3.992–4.629	9.10–11.17	494–543
Severe drought	512–1281	4.629–5.266	7.03–9.10	444–494
Extreme drought	<512	>5.266	<7.03	<444

Table 3 Characteristic value for Jinan City in 1989.

City	Surface water (10^4 m^3)	Groundwater (m)	Soil moisture (%)	Precipitation (mm)
Jinan City	132.00	8.01	9.33	346.00

Calculating the relative membership degree matrix

According to equations (1) and (2), we can work out the RMD regarding the index and standard characteristic value, as follows:

$$\mathrm{R}=(r_{\mathrm{ij}})=\begin{bmatrix}0.58\\0\\0.18\\0\end{bmatrix} \qquad \mathrm{S}=(s_{\mathrm{ih}})=\begin{bmatrix}1,0.93,0.81,0.69,0.56,0.44,0.31,0.19,0\\1,0.93,0.86,0.71,0.57,0.43,0.29,0.14,0\\1,0.88,0.83,0.67,0.50,0.36,0.24,0.12,0\\1,0.91,0.86,0.71,0.57,0.43,0.29,0.14,0\end{bmatrix}$$

Calculating weights

According to equation (7), the unnormalized weight vector of the four process indexes is: $\tilde{\omega}=\{\tilde{\omega}_1,\tilde{\omega}_2,\tilde{\omega}_3,\tilde{\omega}_4\}=\{0.344,1,0.954,1\}$ and according to equation (8), we normalize: $\omega=\{\omega_1,\omega_2,\omega_3,\omega_4\}=\{0.105,0.303,0.289,0.303\}$

from $$u_{\mathrm{h}}=\frac{1}{\sum_{\mathrm{i=1}}^{4}[\omega_{\mathrm{i}}(r_{\mathrm{i}}-s_{\mathrm{ih}})]^2\sum_{\mathrm{k=1}}^{9}\frac{1}{\sum_{\mathrm{i=1}}^{4}[\omega_{\mathrm{i}}(r_{\mathrm{i}}-s_{\mathrm{ik}})]^2}}$$

The RMD vector of drought degree for Jinan City in 1989 is as follows: $u=(0.009,0.012,0.013,0.021,0.033,0.062,0.136,0.384,0.329)$.

According to the maximum membership degree law, we can obtain the result that the drought degree of Jinan city in 1989 is serious drought.

RESULT AND DISCUSSION

During 1989, below normal rainfall and high temperatures caused serious water resources scarcity in Jinan City. This event is regarded as drought disaster, with a return period of 100 years. Calculation of the drought degree with the fuzzy comprehensive model, reflected a serious drought.

A large number of studies are focusing on single-objective methods. Evaluating drought degree is an uncertain problem, to a great extent with fuzziness. The study selected the constraining factors to construct the criterion set for assessment, introduced the fuzzy optimization theory, taking the numerous uncertain influential factors into account. The method presented offers an effective approach to evaluate the degree of the drought. The method used to evaluate drought degree in Jinan City gave satisfying results.

The factors influencing degree of drought are more complex. Drought occurrence is affected by many factors, such as precipitation, geographical environment, social economy and so on, which make it a different thing to identify drought degree. Methods of general drought assessment should satisfy the following points (Ping, 1997): (a) incorporate the principle of water balance; (b) determine drought duration and development; (c) drought indexes of different sites and seasons should be comparable; (d) there should be no qualitative contradictions when comparing with drought disasters that have actually occurred; and (e) the data should be easily obtainable.

This paper takes into account four factors. However, it is suggested that more factors should be considered to improve the drought assessment model in further studies.

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Integrated water resources management in the vulnerable Indian environment

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Abstract India, with 16.7% of the world's human population and 15% of the farm animal population, but with only 2.42% of the geographical area of the globe, receives an average of 1% of global rainfall, which contributes to 4% of the global freshwater resource. The per capita availability of water at the national level has reduced from about 5177 m^3 in 1951 to 1820 m^3 in 2000, with a great variation in water availability in different river basins. By 2025 and 2050, per capita water availability is projected to fall to 1341 and 760 m^3, respectively. The hydrological cycle is greatly influenced by changing land use/land cover and anthropogenic transformations. This paper aims to understand vulnerable environments such as the flood prone, drought prone and climate change prone Himalaya. Integrated watershed management is required for understanding hydrological processes and their interaction.

Key words integrated water management; vulnerable environment; floods; droughts; climate change; community participation; India

INTRODUCTION

Freshwater availability is a key to human survival. Of the global land area, 70%, including large parts of India, will become water stressed by 2050. The average runoff in the river system of the country has been assessed as 1869 km^3. Of this, the portion utilizable by conventional storage and diversion is estimated as about 690 km^3. The large-scale deforestation has caused significant changes in regional and global climate. Rainfall variability in both time and space makes water availability uncertain. India consists of more than 68% rainfed areas, which are difficult to manage in a sustainable manner. With growing urban impact on groundwater, the supply of freshwater is becoming contaminated. Integrated water resources management (IWRM) should focus on understanding relationships between precipitation, evapotranspiration, recharge of aquifers, land-use/land-cover change, agriculture and urban development, etc. (Singh, 2004).

In order to reduce the vulnerability of critical regions, various resources, i.e. demographic, social, physical, etc., need to be utilized properly so that maximum benefits can be extracted from them and vulnerability can be minimized. Unfortunately, due to rapid population growth and development of human settlements in disaster-prone areas, more and more people and their assets are becoming vulnerable to natural hazards. Mitigation focuses on minimizing the adverse impacts of the hazard on communities. This paper also discusses the role played by the local knowledge for achieving IWRM through monitoring, predicting and mitigating vulnerability in economically backward and socially deprived regions.

Geographically, a region can be considered vulnerable on the basis of biophysical and human induced pressures. Identifying vulnerability brings about opportunities to overcome pressures through the focused application of water management technology. The onset of criticality is a manifestation of ill-adapted institutional arrangements, acting without consideration for the vulnerable population (Smith, 1992; Wisner *et al.*, 2004). Different types of vulnerabilities, such as social, institutional or infrastructural, are all inter-related. The consequence of population growth in the context of highly unequal access to land is that more and more marginal land is being encroached. This is particularly true for the low-lying islands of the lower Ganga valley, which have emerged as a result of silt deposition in the river estuaries of the delta regions. Reducing vulnerability to disasters will be tied up with increased resource access and empowerment. Water induced problems are more sensitive to food shortages, climate change and other extremes.

STATE OF VULNERABLE ENVIRONMENT IN INDIA

India has a diverse range of natural environments. Its unique geo-climatic conditions make this country among the countries most vulnerable to natural disasters in the world, affecting about 85% of its total geographical area. Disasters have always posed a threat to human security and their impact on humans has increased considerably in spatial scale as poor and marginal people have moved into disaster prone areas. India is highly vulnerable to drought, floods, cyclones, earthquakes, tsunamis, and landslides. Almost all parts of the country experience one or more of these events (Singh, 2004, 2006).

Vulnerability to floods

In the case of flooding in India (annually about 32 million people are affected, and 11 683 houses damaged), there are three levels of vulnerability: relative vulnerability towards the upper socio-economic group; a highly vulnerable category of marginal and small landholders; and an effectively "sub-vulnerable" category of landless households. This can be understood by an enquiry of flood damage to the material household (Fig. 1). Larger landholders may suffer more crop damage in absolute terms due to their larger landholdings, but they are able to recover from

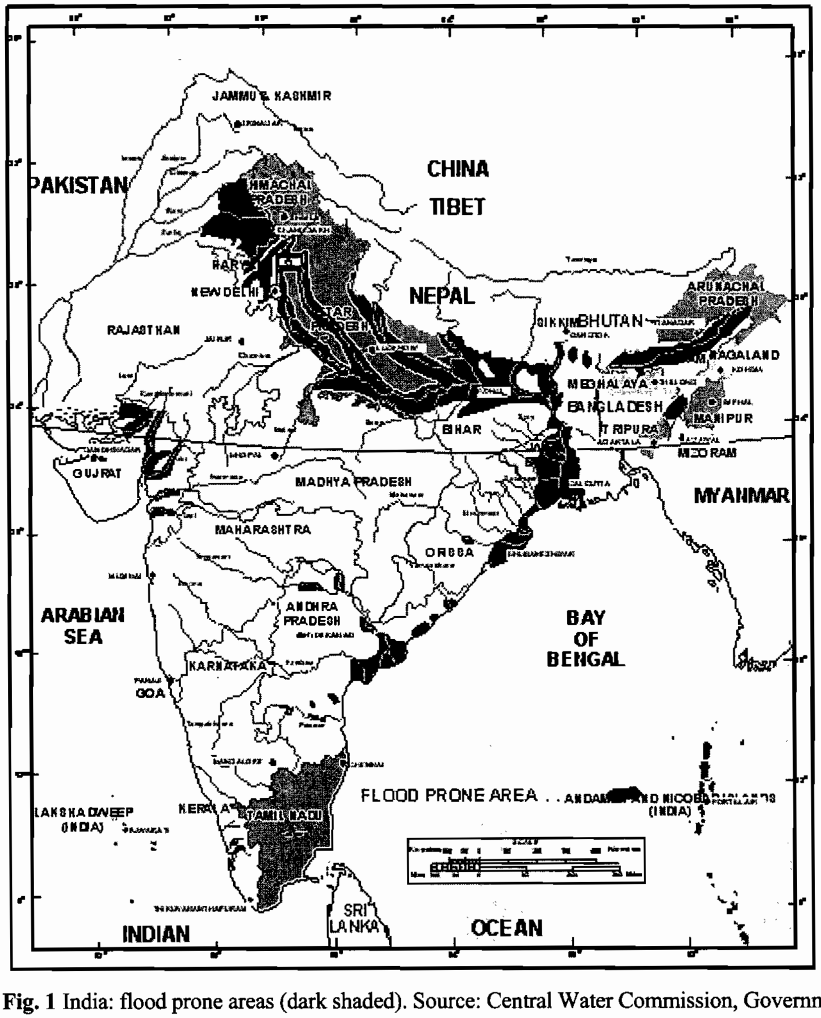

Fig. 1 India: flood prone areas (dark shaded). Source: Central Water Commission, Government of India.

their crop losses by getting in a better crop as soon as the floodwater recedes. Landless households are by definition exempted from crop damage, other than through its impact on labour demand. It is only for the farmers in the small and the marginal landholdings and without any additional occupation that the 100% crop damage automatically means 100% loss of their livelihood.

A household with large landholdings may use their savings to cope with impacts. For a household already landless and but providing agricultural labour, flood damage might mean short-term hardship, with their livelihood restored as soon as their village land owners begin to engage wage labourers. Small and marginal farmers are forced to adopt a range of options in their attempts to recover from flood damage. These might include selling productive assets such as livestock and agricultural implements; and mortgaging or even selling land. Such measures are likely to involve substantial and often irreversible decline in the household's economic base (IGNOU, 2005). This calls for integrated management of flood plains incorporating flood plain zoning, land-use planning, alternative livelihood options and community-based watershed management plans.

Vulnerability to droughts

In India, drought annually affects 50 million people, and during 1999–2000 in 11 states, 37 million people, together with 13 million ha of land, were affected. Nineteen per cent of India's total area with 12% of its population is drought prone. Droughts are caused by shortfall in precipitation, and by disruptions in the conveyance and storage of water. Mostly, it is a consequence of the failure of rains, while urbanisation, overgrazing, deforestation and even farming can reduce the water retention of soils. Droughts cause severe reduction in water-availability and soil moisture levels below the minimum necessary for sustaining plant, animal and human life. Their severity depends on their duration and the extent of the area affected. Their impacts depend on the level of socio-economic development and administrative, financial and technical capacity of the community or the state to respond to them. In India, the states of Rajasthan and Gujarat are severely affected by drought, while Madhya Pradesh, Orissa and Andhra Pradesh are moderately affected (Tables 1 and 2). Starvation, deaths and stories of abject poverty regularly emanate from these states, particularly Orissa. Thus, poverty as the background in disaster-prone states brings out challenging issues of human security in India. Critical changes in agricultural practices, emerging out of the Green Revolution, have resulted in intensive irrigation practices and promoted water intensive crops. This has phased out traditional crops that have inherent drought-proofing mechanisms to survive fluctuations in rainfall (IGNOU, 2005). This requires implementation of IWRM incorporating diversification of the economy, traditional water harvesting, soil moisture and water conservation.

Vulnerability to climate change

Climate change is bringing several impacts to the Indian environment, i.e. variability in rainfall patterns, melting of glaciers, increased runoff, excessive flooding and water scarcity. Impending

Table 1 Impact of droughts in India, 2001.

State	Districts total	Districts affected	No. people affected (10^5)	No. cattle affected (10^5)	Crop areas affected (10^5)	No. of villages affected
Chhatishgarh	16	25	94	NA	11.36	10 252
Gujarat	25	22	291	107	13.50	11 240
M.P.	45	32	127	87.7	39.52	22 490
Orissa	30	28	119	399	11.00	15 000
Rajasthan	32	31	330	399	89.47	31 000
Himachal Pradesh	12	12	46	NA	0.88	NA
Maharashtra	35	26	455	2.5	45.09	20 000
J & K	Severe	Drought	72% less rainfall	NA		

Source: Government of India, Ministry of Agriculture (2001).

Table 2 Drought frequency in different meteorological subdivisions.

Meteorological subdivision	Recurrence of very deficient rainfall
Assam	Once in 15 years
West Bengal, Madhya Pradesh, Coastal Andhra Pradesh, Kerela, Bihar, Orissa,	Once in 4 years
North Karnataka, eastern U.P , Vidarbha, Gujarat,	Once in 3 years
Eastern Rajasthan, , western U.P. TN, Kashmir Rayalaseema, Telengana, Western Rajasthan	Once in 2.5 years

Source: National Committee on Development of Backward Areas (1998).

large variability in the monsoonal precipitation is likely to bring more extreme events and water scarcity problems. The melting of snow and glaciers (more than 5000 in number) feed several rivers, which sustain life on the plains. The Indus-Ganga-Brahmaputra river basin is a large and crowded region with millions of people who are highly dependent on large, international, and monsoon-dependent rivers for their livelihood security (especially rural people) and other benefits to improve their quality of life. The annual rainfall ranges from 300 mm in the semi-arid climate in the west to over 2000 mm in the east. Based on Global Climatic Model (GCM) analysis, scientists suggest a substantial increase of temperature and rainfall over India in the next 100 years (Pant & Kumar, 1997; Lal, 2001).

Therefore, with a changing climate, these glaciated rivers can affect the downstream areas (McCarthy *et al.*, 2001; Singh, 2004; Parry *et al.*, 2007). The rate of glacial retreat in recent times has, however, been much more rapid than the gradual retreat expected in an inter-glacial warming phase. Mountains act as the main hydrological and climatological link of the water cycle. In terms of their role as water towers, mountain regions form an important supply of snow and/or rainfed water to the lowlands. While the relationship between climate and glacier mass balance is complex, the most important effect of recent global warming is the substantial loss of glacier ice that has occurred in all mountainous regions of the world, including the Indian Himalaya (Sen Roy & Singh, 2002).

Gangotri, the headwater of the River Ganga, is receding continuously at an alarming rate because of global warming as well as changes in micro-climate (Sangewar, 1998; Hasnain, 1999). The Bhagirathi River system and the Alaknanda River system meet at Devprayag to be called the Ganga. The Bhagirathi River originates from the Gaumukh, which is the snout of the Gangotri Glacier. As far as climate is concerned, it varies according to altitude. It is quite diverse ranging from tropical to severe cold. To understand the process and pattern of glaciers, their hydrological behaviour and associated hazards, the research work of several relevant governmental agencies have been consulted. Primary data collection involved first-hand experience of the glacial and hydrological behaviour. Most of the glaciers of the Himalaya and Karakoram are receding at varying rates (Table 3) (Vohra, 1981; Singh, 2004).

A recent flood disaster on the Kosi River in Bihar is another example of the cumulative effects of such natural and human vulnerability. The country is debating the interlinking of northern and southern rivers for integrated management of floods and droughts.

WATERSHED AS AN UNIT FOR INTEGRATED WATER MANAGEMENT

A watershed management project involving the community has had far reaching impact on people's lives all over the country. In many districts of India, village watershed committees implement the programmes. People have formed self-help groups (SHGs) to supplement their income. Usually, the rainwater would drain off or will simply dry up. Now, the community along with the state governments and NGOs has started watershed programmes to conserve water in wells, manmade ponds and bunds. The states have set up a Watershed Development Mission – the first of its kind in India – to take up livelihood and community development programmes.

Table 3 Retreat of glaciers in Himalaya and Karakoram.

Mountain range	Name of glacier	Periods	Year	Total retreat (m)	Average retreat (m/year)
Himalaya	Milam	1849–1957	108	1350	12.5
	Pindari	1845–1966	121	2840	23.5
	Shankulpa	1881–1957	76	518	6.8
	Poting	1906–1957	51	262	5.1
	Glacier No.3 in Arwa Valley	1932–1956	24	198	8.3
	Gangotri	1934–1976	41	600	14.6
		1962–2000	38	1341	35.4
	Zemu	1909–1965	56	440	7.9
		1975–1990	15	297	19.8
	Sonapani	1906–1963	57	905	15.9
Karakoram	Minapin	1906–1929	23	502	21.8
	Biafo	1861–1922	61	0	0
	Kichik Kumdan	1946–1958	12	1219	101.6
	Siachin	1929–1958	29	914	31.5
	Yengutsa	1892–1925	33	4134	125.3

Source: Vohra (1981) and Bahadur (2004).

Watershed development is not just about livelihood, but also capacity building. Awareness levels among villagers have increased and community mobilization is easier. This is considered helpful for drought and flood management in order to mitigate climate change in an integrated way (Goel & Singh, 2006a,b).

POLICY FOR ADAPTING TO DISASTER IMPACTS

For vulnerability reduction, integrated planning has to be done in a strategic way. There should be proper land-use planning and regulations for sustainable development of the area and minimization of vulnerability. Preventing or modifying the occurrence of disaster, such as in the case of floods, can reduce the physical impact of hazards. This can be done very efficiently in relatively small catchments by land-use planning and management. Rapid urbanization has led to higher concentrations of people living in hazardous areas and consequently to higher losses when disasters occur. As urbanization also alters the response of watersheds to rainfall, many large cities in the region are becoming increasingly vulnerable to flooding (Carter, 1991). Irrigation and reservoirs may buffer agriculture and other sectors against precipitation deficit. High-yielding varieties and other technological changes may have increased the variability of cereal yields; others believe that the Green Revolution has reduced ecological and social vulnerability to environmental change. Changes are necessary in land and water use, and allocation policy. A biomass strategy must be implemented to raise production through wasteland development. Long-term disaster reduction efforts should, therefore, aim at promoting appropriate land use in the disaster prone areas, by harmonizing land suitability with agricultural development strategies.

COMMUNITY BASED VULNERABILITY MITIGATION

In the recent past, the NGO sector has played a major role in strengthening the community to face their vulnerability. The trend is based on long-term experience of the need of maximum self-reliance at the lowest level. It is illustrative to note the well targeted efforts made in certain areas where communities have formed their own organizations. One such community-based organization, the Church Auxiliary for Social Action (CASA), has formed village task forces in villages of Andhra Pradesh. However, the community as an effective institution is yet to take shape in this country, mainly due to low literacy levels and widespread poverty. Considerable efforts are being made to form and strengthen community-based organized at grassroots levels. A few NGOs have success stories, such as the contributions of Tarun Bharat Sangh of Rajasthan, SEWA of Gujarat, Sukhomajri of Haryana and Ralegaon Siddi of Maharashtra, and are worth mentioning. Such programmes have transformed the lives and environment of the particular region on a sustainable basis. In recent years, the government has encouraged private sector and civil societies in this field. People's resistance to water privatization is emerging as with the water sovereignty (Jal Swaraj) march.

Institutions can be considered as social tools for the management of vulnerability. Institutions also minimise vulnerability and conflicts, and enhance human security, through sustainable management of resources. The level of institutions contributes to mitigating the vulnerability. The more assets people control, the less vulnerable they are and the greater their capacity to successfully cope with risks. In such areas like Punjab, Haryana and Maharashtra, farmers and people are coping more easily with disasters. The main reason is the high level of human security in terms of per capita income and a good standard of social and economic well being. However, regions like Orissa and Bihar that are poor with respect to their infrastructure are prone to disasters with high risk to lives and property.

CONCLUSION

India will be among the countries worst affected by water scarcity. As India's population is more than one billion, we need to find a solution to the water crisis. According to the South Asia Report of the World Commission of Water for the 21st Century, India will need to double its water supply by the year 2025 to ensure food, livelihood, health and ecological security. India receives 1194 mm rainfall, which is 394 mm more than the world average of 800 mm. A large amount of precipitation is allowed to flow as runoff and is not collected properly. Successful watershed management programmes in some regions are contributing substantially to water resources sustainability in an integrated way. Promotion of rainwater harvesting for aquifer replenishment and watershed management is gaining currency. Since 1995, the Participatory Watershed Development Programme has been implemented by the Ministry of Rural Development, Government of India. More recently the Department of Space, Government of India, has launched an Integrated Mission on Sustainable Development using remote sensing and GIS-based spatial information systems as decision support systems for watershed management in India. There is a need to improve water governance in the country towards promoting sustainable development in spatial and temporal perspectives. Therefore, keeping in mind the prevailing situation as well as the potential after the 73rd Constitutional Amendment, Watershed Development has been included in the schedule of subjects to be handled by panchayats (village government). This provides opportunities for combining development of grass roots democracy and natural resources in a symbolic manner. The programme is based on various interventions related to land and water resources at the micro-watershed level. Civil society, particularly business houses (companies) have to take a pro-active role. Recently the Citizen's Front for Water Democracy came into existence. However, there exist regional differences in the implementation. Thus, improving water governance would ensure the supply of water to every field, remove poverty from poor areas, provide green cover over deforested areas and improve the environment. Good water governance is also described as the programme that holds the key to solving problems of employment, the economy, exports, equity and environment.

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5 Change Assessment and Management

Impact of human activities on the flow regime of the Hanjiang

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Abstract The Hanjiang River is the longest tributary of the Yangtze River. With the development of water resources, the flow regime of the Hanjiang River has been altered by dam building to some extent, and consequently water allocation between different water uses may be affected. To assess dam-induced alterations in the flow regime of the Hanjiang River, this paper selected three hydrological stations above and below the Danjiangkou reservoir, as case study sites, and the whole study period was divided into two subperiods by the year when the reservoir started to store water. On the basis of the 51-year long time series of daily discharge data, the alterations in annual, seasonal, monthly and daily runoff in the two subperiods and the driving forces were analysed. The output of this paper could provide references for the integrated management of the Hanjiang River water resources and the assessment of dam-induced impacts on the Hanjiang River ecosystem's health.

Key words Hanjiang River; Danjiangkou Reservoir; flow regime; runoff

INTRODUCTION

The Hanjiang River, having a length of 1577 km and a catchment area of 159 000 km^2, is the longest tributary of the Yangtze River and merges with the Yangtze River at Wuhan, a city of several million. Although its average annual runoff is 51.3 billion m^3, the richness in water resources can not guarantee the sufficient water supply for all water users due to the temporally uneven distribution (the total runoff in May to October roughly accounting for 75% of annual runoff) (Li *et al.*, 2007). The inter-annual variation of the annual runoff is very significant in all tributaries of the Yangtze River, however, the rich water resources of the Hanjiang River plays a critical role in promoting the local socio-economic development (Den, 1981). With the development of water resources, the flow regime of the Hanjiang River has been altered by dam building to some extent (Zou *et al.*, 2007), and consequently water allocation among different water uses may be affected and the environmental water use may be ignored to some degree. Recently, the study of human impacts on river hydrological regimes, an important topic associated with river ecosystem health and stability, has attracted worldwide attention, particularly in large river basins, and even more so where human-induced alterations in river hydrological regimes are increasingly becoming crucial with the intensive and extensive development of their water and hydropower resources. The investigation and assessment of human-induced impacts on river flow regimes could provide a vital basis for water resources management and river restoration. This is the aim of the present study, using the Hanjiang River in China as a case study. On the basis of the past several decades of daily flow discharge data collected by the Changjiang River Water Resources Commission, an attempt was made to quantitatively evaluate the spatio-temporal variations in annual, seasonal, monthly and daily runoff, focusing on the middle and lower reaches of the Hanjing River. This paper also investigates in detail the driving forces of flow regime alterations.

CASE STUDY SITES

With intensified human activities, the flow regime of the Hanjiang River has been altered to some extent. The focus of this paper is on the impacts of the Danjiangkou reservoir on the flow regime of the Hanjing River (Fig. 1). The Danjiangkou reservoir, the second largest in terms of storage capacity in the Yangtze River basin, is a seasonal-storage reservoir and was built in 1967. It is

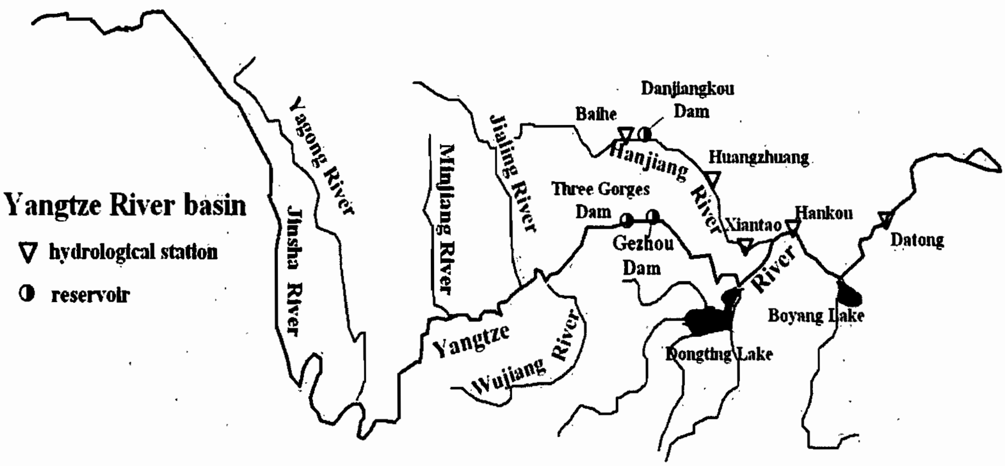


Fig. 1 Location map of hydrological stations and reservoirs.

located at the end of the upper reach of the Hanjiang, and its catchment area accounts for 60% of the total river basin. The reservoir has a normal pool level of 157 m and a capacity of 17.45 billion m^3. It is also the water source of the central route of China's South–North water transfer project. Currently the second-stage engineering works are underway with the aim of increasing the normal pool level to 170 m and the storage capacity to 29.05 billion m^3, and are expected to be completed by 2010. To assess the impacts of the Danjiangkou reservoir on the flow regime of the Hanjiang River, three key hydrological stations, i.e. Baihe, Huangzhuang and Xiantao were selected (Fig. 1). Baihe station, above the Danjiangkou reservoir, is the upper controlling station for water and sediment discharges to the Danjiangkou reservoir, located at the upper reach of the Hanjiang River. Huangzhuang is located at the end of the middle reach of the Hanjiang River, 223 km below the Danjiangkou reservoir. Xiantao, below the Danjiangkou, is the most downstream station on the Hanjiang River, and controls water and sediment discharges to the main stem of the Yangtze River from the Hanjiang.

DATA AND METHODS

The time series of daily flow discharge from 1955 to 2006 for three key hydrological stations were collected from the Yangtze River Water Resources Commission. The whole study period was divided into two sub-periods by the year when the reservoir began to store water (i.e. 1955–1966 and 1967–2006). On the basis of time series of daily flow discharge from the three stations, annual, wet season (May–October), dry season (November–April), monthly and daily runoff at the three stations for the two different sub-periods were computed and analysed.

RESULTS

The variations with time of annual, wet season and, dry season runoff at six stations are presented in Figs 2–4. The mean annual values of annual, wet season and dry season runoff and the percentage of wet season runoff in annual runoff at three stations in the two sub-periods were computed and are presented in Table 1. Figure 5(a) and (b) show the monthly runoff distributions at Baihe, Huangzhuang and Xiantao during pre-dam and post-dam periods. Figure 6(a)–(c) illustrates the frequency of daily discharges falling in different ranges at Baihe, Huangzhuang and Xiantao during the pre-dam and post-dam periods.

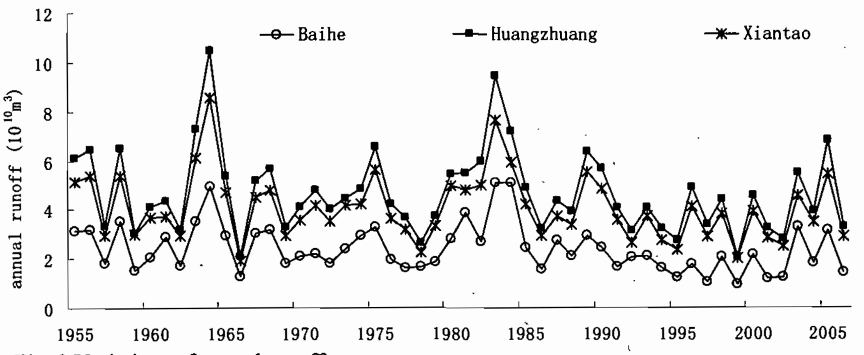


Fig. 2 Variations of annual runoff.

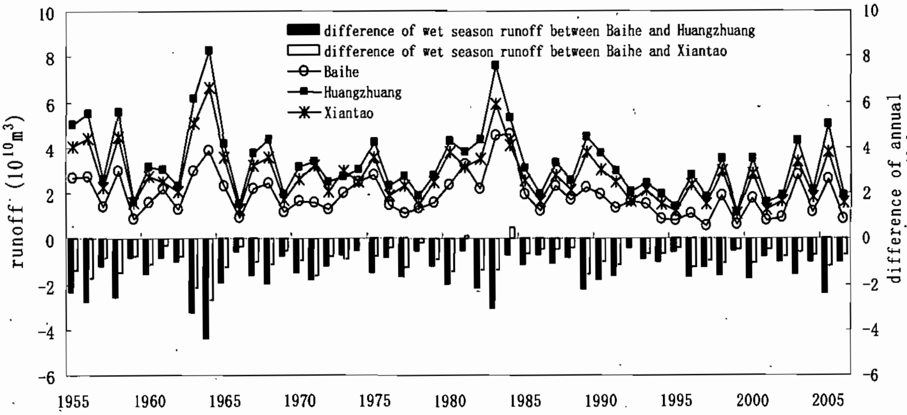


Fig. 3 Variations of wet season runoff.

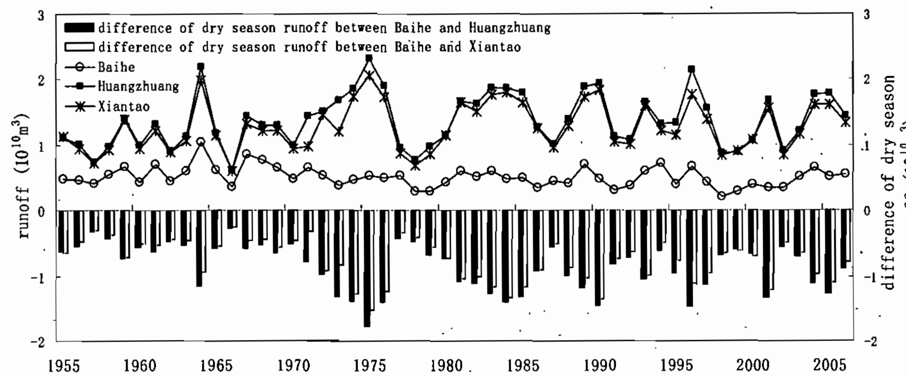
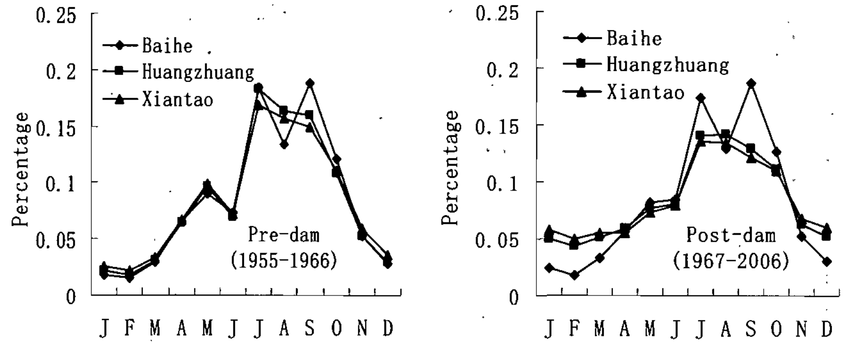


Fig. 4 Variations of dry season runoff.

DISCUSSION

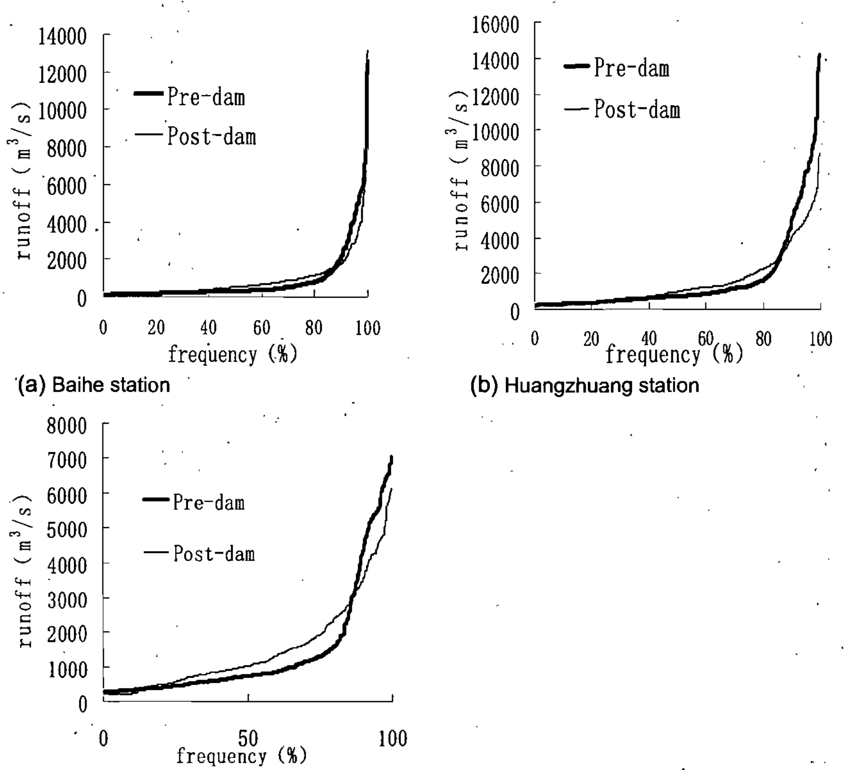
Annual runoff

From Fig. 2 it can be seen that annual runoff at Baihe, Huangzhuang and Xiantao showed similar patterns of variation, with runoff in the upper reach being less than that in the middle and lower reaches in the same year. This means that the annual runoff downstream is closely associated with the upstream, and that the runoffs from the middle and lower Hanjiang River basins are important



(a) Pre-dam (1955–1966) (b) Post-dam (1967–2006)

Fig. 5 Impacts of the Danjiangkou Reservoir on monthly runoff distribution.



(a) Baihe station (b) Huangzhuang station

(c) Xiantao

Fig. 6 Distribution of daily discharges.

components in the total runoffs at Huangzhuang and Xiantao, respectively. The annual runoff at Xiantao is less than that at Huangzhuang due to the offstream water use in the lower Hanjiang River basin. The fluctuations of annual runoff at the three stations corresponded well, but the degree of fluctuation of the annual runoff varies with length. The fluctuation of annual runoff was most significant at Baihe and least at Xiantao. Since the upper Hanjiang River basin is a hilly/ mountainous region, the annual precipitation varies significantly, and the fluctuation of the annual runoff at Baihe was more severe compared to Huangzhuang and Xiantao. It has been recognized

Table 1 Mean annual value at three hydrological stations in two sub-periods.

Station	Periods	Runoff ($10P^{10P}mP^{3P}$) Wet season	Dry season	Annual	% wet seasonal runoff in annual runoff
Baihe	1955–1966	2.14	0.57	2.71	79.1
	1967–2006	1.81	0.50	2.31	78.5
Xiantao	1955–1966	4.07	1.14	5.21	78.2
	1967–2006	3.09	1.44	4.53	68.2
Xiantao	1955–1966	3.37	1.08	4.45	75.7
	1967–2006	2.57	1.36	3.93	65.4

that the mean annual runoff at Baihe accounted for 52% of that at Huangzhuang and 59% of that at Xiantao during the pre-dam period, while during the post-dam period these values were 52% and 61%, respectively. It can be concluded that the Danjiangkou reservoir resulted in little alteration in the annual runoff characteristics of the Hanjiang River. Since the Danjiangkou reservoir is a seasonal-storage reservoir in terms of mean annual runoff, the storage capacity is not big enough to regulate the inter-annual distribution of runoff downstream. The Mann-Kendall statistic for the annual runoff at Baihe, Huangzhuang and Xiantao is –2.05, –1.26 and –1.52, respectively, during the study period, thus the conclusion can be drawn that the annual runoff at the three stations present a decreasing trend. The decreasing trend of annual rainfall is one of the major driving forces.

Seasonal runoff

Before 1967, wet season runoff both at Huangzhuang and Xiantao was generally higher than that at Baihe in the same year (Fig. 3), even more so in wet years. But from 1967 the difference in wet season runoff between Xiantao and Baihe and between Huangzhuang and Baihe in wet years significantly reduced (Fig. 3), and more so between Huangzhuang and Baihe. The dry season runoff at Huangzhuang and Xiantao presented different changing patterns (Fig. 4). Before 1967 the dry season runoff both at Huangzhuang and Xiantao was generally higher than that at Baihe in the same year, but from 1967 the difference between Xiantao and Baihe and between Huangzhuang and Baihe increased markedly, particularly in wet years, and more considerably between Huangzhuang and Baihe. This indicates that the operation mode of "retaining high floods in wet seasons" adopted by the Danjiangkou apparently affected the seasonal runoff distribution downstream by making the seasonal distribution more even. The percentage of wet season runoff in annual runoff at Xiantao, Huangzhuang and Baihe during the same periods (Table 2) also proved this variation pattern. Before 1967, the percentage of wet seasonal runoff at Baihe was 79.1%, 0.6% higher than 78.5% at Baihe during 1967–2006, but the percentages of wet season runoff at Xiantao and Huangzhuang before 1967 were 75.7% and 78.2% respectively, 10.3% and 10% higher than 65.4% and 68.2% at Xiantao and Huangzhuang during 1967–2006. This reflects that the operation of the Danjiangkou has resulted in a reduction in the percentage of wet season runoff and an increase in the percentage of dry season runoff downstream. It should be noted that in both wet and dry seasons, the seasonal runoff at Huanzhuang was generally greater than that at Xiantao due to water withdrawal from the lower reach of the Hanjing River for offstream water uses.

Monthly runoff

From Fig. 5 it can be seen that the Danjiangkou imposed a significant influence on monthly runoff distribution at Huangzhuang and Xiantao. Before 1967, monthly runoff at the three stations generally corresponded well, but from 1967 the percentage of monthly runoff in the wet season months at Huangzhuang and Xiantao clearly decreased due to the retention of high floods by the reservoir in the wet season, and the percentage in the dry season months evidently increased. This indicates that the operation of the Danjiangkou made monthly runoff distribution downstream more variable. Compared with Xiantao, Huangzhuang was affected more seriously owing to Huangzhuang being closer to the Danjiangkou than Xiantao.

Daily runoff

From Fig. 6(a) it can be seen that the distribution of daily runoff at Baihe was very similar during pre-dam and post-dam periods. There was little change in the distribution of daily runoff greater than 800 $m^3 s^{-1}$ and limited change in the distribution of daily runoff less than 800 $m^3 s^{-1}$. Since the operation mode of "retaining high floods in wet seasons" adopted by the Danjiangkou benefited the cutoff of flood peaks, frequencies of daily discharges of less than 6000 $m^3 s^{-1}$ and 8000 $m^3 s^{-1}$ correspondingly increased to 97% and 99%, respectively during the post-dam period at Huangzhuang, from 92% and 96% during the pre-dam period (Fig. 6(b)); frequencies of daily discharges less than 4000 $m^3 s^{-1}$ and 5000 $m^3 s^{-1}$ also increased to 92% and 97%, respectively, during the post-dam period at Xiantao, from 89% and 92% during the pre-dam period (Fig. 6(c)). With a frequency of 99%, daily runoff at Huangzhuang and Xiantao was 12 600 $m^3 s^{-1}$ and 6720 $m^3 s^{-1}$ during the pre-dam period, but significantly decreased by 41.2% and 12.8% to 7410 $m^3 s^{-1}$ and 5860 $m^3 s^{-1}$, respectively, during the post-dam period. This implies that the flow regime of the middle and lower Hanjiang River is becoming more and more dominated by lower discharges. As the runoff from the Huanzhuang-Xiantao river basin is an important component in the total runoff at Xiantao, the daily runoff at Huangzhuang was more affected by the operation of the Danjiangkou reservoir than the Xiantao.

It can be expected that more significant impacts will be induced by the Danjiangkou reservoir on the downstream annual, seasonal, monthly and daily runoff after the second-stage engineering works are completed in 2010, which aim to increase the normal pool level to 170 m and the storage capacity to 29.05 billion m^3 (Liu *et al.*, 2005).

CONCLUSIONS

This paper evaluated the impacts of the Danjiangkou reservoir on the flow regime of the Hanjiang River. The results revealed:

- the impacts of the reservoir on river flow regime vary with reservoir regulation capacity, reservoir operation pattern and the distance between target reservoir and study site;
- the operation mode of "retaining high floods in wet seasons" implemented by the Danjiangkou, did not affect annual runoff downstream, but led to a more even seasonal, monthly and daily runoff distribution downstream;
- water withdrawal and climate change resulted in variations in annual runoff;
- reservoir regulation, water withdrawal and extreme rainfall events were responsible for the variations in wet season, dry season, monthly and daily runoff.

The output of this paper could provide references for the integrated management of the Hanjiang River water resources and the assessment of dam-induced impacts on the health and stability of the Hanjiang River ecosystem.

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Impact of the Three Gorges Reservoir operation on the downstream ecological water use

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Abstract With population increase and economic growth, the flow regime of the Yangtze River has been altered to some extent by human activities, and particularly dam construction. Dam-induced alterations in the flow regime of the Yangtze River will unavoidably influence water allocation among different water uses and instream ecological water requirements may not be satisfied. To assess the impacts of the Three Gorges reservoir operation on the downstream minimum instream ecological water requirements, this paper selected the Three Gorges Reservoir and Yichang hydrological station below the reservoir as case study sites. On the basis of long-term time series of daily discharge data, the reservoir outflow was simulated under two water storing schemes and the degree to which the downstream minimum ecological flow was satisfied was computed. The output of this paper could provide references for the integrated management of the Yangtze River water resources and the assessment of dam-induced impacts on the Yangtze River ecosystem health.

Key words Yangtze River; Three Gorges Reservoir; Yichang hydrological station; minimum instream ecological flow

INTRODUCTION

The Yangtze River (Changjiang) is one of the most important rivers in the world, third largest in annual runoff (Xia *et al.*, 2007) and fourth largest in sediment load (Eisma, 1998). The Yangtze plays a critical role in the global water cycle, sediment cycle, energy balance, climate change and ecological development (Xia *et al.*, 2007). The alterations in its hydrological regime therefore have global-scale impacts. However, with population increase and economic growth, the flow regime of the Yangtze River has been altered by human activities, particularly dam construction. By the end of 2000, 134 large/medium-scale reservoirs had been built with a total storage capacity of 1.064×10^{11} m^3 (Zhang, 2003). Dam-induced alterations in the flow regime of the Yangtze River will unavoidably influence water allocation among different water uses and the instream ecological water requirements may not be satisfied. As a result, the river ecosystem's health and stability may be affected. Recently, study of dam-induced impacts on downstream ecological water use, and associated river ecosystem health and stability, has attracted worldwide attention, particularly in large river basins, and even more so where dam-induced alterations in river flow regime are increasingly crucial to the intensive and extensive development of their water and hydropower resources. The investigation and assessment of dam-induced impacts on downstream ecological water use could provide a vital basis for river restoration and water resources management. This is also the aim of the present study, with the Yangtze River in China as a case study. On the basis of the past several decades of daily flow discharge data collected from the Changjiang River Water Resources Commission, an attempt, focusing on the Three Gorges dam on the main stem of the Yangtze River, was made to quantitatively evaluate impacts of the Three Gorges Reservoir operation on the downstream ecological water flow.

CASE STUDY SITE

With intensified human activities, a large number of dams/reservoirs have been constructed on the Yangtze River, and the flow regimes downstream of them have been altered to some extent, and such that the instream ecological water requirements can not be guaranteed in some months, particularly during phases of reservoir storing water. The focus of this paper is on the impacts of the Three Gorges Reservoir operation on downstream instream ecological water requirements.

To quantitatively assess the impacts of Three Gorges Reservoir operation on the downstream instream ecological water use, the Three Gorges Reservoir and Yichang hydrological station below the reservoir were used as case study sites. Yichang station is the control point of the upper Yangtze River basin and located at the starting point of the middle reach of the Yangtze River, 44 km below the Three Gorges Dam and 6 km below the Gezhouba Dam. The Three Gorges Reservoir, with a capacity of 3.93×10^{10} m^3, spans the Yangtze River and is the largest hydroelectric power station in the world. Except for the ship lift, all the original project plan was completed on 30 October 2008, when the 26th generator was brought into commercial operation. The Gezhouba Reservoir, with a capacity of 1.58×10^9 m^3, is a run-of-river reservoir and located on the main stem of the Yangtze, 38 km below the Three Gorges Reservoir. A few studies reveal that the Gezouba Dam has limited impact on the downstream flow regime (Li *et al.*, 2007), and therefore the degree to which ecological flows are met at Yichang station is mainly affected by the Three Gorges Reservoir operation.

METHODS AND DATA

Given that different reservoir operation modes, particularly the reservoir water storing schemes, may result in different degrees of impacts on the degree to which the ecological instream flow is satisfied downstream, two reservoir storing water schemes were adopted in this paper. The satisfying degree of the ecological instream flow is defined as the ratio of river flow to instream ecological flow (Xia *et al.*, 2007). If the ratio is equal to or larger than 1, the ecological water requirements (Xia *et al.*, 2007) can be satisfied; otherwise the ecological water requirements cannot be met. The two reservoir storing water schemes are as follows:

Scheme I: water storing period is initially set from 21 September to 20 October, and the reservoir water level rises linearly to its normal pool level under the condition that the guaranteed output of power generation can be ensured. If the water level cannot reach its normal pool level by 20 October, the water storing period can be extended until that level is reached.

Scheme II: water storing period is initially set from 16 September to 15 October, and the reservoir water level rises linearly to its normal pool level under the condition that the guaranteed output of power generation can be ensured. If the water level cannot reach its normal pool level by 15 October, the water storing period can be extended until the level reaches normal pool level.

To obtain the reservoir outflow time series, the same reservoir inflow data time series was used to simulate reservoir outflow under the two different schemes. Because the daily inflow is unknown, the daily discharge data at Yichang before the Three Gorges Reservoir came into operation in 2003 was assumed to equal the total reservoir daily inflow. On the basis of the 1950–2002 time series of daily discharge data from Yichang station, the reservoir outflow processes were simulated for the two schemes. Since Chinese sturgeon spawning takes place in October and November each year, and is quite sensitive to flow magnitude, the study period is September, October and November of each year. According to the simulation results and the computed minimum instream ecological flow (Zou, 2008), the monthly satisfying degree of the minimum instream ecological flow at Yichang for September, October and November was computed; the mean daily satisfying degree of the minimum instream ecological flow at Yichang was calculated for five types of year (i.e. extremely wet, wet, average, dry and extremely dry years); the frequency of river flow less than the minimum instream ecological flow in September, October and November was computed each of the five types of year. The computation results for the two schemes were compared.

RESULTS

Table 1 presents the frequencies of river flow less than the minimum instream ecological flow in September, October and November for the five different types of year under two different water

Table 1 Frequency of river flow less than the minimum instream ecological flow (%).

Scheme	Month	EWY	WY	AY	DY	EDY
I	September	4.8	5.4	16.7	25.5	42.2
	October	23.5	24.8	26.6	50.0	64.0
	November	2.9	2.6	1.8	10.7	7.2
II	September	6.7	7.9	20.8	33.1	48.9
	October	16.1	15.4	16.9	36.6	52.2
	November	2.9	2.6	1.8	10.7	7.2

EWY: Extremely wet years; WY: wet years; AY: average years; DY: drier years; EDY: extremely dry years.

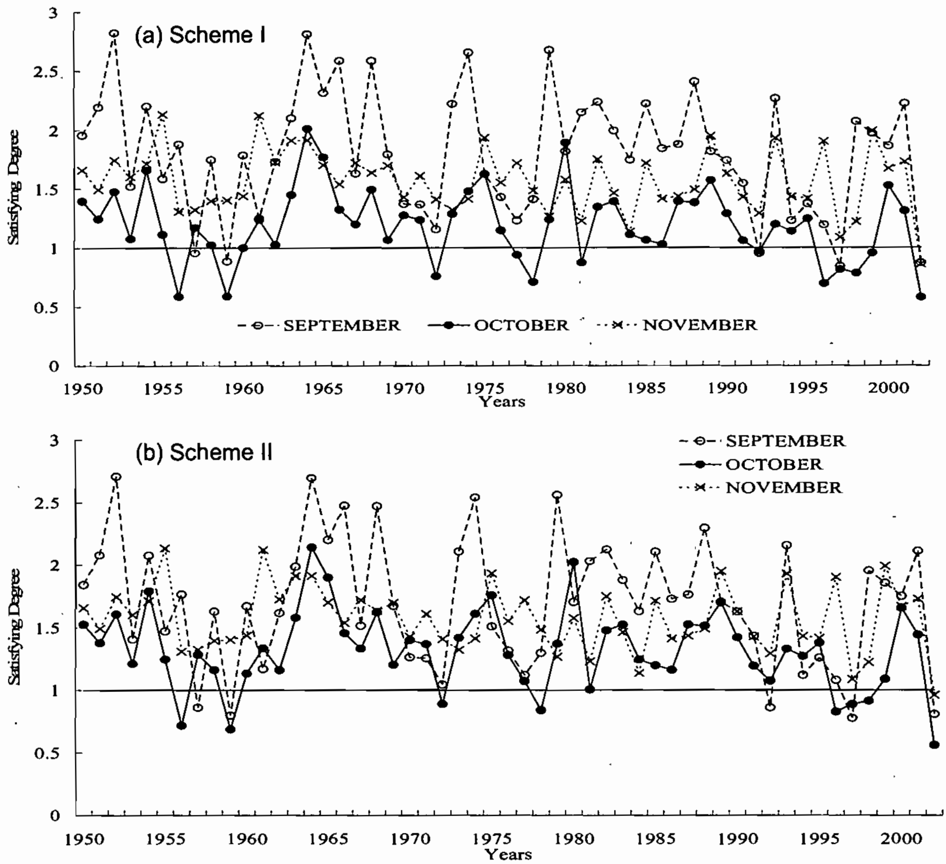

Fig. 1 Monthly satisfying degree of the minimum ecological flow for different years.

storing schemes. Figure 1(a) and (b) shows the monthly satisfying degree of the minimum instream ecological flow in September, October and November of each year for both water storing schemes. Figure 2(a)–(c) and Figure 3(a)–(b) demonstrate the mean daily satisfying degree of the minimum instream ecological flow in September and October, respectively, for the five types of year for both water storing schemes. Since reservoir water storing did not take place in November, the variation patterns of the mean daily satisfying degree in November for all types of years are the same under both water storing schemes; see Fig. 2(c).

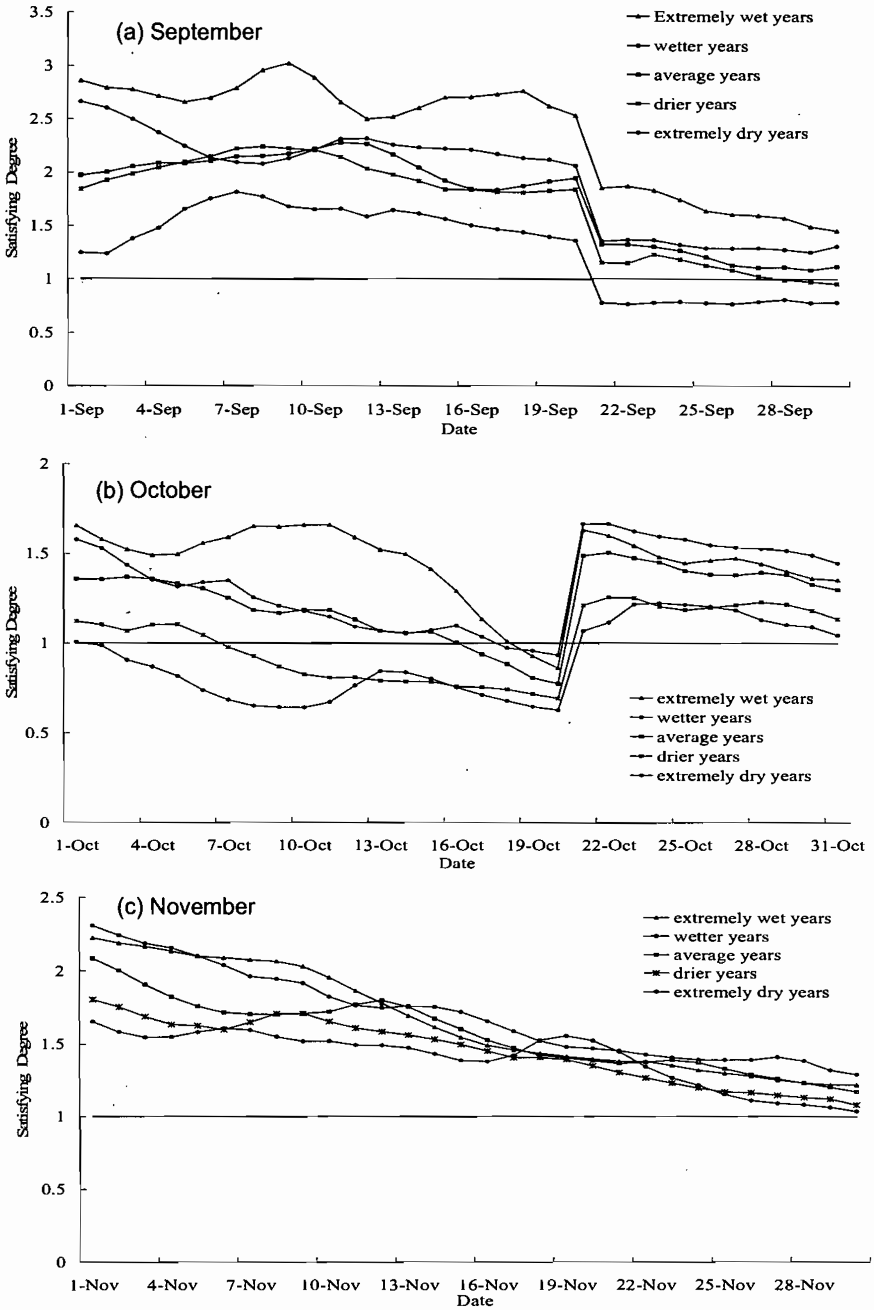

Fig. 2 Mean daily satisfying degree of the minimum ecological flow in 5 types of years (Scheme I).

DISCUSSION

Different water storing schemes adopted by the Three Gorges Reservoir impose different impacts on the downstream ecological water use, therefore, it is necessary to analyse the impacts under different water storing schemes and compare them.

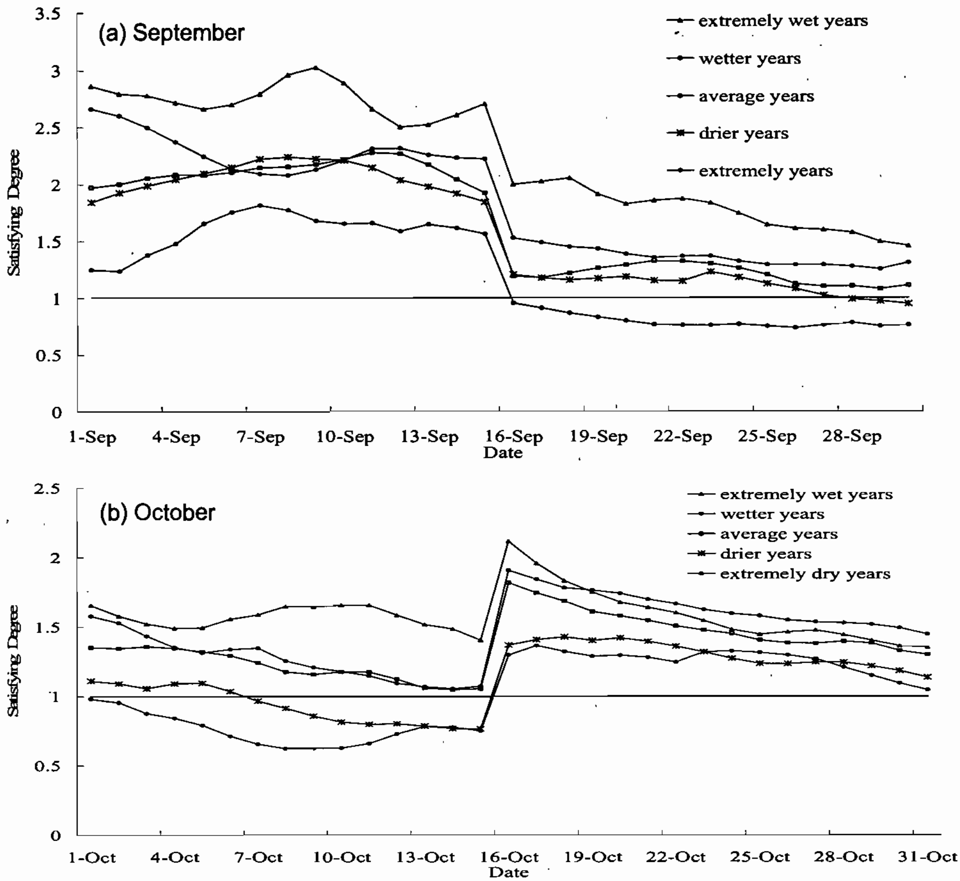

Fig. 3 Mean daily satisfying degree of the minimum ecological flow in five types of years (Scheme II).

Scheme I

From Table 1 it can be seen that the frequency of river flow less than the minimum instream ecological flow in September in drier years were higher than those in wetter years. The same situation occurs in October and November, revealing that the impacts induced by reservoir operation on the downstream ecological water use vary with reservoir inflow conditions. If more water flows into the reservoir from the upstream, the downstream ecological water use can be better guaranteed. Table 1 also shows that for the same type of year the frequency of river flow less than the minimum instream ecological flow in October was generally significantly higher than that in September, and the value in September obviously higher than that in November. This is because more days for reservoir storing water fell in October than in September. No water storing took place in November; thus, compared with October and September, the frequency of river flow less than the minimum instream ecological flow in November was lowest for the same type of year. It can be concluded that the influence of reservoir water storage on the downstream ecological water use is closely associated with reservoir inflow and water storage.

Since the monthly reservoir inflow in September was apparently larger than that in October and November, the monthly satisfying degree of the minimum instream ecological flow in September was generally higher than that in October and November during 1950–2002 (Fig. 1(a)). This indicates that reservoir inflow plays a more critical role in determination of the monthly satisfying degree in September than reservoir water storing, although in a number of years the value was less than 1 due to reservoir water storing. Since November and October fall within the dry season and wet season, respectively, the monthly reservoir inflow in October was obviously larger in all years than that in November, but the monthly satisfying degree was mostly lower than that in November due to reservoir storing water. In comparison with September, the influence of reservoir water storing on the monthly satisfying degree evidently increased in October.

Figure 2(a)–(c) shows that the mean daily satisfying degree was lower than 1 only in extremely dry years in some days of September, while the value in all other types of year was higher than 1 or slightly lower than 1; the mean daily satisfying degree in some days of October for all types of years was lower than 1, and in drier years and extremely dry years the number of days with a value less than 1 increased significantly; the mean daily satisfying degree in November for all types of years was all higher than 1. To satisfy the downstream minimum ecological water requirements, more water should be released on days when the daily satisfying degree is lower than 1.

Scheme II

Similar conclusions were obtained under Scheme II for the frequency of river flow less than the minimum instream ecological flow the (Table 1); however, the difference was that the frequencies of river flow less than the minimum instream ecological flow in September under Scheme II increased for all types of year owing to the water storing time being advanced while the values in October declined for all types of year because the number of reservoir storing water days decreased.

The results for monthly satisfying degree of the minimum instream ecological flow, Fig. 1(a)– (b) demonstrate similar variation of the monthly satisfying degree in September, October and November for both schemes. The distinction was that the monthly satisfying degree in September under Scheme II decreased in all years due to advanced water storing, while the value in October increased in all years because the time for storing reservoir water was shortened, and, as a result, the ecological water requirements could be better met. Compared with Scheme I, the number of years evidently declined with the monthly satisfying degree in September being higher than that in October or November due to water storing time having been advanced.

The daily data, Fig. 3(a), illustrate a similar variation of mean daily satisfying degree in September for all types of year as in Fig. 2(a), but the number of days with a value less than 1 increased in extremely dry years because of advanced water storing. After Scheme II was adopted, the situation improved in October for all five types of years, particularly in drier and extremely dry years, as the days when downstream minimum ecological water requirements could not be met diminished (Fig. 3(b)).

CONCLUSIONS

This paper evaluated the impacts of the Three Gorges Reservoir operation on the downstream ecological water use. The results reveal the impacts varied with the reservoir water storing schemes adopted and inflow conditions. Compared with Scheme I, Scheme II could obviously improve the situation that the downstream minimum ecological water requirements could not be met in October. This will benefit Chinese sturgeon spawning. Therefore, if flood safety can be ensured in September, advancing the reservoir water storing time from 21 September–20 October to 16 September–15 October is a better choice, and could benefit the river ecosystem health and stability more.

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Effects of the South–North Water Transfer Project on surface water and groundwater resources in Haihe River basin in a climate changing world

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Abstract This paper addresses the effects of the South–North Water Transfer Project (SNWT) in China on surface water and groundwater resources in Haihe River basin under conditions of climate change. An integrated model of SWAT with a simplified groundwater model was developed. Impacts on groundwater and surface-water of the Haihe River basin are calculated under different climate change scenarios. Nine types of scenario are assumed in this paper, consisting of combinations of climate change scenarios and water transfer schemes of the SNWT Project. The application shows that: (1) under current and changing climate scenarios, transferring 122 and 143 billion m^3 of water, respectively, are the best ways to relieve water shortage; (2) nearly 90% of the water transferred from the south is consumed by evapotranspiration under current land use conditions; (3) even when the full diversion capacity is used to transfer water, water shortages will still be acute under the worst case climate scenario.

Key words integrated surface-water and groundwater model; Haihe River basin, China; climate change; South-to-North Water Transfer Project

INTRODUCTION

Human activities and climate variability are perturbing the global water cycle in ways that societies have never experienced. Changes in the water cycle impose substantial stresses to regional economic development and ecological environments. Although significant progress in evaluating the impacts of climate change and human activities on surface hydrology and associated ecosystems has been made (Legessea *et al.*, 2003; Dibike & Coulibaly, 2005), little is known about how subsurface water in the vadose zone and aquifers might respond to them (Green *et al.*, 2007).

The Haihe River basin is one of the areas suffering severe shortages of water resources in the north of China. Owing to large-scale human activity, particularly overexploitation of groundwater, 40% of the rivers in the Haihe River basin have become seasonal rivers and more than 70% of the watershed area is overexploited at present (Xia *et al.*, 2007, 2008). In the 1990s, the flow into the sea had decreased by 10% compared with flow in the 1950s. In order to relieve the stress caused by water resources scarcity in the Haihe River basin, the South–North Water Transfer (SNWT) Project will be implemented in 2010. The Haihe River basin is the receiving area of the central route of the SNWT Project. Much research has been done to analyse the effects of the SNWT project on hydrology and ecosystems (Yang *et al.*, 2005); however, the analyses seldom consider groundwater and the effects of climate change. It has been indicated that global warming will cause acceleration of the water cycle and increase the chance of extreme climate events occurring (China Meteorological Newspaper Office, 2007). There is a high probability of flooding in the Haihe River basin in the future (Fei *et al.*, 2005). Thus, it is urgent to analyse the expected impacts of the SNWT project on surface water and groundwater resources in the Haihe River basin simultaneously under changing climate conditions.

Interactions between surface water and groundwater are becoming more complicated and important in a changing world. Single hydrological models focusing on surface water or hydrogeological models focusing on groundwater dynamics are not sufficient to simulate surface water and groundwater as a continuous system. Although many hydrological models include the simulation of baseflow, they treat the aquifer system as a black box. Hydrogeological models deal with their boundary conditions such as rainfall–infiltration recharge (an important input) in a very cursory way. Developments in distributed hydrological modelling and the "3S" technologies (remote sensing, geography information systems, global positioning systems), support integration of surface water and groundwater models. The integrated models can generally be categorized into two types: the first are models such as SWAT-MODFLOW (Perkins *et al.*, 1999; Kim, *et al.*, 2008), HSPF-MODFLOW (Said *et al.*, 2005; Mark *et al.*, 2005), and GSFLOW (Markstrom *et al.*, 2008), that link distributed conceptual hydrological models with numerical groundwater models through conversion and feedback of common variables. The second type of models, such as MIKESHE (Refsgaard & Storm, 1995), MODMHS (Panday & Huyakorn, 2004), and InHM (Loague *et al.*, 2005), utilize partial differential equations of water movement to describe hydrological processes.

The second kind of model tends to be more complicated, time-consuming, and requires a larger number of data. As such, this type of model can not be applied to the Haihe River which has a large basin area and for which detailed data about the vadose zone and aquifers are difficult to obtain. MODFLOW is also difficult to apply in such a large area. Therefore, this paper combined SWAT with a groundwater model (GWM), which has a similar theoretical background to MODFLOW, to evaluate the impacts of climate change and large-scale human activity on groundwater and surface water flow in the basin.

DEVELOPMENT OF THE INTEGRATED SURFACE WATER AND GROUNDWATER MODEL

Overview of SWAT and GWM

"*SWAT represents a watershed as a collection of subbasins and simulates water budgets for the soil profile and pond storage in each subbasin with a daily time step. SWAT itself is a comprehensive watershed model that includes features to represent groundwater and base flow, but it is of limited use in representing stream-aquifer interactions, dynamic groundwater table*" (Perkins & Sophocleous, 1999).

The GWM can be used to solve this problem. GWM calculates groundwater levels based on the continuity equation and Darcy's law. It divides the aquifer system into many grid cells with the same size. Letters i, j, and k are used to indicate the row, column, and layer of the position in a grid. Influx from cell $(i, j-1, k)$ to cell (i, j, k) can be described as:

$$q_{i,j-\frac{1}{2},k} = CR_{i,j-\frac{1}{2},k}(h_{i,j-1,k} - h_{i,j,k}) = \cdot (CR_{i,j,k} + CR_{i,j-1,k})/2 \cdot (h_{i,j-1,k} - h_{i,j,k}) \quad (1)$$

where: $q_{i,j-\frac{1}{2},k}$ represents inflow from cell $(i, j-1, k)$ to cell (i, j, k); $CR_{i,j,k}$, $CR_{i,j-1,k}$ and $CR_{i,j-\frac{1}{2},k}$ represent the hydraulic conductivity of cell (i, j, k), cell $(i, j-1, k)$ and the hydraulic conductivity between them; $h_{i,j,k}$, $h_{i,j-1,k}$ represent the groundwater level of cell (i, j, k) and cell $(i, j-1, k)$.

Inflows from other adjacent cells are quantified in the same way. The water balance of cell (i, j, k) can be described as follows:

$$\begin{aligned} &CR_{i,j-\frac{1}{2},k}(h_{i,j-1,k} - h_{i,j,k}) + CR_{i,j+\frac{1}{2},k}(h_{i,j+1,k} - h_{i,j,k}) + CC_{i,j-\frac{1}{2},k}(h_{i-1,j,k} - h_{i,j,k}) \\ &+ CC_{i,j+\frac{1}{2},k}(h_{i+1,j,k} - h_{i,j,k}) + CV_{i,j,k-\frac{1}{2}}(h_{i,j,k-1} - h_{i,j,k}) + CV_{h,j,k+\frac{1}{2}}(h_{i,j,k+1} - h_{i,j,k}) \\ &+ QS_{i,j,k} = Q_{total,i,j,k} = SS_{i,j,k}(\Delta r_j \Delta c_i \Delta v_k) \cdot \Delta h_{i,j,k} / \Delta t \end{aligned} \quad (2)$$

where: Δr_j, Δc_i, Δv_k represent distances between two adjacent rows, columns, or layers; CR, CC, CV represent the hydraulic conductivity between two adjacent cells along the row, column, or layer; $QS_{i,j,k}$ represents other kinds of recharge or discharge rate, including precipitation; $Q_{total,i,j,k}$ represents total net recharge rate; $SS_{i,j,k}$ represents storage capacity; Δt is the time step, and $\Delta h_{i,j,k}$ represents the increment of the groundwater level, which is calculated according to:

$$\Delta h_{i,j,k} = Q_{total,i,j,k} \cdot \Delta t \Big/ A_{i,j,k} \cdot u_{i,j,k} \tag{3}$$

Then $h_{i,j,k}$ of the next time step can be calculated according to the initial groundwater level and equation (3).

Structure of the integrated model

In the integrated model, GWM replaces the computation of baseflow in SWAT as a subroutine; SWAT is split into two parts, before and after the GWM subroutine. SWAT simulates processes in the vegetation canopy, the soil profile, and pond storage for each subbasin, while GWM simulates processes in streams and aquifers for each cell. After the surface water and soil water components are simulated by the first part of SWAT, rainfall–infiltration recharge and groundwater evapotranspiration of the hydrological response unit (HRU) are formulated. Then they are transformed from the HRU to grid cells, and GWM is "called" by SWAT to simulate groundwater levels and the river–aquifer exchanges based on these boundary conditions. Subsequently, the river–aquifer exchanges of cells are transformed to stream inflow of the corresponding subbasins in the second part of SWAT to calculate the outflow.

HRU–cell conversion method

In this method, HRU numbers are assigned by combining spatial soil and land-use attributes. To match HRUs in SWAT and grid cells in GWM, the HRU distribution map and the aquifer grid cell distribution map are overlayed using ArcGIS. Coordinates of grid cells in the GWM are associated with the number of the HRU that covers the grid cells. The aquifer grids outside of the boundary are defined as zero. Then each HRU is associated with its corresponding grid cells, and the common variables can be transferred between the HRU and corresponding GWM grid. For example, subbasin 1 is divided into HRU 1 and HRU 2. The groundwater recharge or groundwater evapotranspiration of HRU 1 in subbasin 1 is distributed to grids covered by HRU 1, while the volume of river–aquifer exchange added to the streams in subbasin 1 equals the cumulated river–aquifer exchange of all grids covered by the two HRUs in subbasin 1.

River–aquifer exchange

To match the river with the aquifer cells under it, the river network in the Digital Elevation Model is registered by the GWM. GWM distinguishes the relationship between the river and aquifer automatically, and calculates the exchange between river and aquifer based on the concept of "vertical flow" which is similar to the theory in the MODFLOW RIV package. It assumes that there is a saturated, low infiltrative layer at the bottom of the river bed. The exchange between the river and aquifer cell is quantified according to Darcy's law as follows:

$$\begin{aligned} q &= \frac{K'}{b'} B(Z - h) && \text{if} \quad h > RBOT \\ q &= \frac{K'}{b'} B(Z - RBOT) && \text{if} \quad h \le RBOT \end{aligned} \tag{3}$$

where K' is hydraulic conductivity; h is surface water stage; Z is water table head; B is top width of channel; b' is layer thickness; $RBOT$ is elevation of river bed. Then the stream gain (if $q > 0$) or stream loss (if $q < 0$) comes from corresponding cells and is added to the corresponding reach.

CONSTRUCTION OF INPUT DATA FOR INTEGRATED MODEL

The integrated model was applied to the Haihe River basin (area 310 000 km^2). The surface water modelling requires information on weather, topography, soils, land use, reservoir management, stream channels, etc. Given the flatness of the river plain and the interlaced artificial channels of the Haihe River basin, the river network that was extracted on the basis of the DEM (90 m × 90 m) was modified according to the actual river network (1:250 000), and subsequently was put into AVSWAT2000. In total, 48 reservoirs and 17 estuaries were assigned, and the basin was divided into 283 sub-basins. A total of 1820 HRUs were formed by overlaying the MODIS land-use map (250 m × 250 m) and soil association map (1:1 000 000).

The GWM was developed for the plane of the Haihe River basin (the north China plain), which has an area of 129 000 km^2. The north boundary of the research area receives lateral recharge from the adjacent mountains. The south boundary receives lateral leakage from the Huanghe River. The east border with the Bohai Sea is taken as a fixed water head boundary. The groundwater modelling requires information on groundwater level, lateral inflow through the boundaries, pumping wells, groundwater recharge, groundwater evapotranspiration, river networks and river stage, etc. In the GWM, the shallow aquifers are represented as one layer, discretized into grids of 149 rows and 146 columns, with a cell-size of 4 km × 4 km. The groundwater level was obtained by interpolating data from 171 groundwater monitoring wells and the distribution of hydraulic conductivity and storage capacity was provided by the Haihe River Water Conservancy Commission. Groundwater recharge and groundwater evapotranspiration were simulated by SWAT. The information on pumping wells and lateral recharge were obtained from investigations in each prefecture or county.

MODEL CALIBRATION AND VALIDATION

The daily streamflow and groundwater table from 1996 to 2000 were calibrated against the measured daily streamflow and groundwater table. In SWAT, the primary calibration parameters were the SCS curve number, average slope steepness, amount of water in the soil profile and Manning's "n" value for overland flow. In GWM, the primary calibration parameters were the aquifer hydraulic conductivity, the storage capacity and riverbed conductance. The amount of groundwater pumping and the type of recharge to the aquifers also need to be adjusted.

Parameters were optimized by a trial-and-error procedure until the differences between the observed and simulated streamflows, the differences between evaluated and simulated volume of water flowing into the sea, and the differences between observed and simulated groundwater table are smallest. Comparisons of the observed and simulated monthly streamflow at some gauging stations in 1996–2000 are shown in Fig. 1.

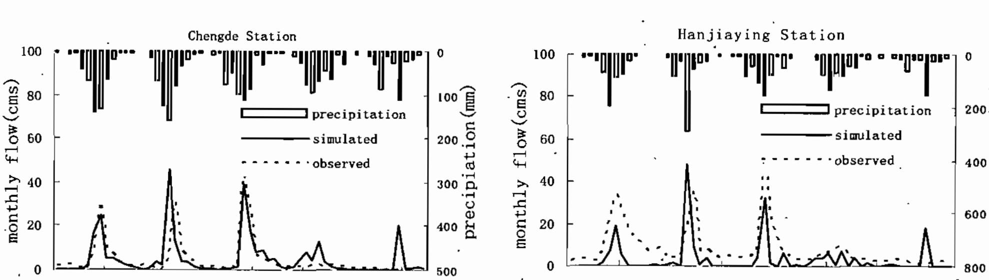
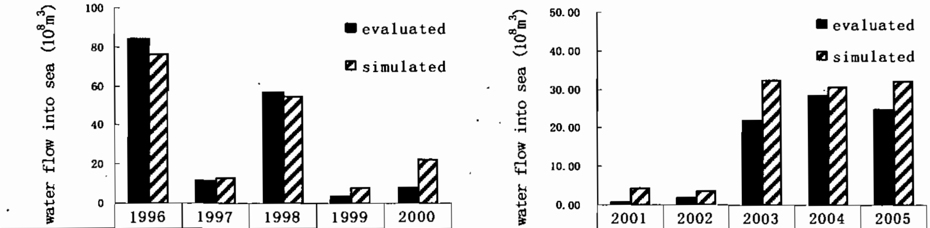


Fig. 1 Calibration results for Chengde and Hanjiaying gauging stations.



	1996	1997	1998	1999	2000
evaluated	84.02	11.36	56.66	3.45	8.01
simulated	76.6	13.04	54.45	7.69	22.12

	2001	2002	2003	2004	2005
evaluated	0.82	1.83	21.83	28.50	24.86
simulated	4.37	3.57	32.33	30.52	32.00

Fig. 2 Simulation results of inflows to sea in calibration and validation period.

Figure 2 shows the comparisons of the simulated and evaluated amounts of water flowing into the Bohai Sea over the calibration period. The evaluated amounts of water flow into the Bohai Sea from 1996 to 2000 were obtained from the comprehensive water resources assessment of the Haihe River basin in 2004.

From Figs 1 and 2 we can see that the amounts and trends of simulated streamflow at gauging stations and water flow into the Bohai Sea are reasonable. Owing to the regulation of many reservoirs in upstream areas and the interlaced artificial channels in the plain, the simulated streamflow hydrographs did not fit very well with the observed values, especially during peak flows.

In order to evaluate the performance of the calibrated model, daily streamflows and groundwater levels were simulated during 2001–2005. The simulation results over the validation period are shown in Fig. 3.

Application to the Haihe River basin indicates this integrated model is able to simulate surface water flow, groundwater flow, and river–aquifer interactions as a continuous system.

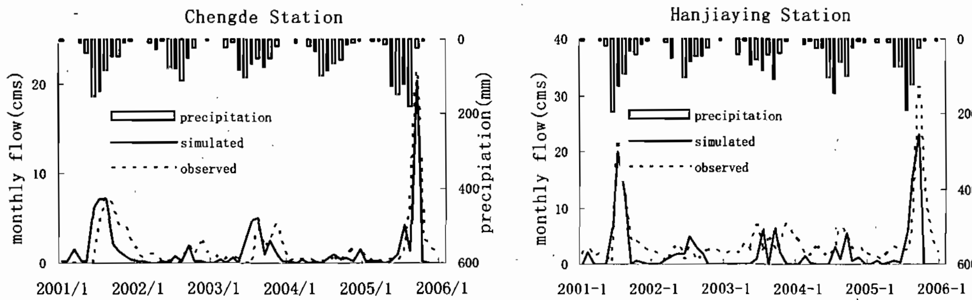


Fig. 3 Validation results of Chengde and Hanjiaying gauging stations.

SCENARIOS ANALYSIS

Assumptions in scenarios

Some reports indicate that the annual mean surface temperature of Haihe River basin is anticipated to increase by 1.0–3.5°C in the next 50 years under the SRES B2 scenario. Annual mean precipitation in the Haihe River basin may be reduced by 8%, or increase by up to 12% in the next 50 years under the SRES B2 scenario (National Climate Center, 2008). Three climate scenarios are assumed in this research: current climate scenario (observed temperature and precipitation data in 2005), the best future climate scenario (ΔT = 1.0°C, ΔP = 12%), and the worst future climate scenario (ΔT =3.5°C, ΔP = –8%). Three water transfer schemes for the central route of the SNWT Project are considered. These scenarios respectively transfer about 95 billion m^3 (smallest diversion), 122 billion m^3 (medium diversion), and 143 billion m^3 (the largest diversion) of water from the Danjiangkou Dam on the Hanjiang River. This water will serve domestic and industrial water needs in Beijing, Tianjin, and some cities in Hebei and Henan province.

In total, we used nine combined scenarios (Table 1) to analyse the impact of different future climate change and water transfer schemes of the SNWT Project on groundwater and surface water in the Haihe River basin using the calibrated coupled surface water/groundwater model. In the modelling, the diversions of 95 billion m^3, 122 billion m^3 and 143 billion m^3 were distributed to individual reservoirs according to design specifications.

Results of scenarios analysis

The simulated results under all scenarios are shown in Table 1. Table 2 shows the volume of transferred water converted to evapotranspiration, soil water, inflow to sea, and groundwater.

From Table 1 and Table 2 we can see that:

(a) In Table 2, because the climate conditions do not change, the total variation of water in evapotranspiration, soil moisture, water flowing into sea and groundwater storage equals the

Table 1 Simulation results under different scenarios.

Scenarios	Evapotranspiration		Soil moisture		Water flow to sea		Groundwater storage change	
	V (10^9m^3)	C (%)	V (10^8m^3)	C (%)	V (10^8m^3)	C (%)	V (10^8m^3)	C (%)
Current scenario (D=0, ΔT=0Δ,P=0)	131	0	75	0	32	0	–36	0
Scenario 1 (D=95, ΔT=0, ΔP=0)	139	6	79	5	33	3	–33	10
Scenario 2 (D=95, ΔT=3.5, ΔP= –8)	134	2	60	–20	24	–5	–41	–13
Scenario 3 (D=95, ΔT=1, ΔP=12)	150	14	88	17	42	31	–32	13
Scenario 4 (D=122, ΔT=0, ΔP=0)	142	8	80	7	34	6	–31	16
Scenario 5 (D=122, ΔT=3.5, ΔP= –8)	137	5	61	–19	25	–22	–40	–9
Scenario 6 (D=122, ΔT=1, ΔP=12)	152	16	89	19	43	34	–30	17
Scenario 7 (D=143, ΔT=0, ΔP=0)	144	10	81	8	34	6	–34	8
Scenario 8 (D=143, ΔT=3.5, ΔP= –8)	139	6	62	–17	25	–22	–40	–8
Scenario 9 (D=143, ΔT=1, ΔP=12)	154	18	91	21	44	38	–29	21

D: the volume of water to be transferred (10^8 m^3); ΔT: the change of annual mean surface temperature (°C); ΔP: the change of annual mean surface temperature (%); V: the annual mean value; C: the increase or decrease between the value in a scenario and value in current scenario divide the value in current scenario.

Table 2 Comparison of the percentage of transferred water converts to other components.

Scenarios	Evapotranspiration		Soil moisture		Water flow to sea		Groundwater storage change	
	V (10^9m^3)	R (%)	V (10^8m^3)	R (%)	V (10^8m^3)	R (%)	V (10^8m^3)	R(%)
Scenario 1	139	91	79	4.2	33	1.1	–33	4.2
Scenario 4	142	89	80	4.1	34	1.6	–31	4.8
Scenario 7	144	92	81	4.2	34	1.4	–34	2.1

R: the increase between the value in scenarios with different water transfer schemes but without climate change and values in the current scenario divided by the volume of water to be transferred.

diversion capacity of water. Respectively, 89–92%, 4.1–4.2%, 1.1–1.6% and 2.1–4.9% of the diverted water will be converted into evapotranspiration, soil moisture, inflow to sea and groundwater under current land use, crop structure and irrigation ways.

(b) Under current climate conditions, the percentage of transferred water converted into evapotranspiration in Scenario 4 is the smallest, the percentages of transferred water converted into inflow to the sea and groundwater are the largest and the percentage of transferred water converted to soil water is similar to that under scenarios 1 and 7 (Table 2). Simulation results in Table 1 show that groundwater storage increases most under Scenario 4, and the variations of soil moisture and water flow to the sea under Scenario 4 are approximately similar to the values under scenarios 1 and 7.

(c) Compared to the current scenario, evapotranspiration increases under all scenarios with a range from 2.68% to 17.89%. In scenarios with the same climate conditions (scenarios 2, 5 and 8), evapotranspiration increases with the increase in transferred water. In scenarios with the same water transfer schemes (scenarios 4, 5 and 6), evapotranspiration increases with increasing precipitation. As shown in the third column of Table 2, approximately 90% of the water increment becomes evapotranspiration.

(d) Comparing results of scenarios 1, 4 and 7 in Table 1 under current climate conditions, soil moisture increases by 5.3–8.0%, and inflow to the sea increases by 3.1–6.3% with the diversion of 95 billion m^3, 122 billion m^3 and 143 billion m^3 of water. Groundwater storage increases by 13–20%. Comparing the results of scenarios 3, 6 and 9 in Table 1, under the expected best climate condition of the Haihe River basin in the next 50 years, soil moisture increases by 7.6–15.7%, inflow to the sea increases by 31–38%, and groundwater storage increases by 13–20% with the diversion of 95 billion m^3, 122 billion m^3, and 143 billion m^3. However, under the worst case climate scenario, soil moisture decreases by 20–17%, inflow to the sea decreases by 25–22% and variation of groundwater storage decreases by 13–9%.

Comparing the size of increase or decrease caused by the water diversion and climate change separately, we can see that climate takes an important role in the variation of soil water, groundwater and water flow to the sea under the current land use, crop structure and irrigation techniques. In addition, under the predicted worst climate condition, even if the largest amount of water (143 billion m^3) is transferred, soil moisture, inflow to the sea and groundwater storage will still decline.

CONCLUSIONS

This paper proposed an integrated surface water and groundwater model to analyse the effects of the SNWT project and climate change on surface water and groundwater in the Haihe River basin. Application indicates that this integrated model is capable of simulating surface water flow, groundwater flow, and river–aquifer interactions as a continuous system. Scenario analysis shows:

(a) Under current climate and land use conditions, transferring 122 billion m^3 of water in three kinds of water transfer schemes is the best way to increase groundwater storage and flow into the sea. Under the predicted best and worst climate scenarios, and current land use conditions, transferring 143 billion m^3 of water is the best way to relieve water shortages in Haihe River basin.

(b) Under current land use, crop structure and irrigation techniques, nearly 90% of the water transferred from the Danjiangkou dam is consumed by evapotranspiration. Little water contributes to soil water, groundwater storage and flow into the sea. In order to make better use of the SNWT Project, effective measures such as adjusting crop structure and changing irrigation techniques must be implemented to reduce evapotranspiration in the Haihe River basin.

(c) Because most of the water is consumed by evapotranspiration, soil water, groundwater storage and flow into the sea are mainly affected by climate change. Under the worst case climate scenario, even if the largest diversion capacity (143 billion m^3) of water is transferred, water shortage will still be acute.

Of course, there are many aspects that need further research. For example, the effect of the SNWT project on climate should also be considered, the conversion between HRU and grid cells needs to be improved, as does the prediction of future land use, crop structure and irrigation techniques.

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Changes in the reliance on groundwater *versus* surface water resources in Asian cities

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Abstract Changes in the relative reliance on surface water and groundwater water resources, due to land-use/cover changes in Tokyo, Osaka and Seoul, to water law and institutions in Osaka and Bangkok, and from the point of view of climate change in Taipei, have been analysed. Urbanization causes a reduction of groundwater recharge and increase in thermal transfer into the subsurface environment. The regulation of groundwater pumping due to serious land subsidence did not work in Bangkok, in the absence of alternative water resources, in particular because the private sector owns groundwater. The price of water is another major factor for the change in the relative importance of groundwater and surface water for reliable water resources.

Key words urbanization; groundwater; climate change; water resources; Asia; land cover change; law

INTRODUCTION

Subsurface environmental problems, such as land subsidence due to excessive pumping, subsurface thermal anomalies, and groundwater contamination, have occurred repeatedly in major Asian cities with a time lag depending on the development stage of urbanization (Taniguchi *et al.*, 2009a). Therefore, we may be able to assess future scenarios if we can evaluate the relationships between subsurface environmental problems and the development stage of the city. Changes in the relative importance of groundwater and surface water for providing reliable water resources has occurred in many Asian cities depending on the development stage of urbanization (Landis, 2003; Lieberman, 2003; Ness, 2004; Taniguchi *et al.*, 2009a). Although the surface water is relatively easy to evaluate, evaluating the changes in the subsurface environment, including groundwater, remain a difficult task. In this study, changes in land-cover/use have been evaluated as factors controlling subsurface environments. Effects due to legal and institutional arrangements on groundwater management have been also evaluated.

STUDY AREA AND METHODS

In order to evaluate the relationship between development stage of the city and various subsurface environments in Asia, four subjects: urban, groundwater, subsurface heat, and subsurface contamination have been chosen, and intensive field observation and data collection were made in selected basins, including Tokyo, Osaka, Bangkok, Jakarta, Manila, Seoul and Taipei (Taniguchi, *et al.*, 2009a,b, Fig. 1). Effects of land-use/cover changes on subsurface environments have been evaluated in Tokyo, Osaka, and Seoul in this study. The effects of laws and institutions on change

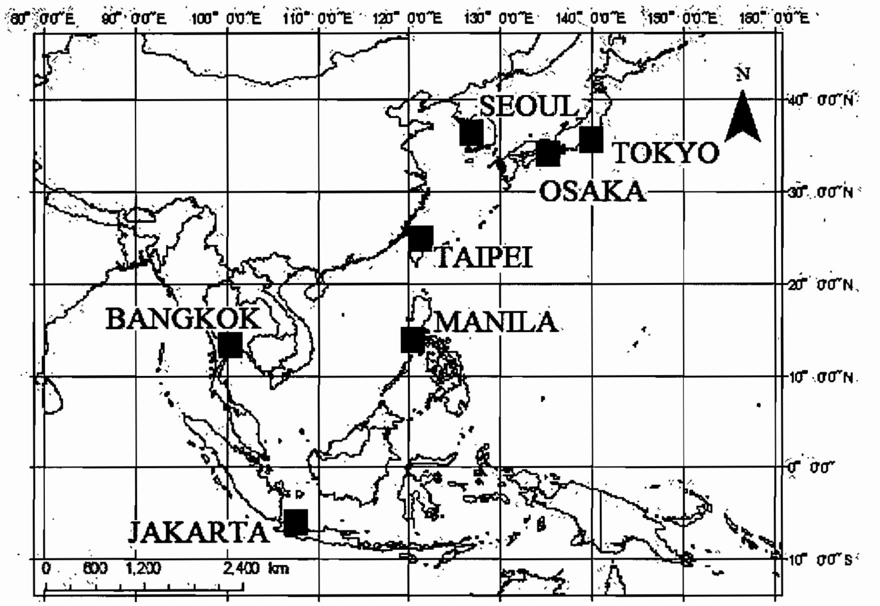


Fig. 1 Location of the study area.

to the relative importance of groundwater and surface water as reliable water resources have been investigated in Osaka and Bangkok in this study. In addition, the effect of global warming on the subsurface environment in Taipei is discussed.

LAND COVER/USE ANALYSES FOR SUBSURFACE ENVIRONMENTS

In order to evaluate the surface environment, which distributes water, heat and materials to the subsurface and others, land-use/cover changes in three periods (1930s, 1970s and 2000s) have been analysed using GIS with 0.5-km grid for the seven targeted cities. The results of expanding urban area (houses) in Tokyo, Osaka, and Seoul are shown in Fig. 2. As can be seen from Fig. 2, the urban area (houses) expanded in Tokyo between 1930 and 1970 by 788 km^2, between 1970 and 2000 by 2991 km^2, in Osaka by 569 km^2 and 907 km^2, and in Seoul by 196 km^2 and 954 km^2, respectively. In general, the decrease of permeability associated with increased urban extent occurs later, and then a subsequent reduction of the groundwater recharge rate. The increase of urban area also causes an increase in the thermal index which is shown by the magnitude of the heat island effect.

The analyses of land-cover/use changes have been made in Tokyo, Osaka, and Seoul with 10 categories including forest, housing, industry, paddy field, other agriculture field, grass/waste land, ocean, water and wet land, and others. In Fig. 3, the magnitude of the change of the area and flow are shown by the size of circle (solid is present and dotted is the past) and the arrow, respectively (Fig. 3). The magnitude of the change in area is greatest for Tokyo from 1970 to 2000. The magnitude of the change in area is also larger in Seoul than that in Osaka, from 1970 to 2000.

The reliability of groundwater as a water resource may be decreased after urbanization due to reduction of the groundwater recharge rate, and increase of groundwater contamination including introduction of subsurface thermal anomalies. The development of integrated indicators, based on GIS for understanding the relationship between human activities and the subsurface environment, is needed in future studies.

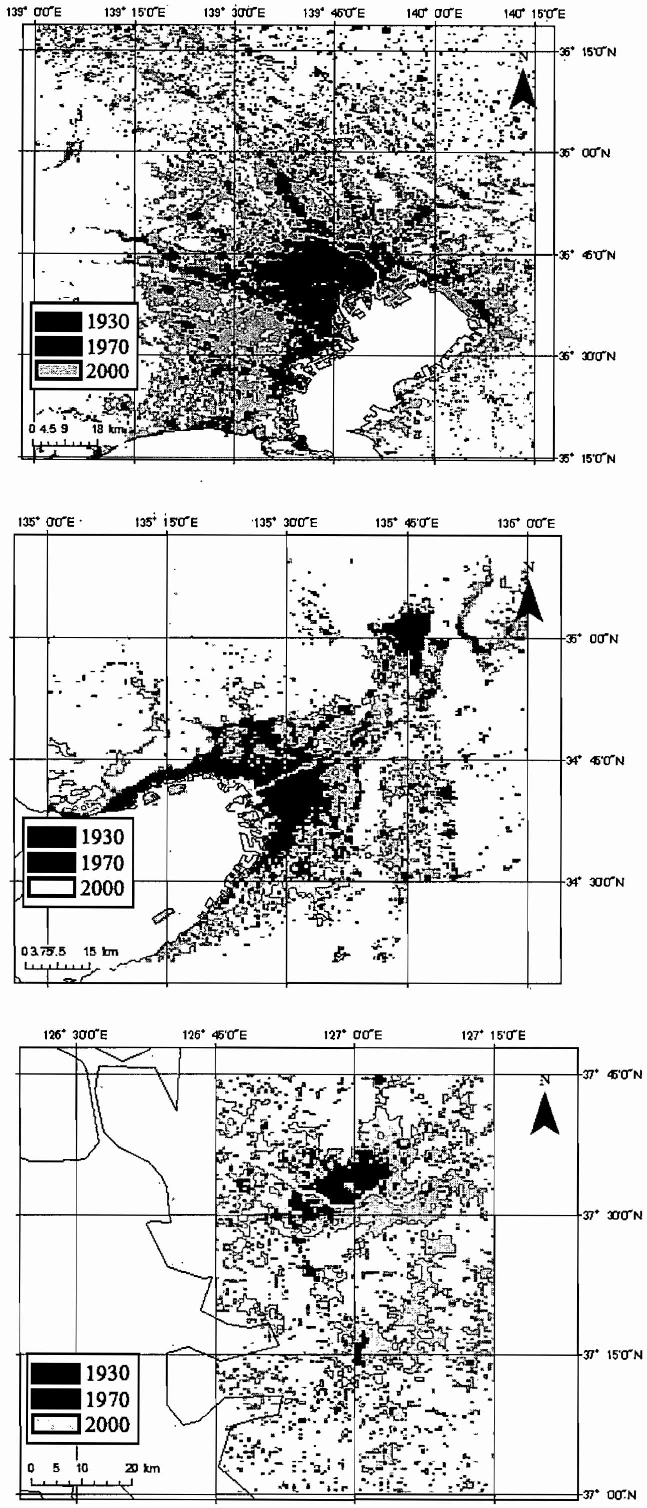


Fig. 2 Changes in urban area in Tokyo (top), Osaka (middle), and Seoul (bottom).

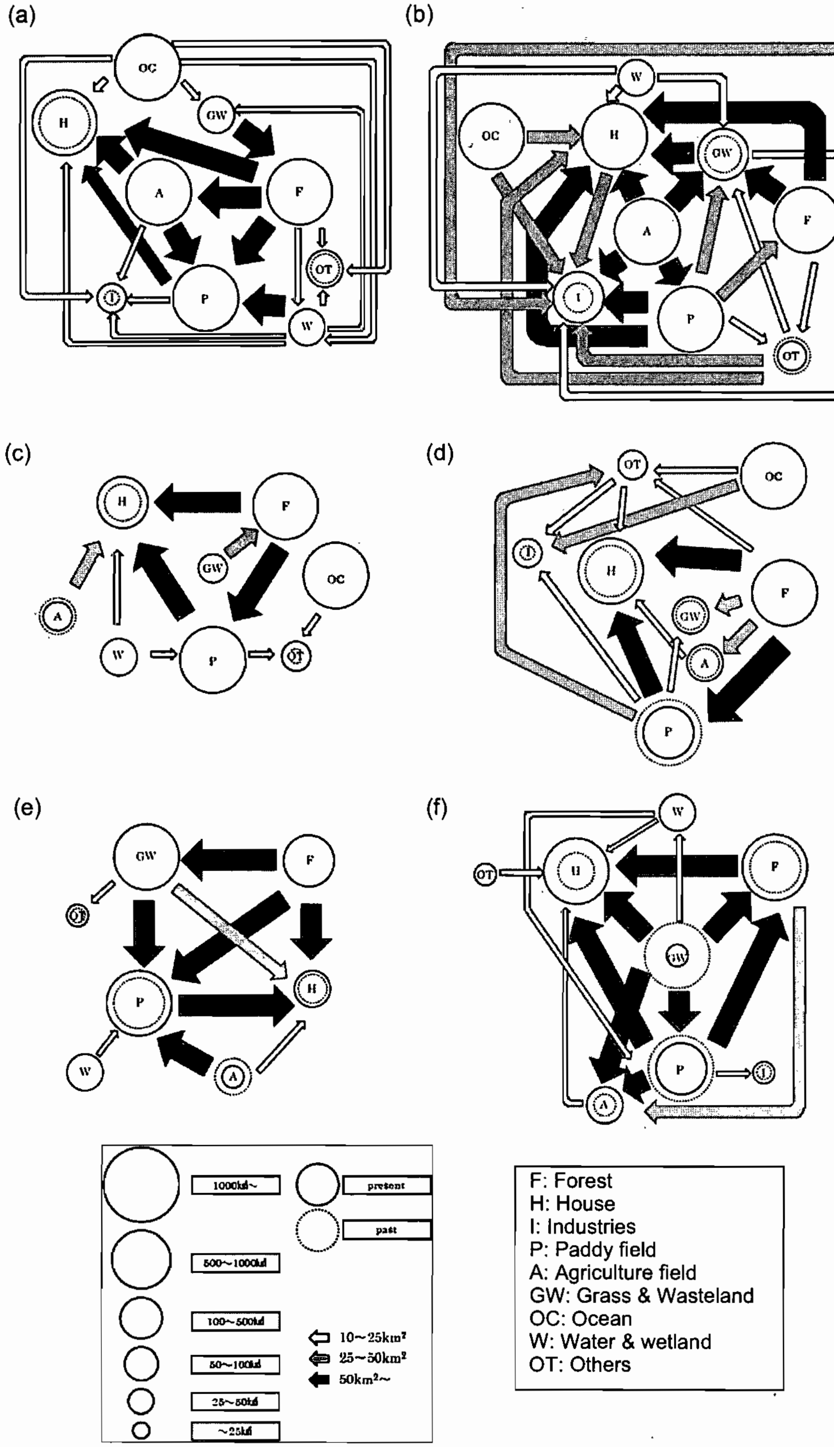


Fig. 3 Changes in land-cover/use in: (a) Tokyo from 1930 to 1970, (b) Tokyo from 1970 to 2000, (c) Osaka 1930–1970, (d) Osaka 1970–2000, (e) Seoul 1930–2000, and (d) Seoul from 1970 to 2000.

CHANGES IN RELIABLE WATER RESOURCES BETWEEN SURFACE WATER AND GROUNDWATER

Changes in reliable water resources from groundwater to surface water through dam construction have occurred, including in Asian cities. This is attributed to the increase of water demand due to an increase in population. At the early stage of settlement in the city, people use more groundwater because it is relatively easy to access without major infrastructure. With increasing water demand due to population growth in the city, the groundwater itself does not meet the demand and then new facilities such as dams are built for water storage on the surface instead of groundwater storage in the subsurface. This is one approach to the change in reliable water resources (Taniguchi *et al.*, 2009a).

However, climate change, such as changes in precipitation patterns due to global warming, have resulted in some Asian cities shifting their water resources in the opposite direction, i.e. from surface water to groundwater. For example, Taiwan is now using more groundwater because of the decrease in reliability of their surface water resources stored behind dams (Wang, 2005, Fig. 4). In Taiwan, the decrease in precipitation days due to global warming, without a change in the total amount of precipitation, has caused a decrease of the reliability of their surface water resources. Therefore in this case, the climate change may be one of the reasons for switching reliable water resources between surface water and groundwater.

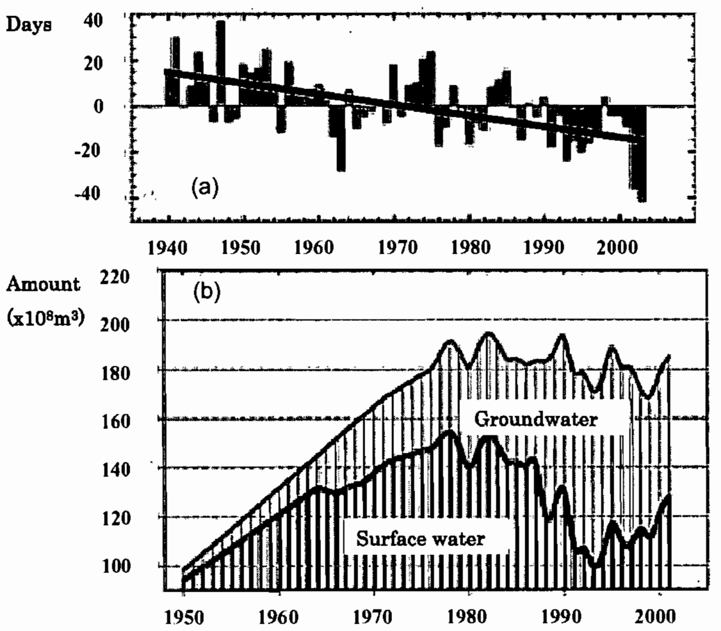
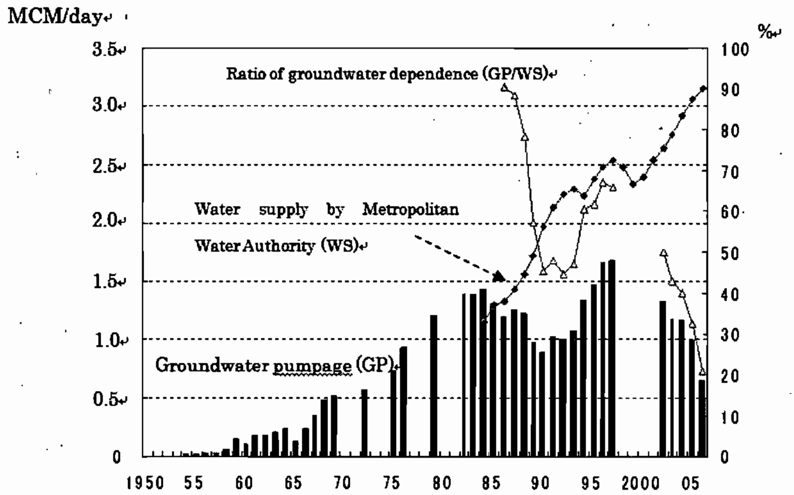


Fig. 4 Change in: (a) days of rainfall, and (b) consumption of water (surface water *vs* groundwater) in Taipei (Wang, 2005).

LAWS AND INSTITUTIONS TO MANAGE GROUNDWATER

The groundwater is "private water" in many countries, including Japan and Thailand. This is mainly attributed to the fact that the groundwater belongs to the private land owner, even though the groundwater moves in the natural flow system. However, the surface water is "public water" which means the local or federal government controls and manages the surface water. The shifts of reliance on water resources between surface water and groundwater causes many problems, both under and above the ground.

(a) **Development of groundwater laws and conversion of water supply in Bangkok**



Source: Department of Groundwater Resources, Thailand

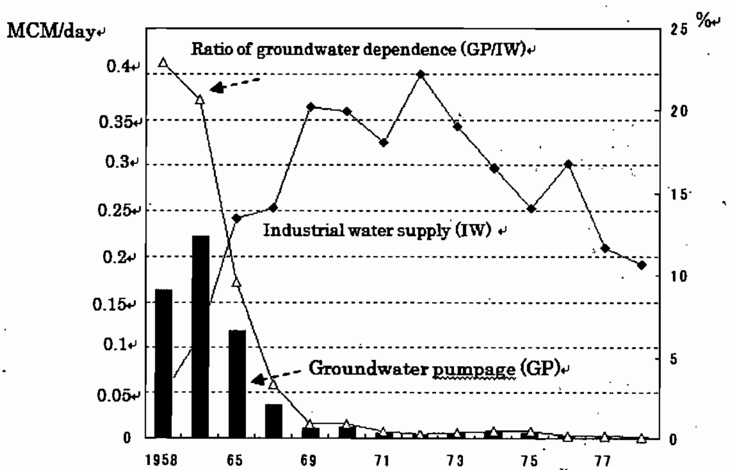
'77 Groundwater Act

'83 Mitigation of Groundwater Crisis and Land Subsidence in the Bangkok Metropolis

'85 Groundwater Charge System→ Rate was increased gradually until 2003.

'04 Groundwater Preservation Charge

(b) **Development of groundwater laws and conversion of water supply in Osaka city**



Source: Research and Statistics Department, Minister's Secretariat, Census of Manufactures, Report by Industrial Land and Water

'56 Industrial Water Law

'59 City Ordinance on Prevention of Land Subsidence

'62 Industrial Water Law Revised

'62 Law on Regulation of Groundwater Pumpage for Building

Fig. 5 Change in groundwater law and managements in Bangkok and Osaka.

The groundwater pumpage (GP), total water supply (WS), and the ratio of groundwater dependence (GP/WS) in Bangkok and Osaka are shown in Fig. 5. Some of the main laws related to water in both areas are also shown in Fig. 5. In the case of Osaka, the ratio of groundwater dependence has decreased since 1958, due to several water laws, including industrial water law and laws on regulation of groundwater pumpage for building.

As can be seen in Fig. 5, mitigation of groundwater and land subsidence in Bangkok began around 1985 due to the beginning of charging for groundwater use. However, use of groundwater started increasing again from 1991, because water demand exceeded water supply. This shows that the regulation of groundwater pumping due to serious land subsidence did not work without alternative water resources, because the groundwater is controlled by the private sector. The price of water is another major factor influencing the relative reliance on groundwater *versus* surface water.

CONCLUSION

Analyses of land-cover/use changes in three Asian cities (Tokyo, Osaka, and Seoul) show that in each, the urban area (houses) expanded much faster between 1970 and 2000, than between 1930 to 1970. Urbanization causes a decrease of groundwater recharge rate and increase of thermal transport into the subsurface environment. Analyses of water law and institutions in Bangkok show that the regulation of groundwater pumping due to serious land subsidence did not work in the absence of alternative water resources, particularly because the private sector owns the groundwater. The price of water is another major influence on changes of reliable water resources between groundwater and surface water. The development of integrated indicators based on GIS for understanding the relationship between human activities and subsurface environment, is needed in future studies.

Acknowledgements We thank the members of the USE (Urban Subsurface Environment) project of RIHN (Research Institute for Humanity and Nature) for helping to conduct this research project. This research is closely connected with other international research programmes, including UNESCO-GRAPHIC (Groundwater Resources under the Pressures of Humanity and Climate Changes; Aureli & Taniguchi, 2006). We are also connected to the international organizations IAHS, IGRAC, IAH, GWSP, IGBP-LOICZ, and IHDP. We thank those organizations for their useful collaborations.

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Caractérisation du ruissellement et de l'érosion de la parcelle au bassin versant en zone sahélienne: cas du petit bassin versant de Tougou au nord du Burkina Faso

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Résumé Cette étude s'inscrit dans le cadre du programme AMMA (Analyses Multidisciplinaires de la Mousson Africaine) et vise à mieux caractériser le ruissellement et l'érosion hydrique des sols depuis l'échelle de la parcelle jusqu'à celle du petit bassin versant dans un contexte de changements environnementaux. Pour atteindre les objectifs de l'étude, le bassin versant de Tougou, d'une superficie de 37 km^2 a été retenu et équipé. Il est situé dans la partie supérieure du bassin du Nakambé, en zone sahélienne du Burkina Faso. Les résultats obtenus montrent une réponse fortement contrastée à l'impulsion pluvieuse en fonction des différents états de surface du bassin. Comparées aux zones de cultures, les zones dégradées à croûtes superficielles semblent les plus aptes au ruissellement (avec des taux de ruissellement avoisinant les 100%) et présentent les taux d'érosion les plus élevés (jusqu'à 103 t/ha en 2006). Ces zones nues semblent donc être les plus sensibles à l'érosion hydrique. A l'exutoire du bassin versant, les coefficients de ruissellement et les taux d'érosion augmentent avec la pluviométrie passant respectivement de 11% et 0.5 t/ha en 2004 (pluie = 392 mm) à 23% et 7 t/ha en 2006 (pluie = 726 mm). A partir de ces résultats et en considérant l'occupation des sols, nous avons agrégé l'érosion à la parcelle pour obtenir l'érosion brute sur l'ensemble du bassin. Il ressort que les exportations de sédiments mesurées à l'exutoire du bassin représentent respectivement 4%, 15% et 27% de l'érosion brute en 2004, 2005 et 2006. Ces chiffres témoignent des faibles taux d'exportations, et des dépôts importants de sédiments, surtout les matériaux grossiers, à l'intérieur du bassin. Ces dépôts se font essentiellement dans les axes d'écoulement.

Mots clefs changement climatique; érosion des sols; ruissellement; états de surface; Nakambé; AMMA; Burkina Faso

Characterisation of runoff and soil water erosion at the plot and catchment scale in the Sahel: Tougou catchment case study, northern Burkino Faso

Abstract This study comes within the framework of the AMMA (Multidisciplinary Analyses of the African Monsoon) programme and aims to better characterize runoff and soil water erosion from the plot scale to the catchment scale in a context of environmental changes. To reach these objectives, the Tougou catchment, 37 km^2 in area, was selected and equipped. It is situated in the upper part of the Nakambe River basin in the Sahelian zone of Burkina Faso. The results obtained at plot scale show a strong contrast in the response to rainfall events, depending on the different soil surface characteristics of the catchment. Compared to cultivated zones, degraded zones with crusted surfaces appear to have the highest aptitude to runoff (with runoff rates reaching 100%) and present the highest erosion rates (up to 103 t/ha in 2006). These crusted and bare soils thus seem to be the most sensitive to water erosion. At the basin outlet, runoff coefficients and erosion rates increase with rainfall, changing, respectively, from 11% and 0.5 t/ha in 2004 (rainfall = 392 mm) to 23% and 7 t/ha in 2006 (rainfall = 726 mm). From these results, and considering land uses, we aggregated plot erosion to obtain raw erosion over the entire catchment. It emerges that sediment export measured at the catchment outlet represent 4%, 15% and 27% of the raw erosion estimated, respectively, in 2004, 2005 and 2006. This demonstrates the low rates of sediment export, and the importance of sediment deposition, especially of coarse materials, inside the catchment. These deposits occur mainly in the channels.

Key words climate change; soil erosion; runoff; soil surface characteristics; Nakambe; AMMA; Burkina Faso

INTRODUCTION

Dans un contexte de variabilité et changement climatique, la question des ressources en terres pour les activités humaines (agriculture, pastoralisme, sylviculture, ...) reste plus que jamais d'actualité dans des zones arides tel que le Sahel. En effet, cette région est soumise à une dégradation avancée du milieu du fait de la faible couverture végétale et des fortes intensités de pluies. La dégradation de l'environnement dans cette région se manifeste sous forme d'érosion et de réduction de la fertilité des sols. Plusieurs études sur la quantification des terres érodées ont déjà été menées dans la région (Diallo, 2000; Thioubou, 2001; Yacouba *et al.*, 2002). La plupart des ces études se sont intéressées à la quantification de l'érosion au niveau parcellaire. Le passage de la parcelle au bassin versant reste toujours un problème. D'autre part, les études au niveau des petits bassins versants (Karambiri, 2003) ne permettent pas de comprendre la réponse érosive élémentaire. En effet les aspérités de surface peuvent piéger les terres érodées, ce qui se traduit dans ces zones de piégeage par un enrichissement en éléments fins et en éléments chimiques réduisant globalement les pertes en terre et de fertilité mesurées sur les parcelles. La mesure à l'exutoire du bassin versant donne une estimation globale de l'érosion, mais ne tient pas compte non plus, des pertes locales qui peuvent fortement influencer la productivité des terres. C'est pourquoi une étude visant à faire le lien entre l'érosion au niveau des parcelles et la quantification du transport à l'exutoire s'avère nécessaire. Elle aura pour avantage de mieux comprendre, localement, la dégradation des terres et ses conséquences à court terme sur la productivité des terres agricoles et, à long terme, ses effets néfastes sur l'environnement (ensablement et envasement des cours d'eau et retenues, eutrophisation). La présente étude vise à caractériser l'érosion à différentes échelles et à faire le lien entre les mesures au niveau des parcelles et les quantités des terres mesurées au niveau de l'exutoire d'un bassin versant.

MATÉRIELS ET MÉTHODES

Site d'étude

Le bassin versant de Tougou est situé dans la partie amont du bassin du Nakambé, en zone sahélienne du Burkina Faso, à 25 km à l'Est de Ouahigouya sur l'axe Ouahigouya-Titao (Fig. 1). D'une superficie de 37 km^2, ce bassin est fortement cultivé (à environ 70%) et se caractérise par un relief très peu accidenté avec quelques monticules et collines par endroit, d'altitude faible. La morphologie présente une succession de collines (haut glacis) raccordées aux axes d'écoulement (bas-fonds) par de longs glacis. La végétation est surtout dominée par *Acacia albida* et le karité. Les cultures les plus courantes sont le mil, le sorgho, le maïs, l'arachide et le niébé. Sur le plan pédologique, on rencontre des sols sableux et sablo-limoneux au niveau du bas-fond, des sols sablo-limoneux sur le glacis et un recouvrement gravillonnaire sur les têtes du bassin et les collines.

L'occupation du sol a été déterminée par traitement d'images satellite (image Landsat-7 ETM+ du 22/11/2002). En 2002, le bassin était occupé à 79% par des cultures et à 21% par des sols nus.

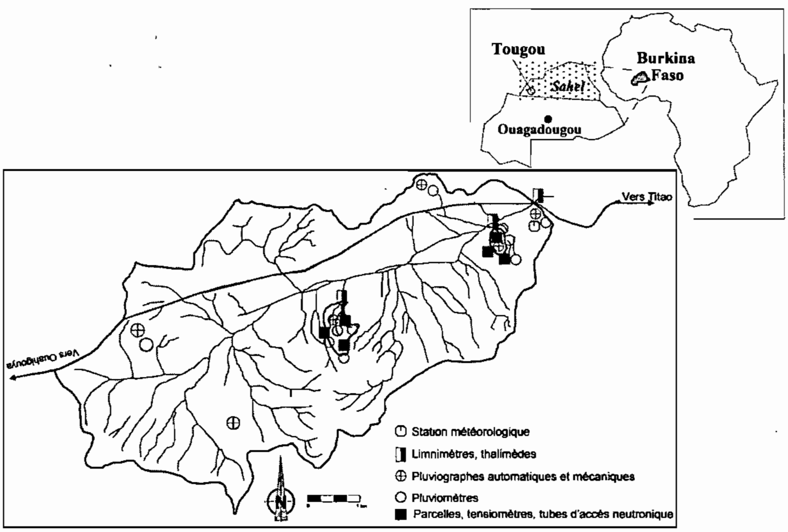

Fig. 1 Localisation et équipements du bassin versant de Tougou.

Le climat est de type sahélien, caractérisé par une seule saison des pluies qui va de juin à septembre. La pluviométrie moyenne annuelle à Ouahigouya est de 620 mm sur la période de 1961 à 2004. Les températures moyennes mensuelles sur la même période varient de 16° en janvier à 27° en avril. L'ETP moyenne annuelle pour la période 1971–2000 est de 2088 mm.

Equipements et mesures

Le bassin versant de Tougou est équipé d'un thalimède et d'échelles limnimétriques pour les mesures hydrométriques. Il dispose pour les mesures de pluies d'un réseau de trois pluviographes automatiques, de trois pluviographes mécaniques, de dix pluviomètres standards et de deux pluviomètres au sol de type Snowdown. Le bassin comporte six parcelles d'érosion situées sur différents types d'états de surface (3 parcelles en zone de cultures et 3 parcelles en zones dégradées sans culture). Les parcelles sont métalliques et constituées d'une surface réceptrice des eaux de pluie de 1 m^2, munie d'un collecteur et reliée à un fût plastique enterré (200 L) à l'aide d'un tuyau PVC (ϕ 64 mm).

Les mesures se font en saison des pluies depuis l'échelle de la parcelle de 1 m^2 à celle du petit bassin versant de 37 km^2 et concernent la pluie, les écoulements aux exutoires, le ruissellement à la parcelle et les prélèvements d'échantillons d'eau. Elles se sont étalées sur trois années (2004–2006).

Pour la quantification de l'érosion, les prélèvements d'échantillons d'eau se font manuellement à l'aide de flacons plastiques d'un litre, en fractionné à des pas de temps variables (10 à 30 minutes en fonction de la forme des crues) à l'exutoire du bassin, et en cumulé à la fin de la pluie à la parcelle.

Après avoir bien homogénéisé les échantillons d'eau, on retient 500 mL qui sont évaporés à l'étuve à 105°C pendant 24 h. Les teneurs en matières en suspension (MES) sont déterminées par pesée de la masse sèche avec une balance de précision. Le charriage de fond (CDF) est collecté au niveau des collecteurs des parcelles après chaque crue et séché au soleil.

RÉSULTATS ET DISCUSSIONS

Bilans des pertes en terre

Au niveau des parcelles d'érosion Les pertes en terre au niveau de la parcelle dépendent des types d'états de surface et des conditions pluviométriques annuelles (Tableau 1). Sur les parcelles en zone de cultures, elles restent faibles et s'élèvent en moyenne à 6.5 t/ha en 2004, 19.5 t/ha en 2005 et 14.1 t/ha en 2006. Tandis que pour les parcelles en zones dégradées, elles sont en moyenne de 42.8 t/ha en 2004, 60.5 t/ha en 2005 et 74.8 t/ha en 2006. Malgré ces variations annuelles, les différences entre les pertes en terre sur ces deux zones restent significatives (Fig. 2), les valeurs individuelles d'érosion sur les parcelles en zone de cultures restant inférieures à 25 t/ha alors que celles sur les parcelles en zones dégradées sont supérieures à 30 t/ha sur les trois années.

Ces résultats montrent une plus grande sensibilité à l'érosion hydrique des sols dégradés à croûtes superficielles comparativement aux sols en zone de cultures, et corroborent ceux trouvés

Tableau 1 Résultats des mesures à la parcelle en 2004, 2005 et 2006.

	Parcelle	2004 P (mm)	Lr (mm)	CTS (t/ha)	2005 P (mm)	Lr (mm)	CTS (t/ha)	2006 P (mm)	Lr (mm)	CTS (t/ha)
Sols cultivés	PZC1	392	63	5	449	204	17	726	319	14
	PZC2		9	9		93	18		169	10
	PZC3		22	6		95	24		200	18
Sols dégradés	PZN1		252	42		356	44		576	64
	PZN2		327	57		472	62		661	57
	PZN3		44	30		130	76		445	103

P: pluie; Lr: lame d'eau ruisselée; CTS: charge totale solide.

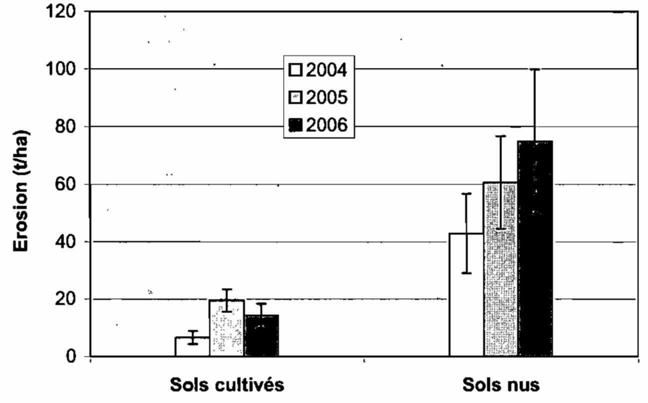

Fig. 2 Taux d'érosion annuelle à la parcelle en 2004, 2005 et 2006.

par Karambiri (2003) en zone sahélienne du Burkina où les parcelles à croûtes d'érosion donnent des taux d'érosion élevés. Les valeurs trouvées lors de cette précédente étude (19 t/ha au maximum en 2000) sur les parcelles de 1 m^2 (bassin versant de Katchari) restent faibles par rapport à ceux de Tougou qui atteignent 103 t/ha en 2006. Cette forte sensibilité à l'érosion des sols dégradés s'explique essentiellement par leur nature et leur perméabilité. En effet, ces sols présentent des taux d'infiltration faibles les rendant plus sensibles au détachement des particules par effet "splash". Les forts taux de ruissellement (Tableau 1) assurent le transport de ces particules détachées. Ces sols dégradés et encroûtés contiennent également une grande proportion d'argiles, qui subit une dispersion colloïdale ou argileuse lors de l'humectation du sol (Casenave & Valentin, 1989; Valentin & Bresson, 1992; Le Bissonnais *et al.*, 1996). En outre, l'horizon de surface prend un aspect "boueux" du fait de l'infiltration dans les premiers centimètres du sol, il devient par la suite fluide et facilement transportable par l'eau (Karambiri, 2003). Ce phénomène est limité au niveau des sols en zone de cultures qui présentent une bonne infiltrabilité et sont de nature plus sableuse. La mise en cultures assure également une protection du sol contre l'effet des gouttes de pluie et réduit considérablement l'érosion. Cela se traduit par les écart-types de variation des pertes en terres qui restent faibles au niveau de ces parcelles comparativement à ceux concernant les parcelles en zone dégradée.

Les sédiments exportés sont essentiellement sous forme de charriage de fond constitué de matériaux grossiers piégés au niveau du collecteur des parcelles. Les matières en suspension (MES) représentent en moyenne 3%, 20% et 11% (en zone cultivée) et 5%, 19% et 14% (en zone dégradée) de la charge totale solide (CTS) pour les années 2004, 2005 et 2006 respectivement.

Au niveau du bassin versant Les résultats d'érosion mesurés à l'exutoire du bassin de Tougou sont regroupés dans le Tableau 2. Pour les trois années de suivi, les exportations solides varient de 0.5 à 7 t/ha. Ces résultats varient linéairement avec de la pluviométrie moyenne sur le bassin avec une pente de 2%.

Ces pertes en terres mesurées à l'exutoire du bassin versant de Tougou méritent d'être comparées à celles obtenues par Karambiri (2003) sur un petit bassin versant (1.4 ha) dans la

Tableau 2 Résultats des mesures à l'exutoire du bassin versant en 2004, 2005 et 2006.

Année	P (mm)	Lr (mm)	CTS (t/ha)
2004	392	44	0.5
2005	449	91	4
2006	726	164	7

P: pluie; Lr: lame d'eau ruisselée; CTS: charge totale solide.

région de Dori (Pluviométrie moyenne annuelle de 512 mm) qui sont de 6.8 t/ha en 1998, 4.0 t/ha en 1999 et 8.4 t/ha en 2000. Les résultats sur les deux bassins restent du même ordre de grandeur et confortent la forte interdépendance de l'érosion et de la pluviométrie dans ce milieu. Nous avons également mis nos résultats dans un contexte sahélien plus général, et la Fig. 3 permet de visualiser la répartition de l'érosion en fonction de la taille des bassins étudiés. On note que les données sur Tougou respectent l'allure générale de la décroissance des quantités de sédiments avec l'augmentation de la taille des bassins étudiés, mais restent légèrement supérieures aux autres valeurs de la littérature, certainement à cause des nouvelles conditions environnementales marquées par une plus forte influence des activités anthropiques sur la dégradation des sols.

Sur les trois années d'observation, les quantités annuelles de sédiments exportées dépendent beaucoup plus de l'importance des événements averses-crues que de leur répartition dans la saison. En effet, on remarque qu'environ 80% des pertes en terre annuelles sont causées par moins de 40% des événements. Ce trait est caractéristique du milieu sahélien comme également démontré par Karambiri (2003).

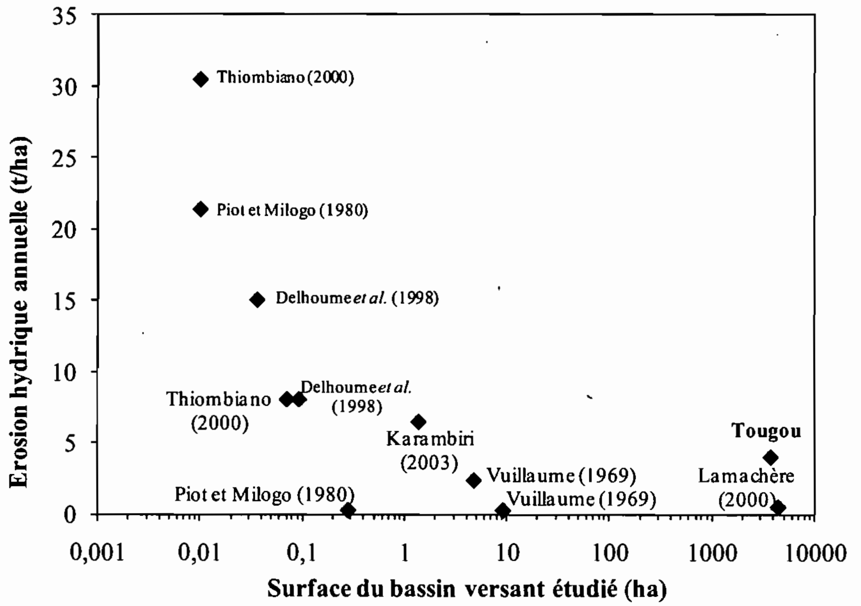

Fig. 4 Evolution des différentes données d'érosion en fonction de la surface du bassin versant en zone sahélienne. Les données de Tougou sont les valeurs moyennes sur les périodes d'observation (2004, 2005 et 2006).

De la parcelle au bassin versant

Il est connu que les mesures d'érosion à la parcelle permettent de quantifier la production brute élémentaire, alors que celles au niveau des bassins versants permettent d'estimer les pertes en terre ou exportation solides (Stroosnijder, 2005) intégrant d'autres formes de processus d'érosion (dépôts, érosion en ravine, etc.). Nous avons comparé les moyennes des taux d'érosion des trois parcelles avec les taux d'érosion obtenus sur le bassin. Les taux d'érosion à la parcelle ont été extrapolés à l'ensemble du bassin en considérant les proportions d'occupation du bassin en sols nus et cultivés qui sont respectivement de 21% et 79%. Les résultats sont présentés dans le Tableau 3.

On note que les taux d'érosion mesurés à l'exutoire du bassin restent faibles par rapport à ceux obtenus par extrapolation des taux élémentaires. Ces résultats démontrent une forte dynamique d'érosion et de dépôt à l'intérieur du bassin. En effet, les mesures faites à la parcelle indiquent des taux d'érosion élevés, surtout pour les sols nus (supérieurs à 30 t/ha); alors que les

Tableau 3 Extrapolation des résultats d'érosion de la parcelle au bassin versant.

Année	Taux d'érosion extrapolé (t/ha)	Taux d'érosion mesuré (t/ha)
2004	14	0.5
2005	28	4
2006	27	7

mesures au niveau de l'exutoire du bassin indiquent des taux faibles d'exportations solides (<10 t/ha). Sur les trois années 2004, 2005 et 2006, les pourcentages d'exportation sont respectivement de 4%, 15% et 27% (Tableau 3). L'érosion intense au niveau des parcelles est suivie de dépôts importants à l'intérieur du bassin surtout dans les axes d'écoulement. Cela s'explique par la granulométrie des sédiments érodés et les faibles pentes d'écoulement. En effet, les matériaux arrachés au niveau des surfaces élémentaires sont essentiellement composés de particules grossières avec de faibles quantités de matières en suspension. Ces résultats ne doivent toutefois pas cacher la forte variabilité des réponses élémentaires et la non linéarité des processus d'érosion et de ruissellement. Dans ces conditions, l'extrapolation des résultats d'érosion de la petite à la grande échelle reste délicate et dépend du milieu considéré (Boix-Fayos *et al.*, 2007).

CONCLUSION

A l'issue de cette étude, il ressort une forte dynamique spatiale des processus d'érosion à l'échelle des petits bassins versants sahéliens, en ce sens qu'on observe des forts taux d'érosion à la parcelle (arrachement et transport de particules) et une redistribution importante (dépôt) des sédiments à l'intérieur du bassin. Les fortes pertes en terres au niveau des versants, démontrées par les mesures de taux d'érosion à la parcelle, ont pour conséquence une perte de fertilité des champs conduisant à des baisses de rendements culturaux. Les dépôts dans les axes d'écoulements et les dépressions superficielles contribuent au comblement des cours d'eau peuvent entrainer des difficultés d'évacuation des eaux et donc des débordements et des inondations. Ces résultats doivent être complétés par une analyse de la qualité chimique des eaux de ruissellement afin d'estimer la perte d'éléments fertilisants, indispensables aux plantes et cultures.

Remerciements Les auteurs remercient tous les observateurs de terrain et les stagiaires qui ont contribué à la collecte des données. Cette étude a bénéficié d'un soutien financier de l'Union Européenne dans le cadre du programme AMMA (Analyses Multidisciplinaires de la Mousson Africaine), de la Coopération suisse (DDC) et de l'IRD (Institut de Recherche pour le Développement).

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Retreating snowpacks under climate change: implications for water resources management in the Austrian Alps

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Abstract This paper presents three different hydrological regimes (two mountainous and one lowland) within the Danube River basin. Their importance for Integrated Water Resources Management (IWRM) within the river basin is discussed. One of these hydrological regimes – a non-glacierised mid mountain region in the Austrian Alps – is selected as a case study in order to assess the impact of climate change on both IWRM in the alpine region and the lowlands of the Danube River basin. The disproportionate hydrological influence of the Alps is demonstrated for the Danube River basin, especially during summer, late spring and early autumn. The climate normal period from 1961 to 1990 is chosen as the baseline. As future climate change scenarios, 2°C and 4°C warming with seasonal precipitation changes – as predicted for the middle and the end, respectively, of the 21st century – are chosen. The climate change scenarios show a substantial decrease in snow cover duration and snow accumulation within the case study area. The impact of this loss in the natural storage reservoir on the IWRM in both the alpine region (case study) and the lowlands of the Danube River basin is discussed. Generally, a shift in flow seasonality is observed. Total and base flow increase in winter, and decrease from spring to autumn. Regional water availability within the alpine case study, however, is sufficient to provide all demand holders thoughout the whole year. However, this shift in alpine water hydrology has a strong effect on water availability in the downstream Danube River. The Alps will not provide the same amount of water to the lowlands during the critical seasons of late spring, summer and autumn.

Key words climate change; snowpack; mountain hydrology; IWRM; Alps; Austria

INTRODUCTION

Integrated water resources management (IWRM) requires the balancing of available water resources against the water demands of all different stakeholders at the catchment level. Typically both water demands and available resources fluctuate over time. This temporal variation is particularly strong in mountainous areas affected by seasonal snow cover (Vanham *et al.*, 2008a). Water availability is low in winter due to the temporal storage of water in the form of snow and ice. River flows are generally at their lowest during this season and at their highest in spring during snowmelt. However, in winter water demand is highest due to winter tourism. The understanding of current hydrological regimes and the impact of climate change on the latter is therefore essential within IWRM. This paper presents different hydrological regimes within the Danube River basin. Their importance for IWRM within the river basin is discussed. In order to assess the impact of climate change on these regimes, one non-glacierised case study area within the mountainous part of the Danube River basin (Fig. 1) in the Austrian Alps is selected. The hydrological response to global warming for this case study area is assessed, as well as its implications for IWRM. The European Alps are a major water tower for both the mountainous areas themselves, as well as for the surrounding lowlands. The hydrological importance of the mountainous parts within river basins like the Danube for the lower lying areas has been described by Viviroli *et al.* (2007).

METHODOLOGIES

The hydrological model PREVAH (Gurtz *et al.*, 1999) is applied for the case study area. The WMO climate normal period from 1961 to 1990 was chosen as the reference period. The first climate change scenario implies an average temperature rise of 2°C without precipitation change, a possible scenario for 2050. The second climate change scenario is the Climate High Resolution Model (CHRM) developed by the ETH Zürich under the A2 scenario (Christensen & Christensen, 2007). This scenario is valid for the end of the 21st century (2071–2100). The case study area

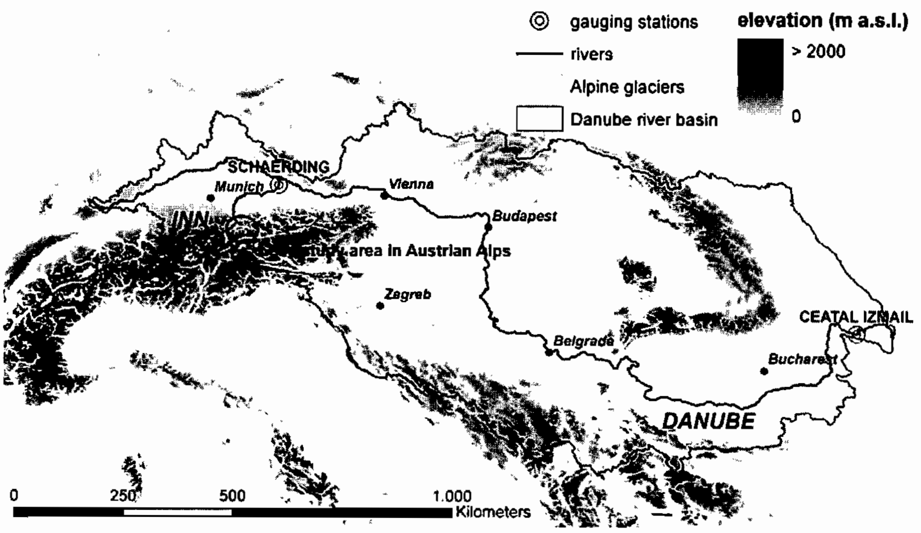

Fig. 1 Location of the case study area in the Danube River basin.

consists of four subcatchments (Fig. 3 and Fig. 5) for which the hydrological water balance (equation (1)) was calculated by means of the model, and calibrated with measured daily flow time series of the reference period 1961–1990 at the respective gauging stations (BRIX, WEI, KITZ, FIE in Fig. 5). The same modules and parameters were used as in the prior analysis of Vanham *et al.* (2009a). The components of equation (1) are precipitation (P), total runoff (Q), evapotranspiration (ET) and change in water storage (ΔS).

$$P = Q + ET + \Delta S \qquad (1)$$

RESULTS AND DISCUSSION

Hydrological importance of the Alps in the Danube River basin

Weingartner *et al.* (2007) have already stated that with a mean contribution of 26% of the total discharge, the mountain region of the Danube system (10% of total area) supplies 2.6 times more water than might be expected on the basis of surface area alone. The mountainous part of the Danube thus plays a distinctive role in the hydrology of the whole river basin. The disproportional influence of the largely alpine Inn catchment (up to gauge "Schärding") to the total Danube basin (up to gauge "Ceatal Izmail") shows large variations on a monthly basis (Fig. 2). The latter gauge is located at the Danube's entrance into its extensive delta, which flows into the Black Sea. Figure 1 shows the location of both gauges. It has to be stressed that water for certain stakeholders like irrigation (Fig. 10) can be "lost" from the Danube River before it reaches gauge Ceatal Izmail. The areal proportion of the Inn River catchment is only 3.2%, but its mean contribution to total discharge ranges from a minimum of 6.6 in the winter months to a maximum of 18.5 in August. Expressed in disproportional influence, these values range from 2.1 to 5.8.

To understand these results the average monthly water balances (period 1961–1990) of three catchments (Fig. 3) within the Danube River basin are compared. The first catchment is the strongly glacierised upper Ötztal up to the gauge "Huben", with an average altitude of 2625 m a.s.l. (min. 1185 and max. 3770 m a.s.l.). The second is the non-glacierised Brixentaler Ache catchment up to the gauge "Bruckhäusl", with an average altitude of 1326 m a.s.l. (min. 519 and max. 2427 m a.s.l.). This catchment is one of the four catchments of the case study area. The third catchment is the Kamp catchment up to the gauge "Stiefern", with an average altitude of 613 m a.s.l. (min. 263 and max. 1015 m a.s.l.).

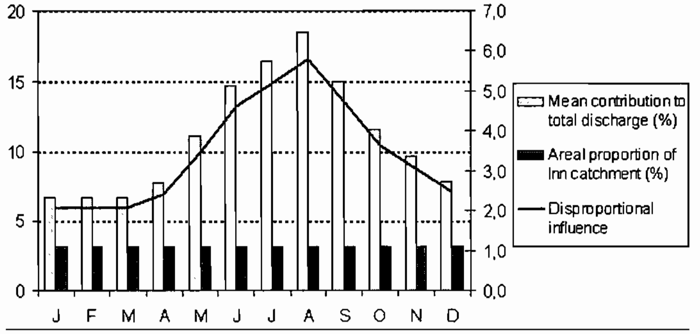

Fig. 2 Monthly contribution of the Inn catchment (gauge "Schärding") to total Danube discharge (gauge Ceatal Izmail) (1961–1990) (primary y-axis); and disproportional influence (mean contribution related to surface area) of the Inn catchment (secondary y-axis). Data source: GRDC, 56058 Koblenz, Germany.

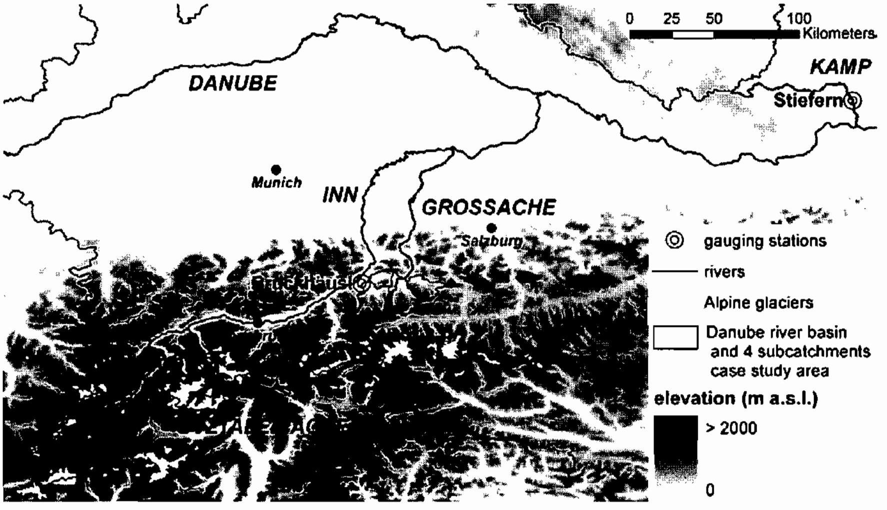

Fig. 3 The four different subcatchments of the case study area, of which two drain to the Inn and two to the Grossache. Location of the gauges Huben on the Ötztaler Ache, Bruckhäusl on the Brixentaler Ache and Stiefern on the River Kamp

The water balance components of the three catchments show clear differences both in average yearly and monthly amounts. Precipitation is much higher and evapotranspiration much lower in the two mountainous catchments due to the effect of temperature decrease with altitude. This results in a more effective runoff generation in the mountainous catchments as compared to in the lowlands. The discharge difference between the two mountainous catchments and the lowland catchment is particularly large in spring and summer, when snowmelt and glacier melt occurs. For the high altitude and glacierised catchment of the Upper Ötztal, maximum snowmelt occurs some weeks later in the year compared with the mid-mountain catchment of the Brixentaler Ache. Maximum glacier melt occurs during summer (July–August). For the Brixentaler Ache maximum snowmelt occurs during spring (April– June). In winter a large proportion of precipitation is stored as snow (dS in Fig. 4 is positive), thereby reducing the runoff from mountainous regions substantially. The typical low flow period is winter. However, as the late spring and summer runoff from the Alps arrive at a time of low flows in the lowlands, evapotranspiration can exceed precipitation. As generally the largest water demands (municipalities, trade and industry,

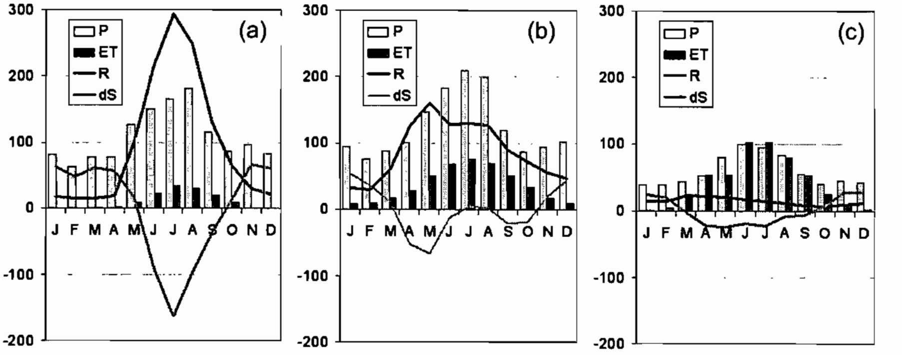

Fig. 4 Monthly average water balance components (mm) for the period 1961–1990 for: (a) the catchment of the Ötztaler Ache up to gauge Huben, (b) the catchment of the Brixentaler Ache up to gauge Bruckhäusl, (c) the catchment of the Kamp up to gauge Stiefern. Data source, (a) and (c): Hydrological Atlas of Austria.

agriculture, cooling water, etc.) are concentrated in the lowlands, the Alps can be defined as a very important seasonal water tower within the Danube River basin.

Case study in the Austrian Alps

The case study area is a topographical, administrative, historical and tourism homogenous area which encompasses four different catchments. A detailed description can be found in Vanham *et al.* (2008a). (Winter) tourism is by far the most important industry, and two of the largest alpine ski regions (Skiwelt Kitzbühel and Skiwelt Wilder Kaiser Brixental) are located within its borders. The annual average duration of snow cover for the reference period extends from 2 to 3 months in the valleys to longer than half a year on the mountain tops (Fig. 5).

The relationship between the elevation within the case study area and the annual water balance components – a result from the hydrological model – shows some important changes for the future scenarios with respect to the reference period (Fig. 6). General tendencies do not change:

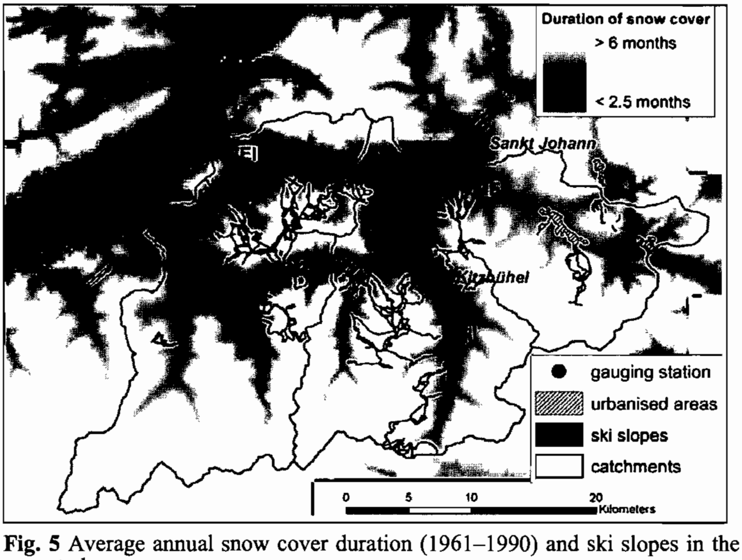

Fig. 5 Average annual snow cover duration (1961–1990) and ski slopes in the four catchments of the case study area.

precipitation in the form of rain and snow (snow water equivalent) increases with altitude, evapotranspiration decreases with altitude, and specific runoff decreases with altitude. However, both snow storage and melt amounts decrease (Fig. 6) as well as snow cover duration (Fig. 7). The effect is stronger for the CHRM scenario, and for both scenarios is elevation dependent. This means that the largest reductions in snow cover duration and snow amount (snow water equivalent) are at the lower elevations within the case study area. The effect decreases with increasing altitude. Up to an elevation of almost 2500 m a reduction is observed. There is a possibility that at very high altitudes snow cover duration and snow amounts could actually increase in future. For the CHRM scenario, an increase in winter precipitation is predicted. However, the areal proportion of these elevations is minor compared to the alpine mountain range within the Danube River basin. Therefore a general decrease over the alpine Danube area is predicted. Within the case study, a reduction in precipitation occurs in the CHRM scenario, and an increase in evapotranspiration due to higher temperatures in both scenarios. In both scenarios a decrease in total average runoff occurs, whereas this decrease is small within the first scenario.

The most important effect of the climate change scenarios for IWRM is on the component runoff (and base flow). Figure 8 shows that a shift in flow seasonality occurs. Both total and base flow shift from the low flow period in winter to the low flow period in summer (and to a minor extent in autumn. As less water is stored as snow in winter, the spring peak flows decrease and the natural reservoir effect of the snow cover decreases.

Implications for water resources management

Case study in the Austrian Alps. Approximately 60 000 people live within the 19 municipalities of the case study area. During the last decade, an average number of annual overnight

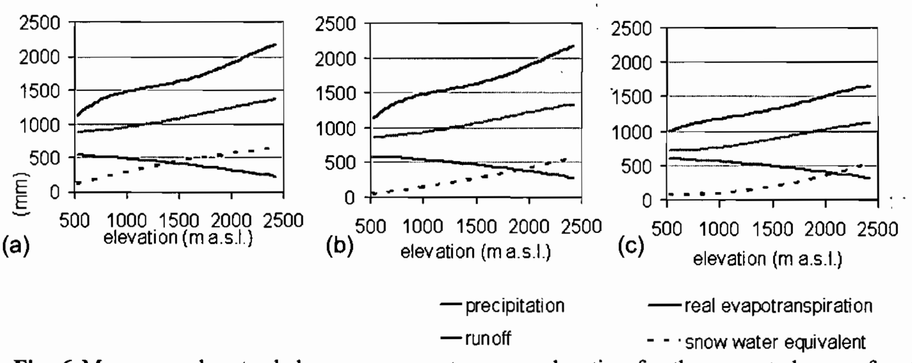

Fig. 6 Mean annual water balance components *versus* elevation for the case study area for: (a) the reference period 1961–1990, (b) an average 2°C temperature increase, and (c) the CHRM scenario.

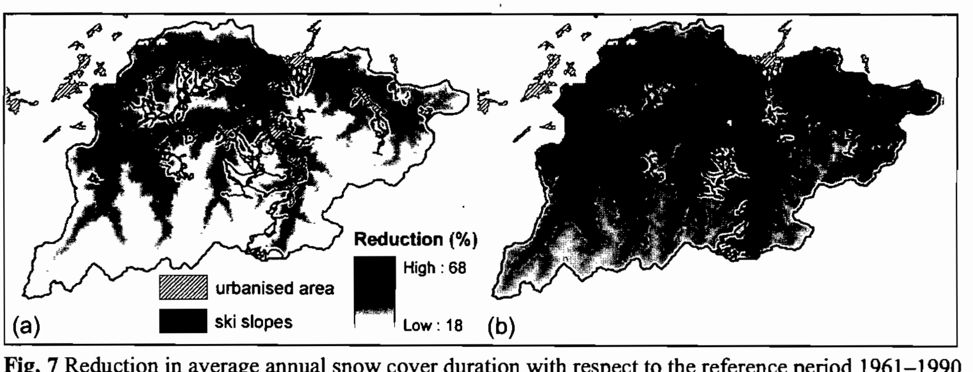

Fig. 7 Reduction in average annual snow cover duration with respect to the reference period 1961–1990 for: (a) an average 2°C temperature increase, and (b) the CHRM scenario.

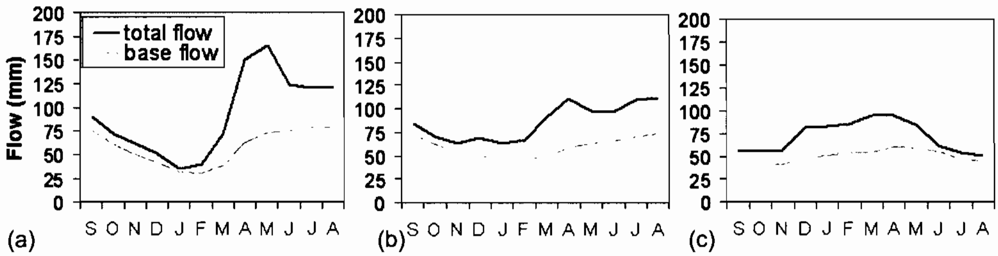

Fig. 8 Mean monthly total and base flow in the case study area for: (a) the reference period 1961–1990, (b) an average 2°C temperature increase, and (c) the CHRM scenario.

stays of 6.5 million has been recorded, of which 50–60% occur during the winter months December to March. The water supply system is not very structured. Generally every municipality has its own supply system. About 10% of the population have their own decentralised supply systems. Domestic water is primarily fed by springs and secondarily by groundwater. For snowmaking surface water is used (Vanham *et al.*, 2008b). In Austria, surface water is not used for drinking water purposes. There is no water storage for hydropower generation. A few small hydropower plants are located directly on the rivers. Typical for alpine regions, the water demand is concentrated in the valleys (except for snowmaking). Figure 9 displays the rasterised water demand (L/s) in the eastern part of the case study area, calculated according to a methodology described in Vanham *et al.* (2009b). In the current situation (reference period) no regional water deficits occur for any stakeholder (Vanham *et al.*, 2008b). Water availability is high compared to water demands. Reservoirs are required for snowmaking in order to maintain ecological flows during the base snowmaking month of December. Previous analyses (Vanham *et al.*, 2008b, 2009a) showed that for the future conditions no regional water balance between water availability and water demand will occur. Water availability will actually increase in winter, the current low flow period; however, water demands for snowmaking will increase. As the low flow period shifts to summer, critical situations are expected in future here. However, Vanham *et al.* (2009a) showed that on a regional level, no deficits will occur. Local temporal shortages may occur due to reduced groundwater recharge in summer months. However, this can be easily solved by connection pipes between the water supply infrastructure systems of the different municipalities.

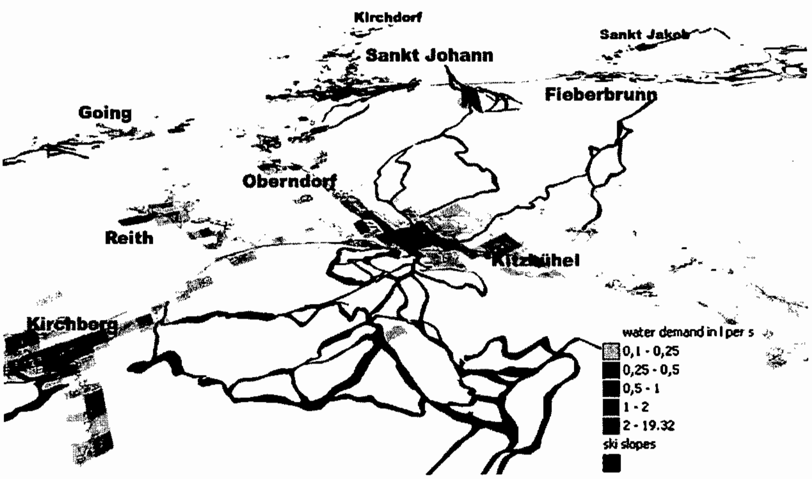

Fig. 9 Rasterised (250 m resolution) average water demand (L/s) in the eastern part of the case study area (not taking into account snowmaking).

Danube River basin lowlands The different water demand stakeholders include water for domestic purposes, small trade, industry, cooling water, agriculture, tourism (e.g. recreation and snowmaking), navigation, hydropower and ecology (including wetlands). Figure 10 displays the population density (persons per km^2) within the basin. Due to different national scales of resolution, the resolution is not very homogenous. However, it can roughly be seen that the mountainous areas are less populated than lowland areas. The large cities are concentrated in the lowlands, with several located along the river itself. Figure 10 shows the percentage of area equipped for irrigation in the Danube River basin. Irrigation, in many regions of the world by far the largest water demand stakeholder, is in most regions of the Northern Alps a minor water user. In the Southern Alps, however, a larger percentage of agricultural area is irrigated (such as the Po River valley). Notable irrigation regions within the Danube River basin are the Danube River region between Vienna and Budapest, and a large region along the lower Danube.

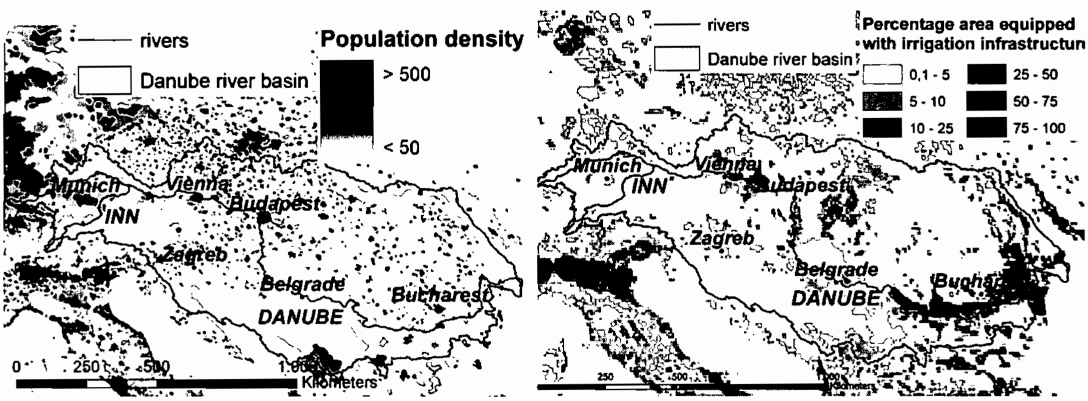

Fig. 10 (left) Population density (persons per km^2) in 2005 in the Danube River basin. Note the different resolution scales on national levels. Data Source: CIESIN. (right) Percentage of area equipped for irrigation in the Danube River basin, Data Source: FAO

A critical period in the lowlands of the Danube River basin regarding IWRM will be summer and autumn. Water availability will be the lowest, and the demands of different water demand stakeholders (such as irrigation or cooling water) will be the highest. As described before, the importance of alpine mountain water for the lowlands during these periods is even greater than at other times of the year. The detailed analysis of the case study within the Austrian Alps showed a shift in flow seasonality in the climate change scenarios. The seasonal snow cover is reduced both in amount and in duration. Due to the loss of this natural reservoir, flows in winter will be higher in future in the Alps. However, flows during spring to autumn will be lower. This means that the Alps will not provide the same amount of water for the lowlands during the critical seasons. More water will flow from the Alps in winter. River basin management plans will have to take these shifts in hydrological regimes into account.

CONCLUSIONS

The disproportional influence of the Alps for the Danube River basin was shown by means of the alpine Inn catchment. Its mean contribution to total discharge ranges from a minimum in winter to a maximum in summer. To explain these results the average monthly water balances (1961–1990) of three catchments (Fig. 3) within the Danube River basin are compared. Two mountainous ones (of which one is the case study area) and one lowland catchment.

The climate change scenarios for the case study area show a decrease in snow cover duration and snow accumulation, an effect dependent on elevation. A shift in flow seasonality is observed.

Total and base flow increase in winter, and decrease from spring to autumn. Spring peak flows decrease and the natural reservoir effect of the snow cover decreases.

The impact of this loss in natural storage reservoir on the IWRM in both the alpine region (case study) and the lowlands of the Danube River basin is discussed. Regional water availability within the alpine case study, however, is sufficient to provide all demand holders during the whole year. The shift of low flow period from winter to summer does not result in regional deficits, but can result in local deficits which can easily be solved. However, this shift in alpine water hydrology has a strong effect on water availability in the downstream Danube River. A critical period in the lowlands of the Danube River basin will be summer and autumn. In future there will be a reduction of the water contribution of the Alps during these critical seasons.

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Hydrological modelling and possible climate change impacts in a wetland system: the case of the Okavango Delta, Botswana

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Abstract An attempt has been made to model the flooding patterns of the Okavango Delta, Botswana, with the objective of reproducing the historical flooding patterns as well as water resources situation, and then to study the impact of possible regional climate change on the flooding patterns and/or size of the delta. The study employed a spatially-distributed surface water–groundwater interaction model which was calibrated for the current climate and then coupled with global climate model (GCM)-based climate change scenarios. The HadCM2 and UKTR simulated changes in the 2050s in relation to the baseline climate, and resulted in reductions in precipitation of 5.6 and 2%, and increases in temperature of 2.5 and 1.7°C, respectively. The simulated average annual recharge of the delta for the baseline period (1961–1990) was found to be 6.3 mm/year, which is 12% of the mean annual rainfall of 485 mm, and also comparable with regional recharge values. The 2050s GCM simulations with HadCM2 and UKTR scenarios were found to result in recharge values of 6.2 and 5.8 mm/year, respectively.

Key words climate change impact; flooding pattern; Okavango Delta; wetland; GCM

1. INTRODUCTION

Climate change associated with increased CO_2 and other greenhouse gases poses significant threats to many of the world's coastal estuarine and non-tidal wetland ecosystems (Kusler *et al.*, 1999). Some predictions indicate a warmer climate over southern Africa for the next century, accompanied by changes in precipitation patterns (Hulme *et al.*, 1996). Such changes will affect the ecological function of wetlands through changes in their hydrology, biogeochemistry and biomass accumulation. A rise in mean global temperature of 1–3.5°C over the next century, combined with reduced, stable, or even slightly increased total precipitation, could seriously impact on some freshwater wetlands (Kusler *et al.*, 1999).

Andersson *et al.* (2006) examined the impact of climate change impact in the Okavango basin. This involved use of the Pitman hydrological model configured for all the sub-catchments of the Okavango basin, with modifications to cater for wetland areas that were represented as reservoirs.

Previous studies have focused on understanding processes that influence the hydrology of the Okavango Delta (Ellery *et al.*, 1990). These studies led to the development of mathematical and hydrological models (Dincer *et al.*, 1978, 1987; SMEC, 1987; Scudder *et al.*, 1992; Gieske, 1997; Alemaw *et al.*, 2003), which provided information on the amount of water that could technically be abstracted from the delta. These models were mostly reservoir-type models (Dincer *et al.*, 1987; SMEC, 1987; Scudder *et al.*, 1992) that could not reproduce the spatially-distributed flooding patterns of the Okavango Delta. Time series of flooding patterns derived from remote sensing images provide an opportunity for calibrating a spatially-distributed hydrological model (McCarthy *et al.*, 2003). A spatially-distributed surface water–groundwater interaction model (SGIM) or Integrated Surface–Groundwater Model (ISGM) calibrated using time series of flooding patterns derived from satellite images (Chiyapo, 2006) is therefore used in this study. The approach followed was based on a GIS-based accounting of water balances at regional or sub-regional level (Alemaw, 1999; Alemaw & Chaoka, 2003).

2. THE STUDY AREA

The Okavango Delta is a large alluvial fan (Fig. 1), with an approximate area of 30 000 km^2, and is situated in northwestern Botswana (Bauer, 2004). The study concentrated on the flooding area of the Okavango Delta that falls with a square area bounded by 18.0–21.0°S and 21.5–24.5°E. The

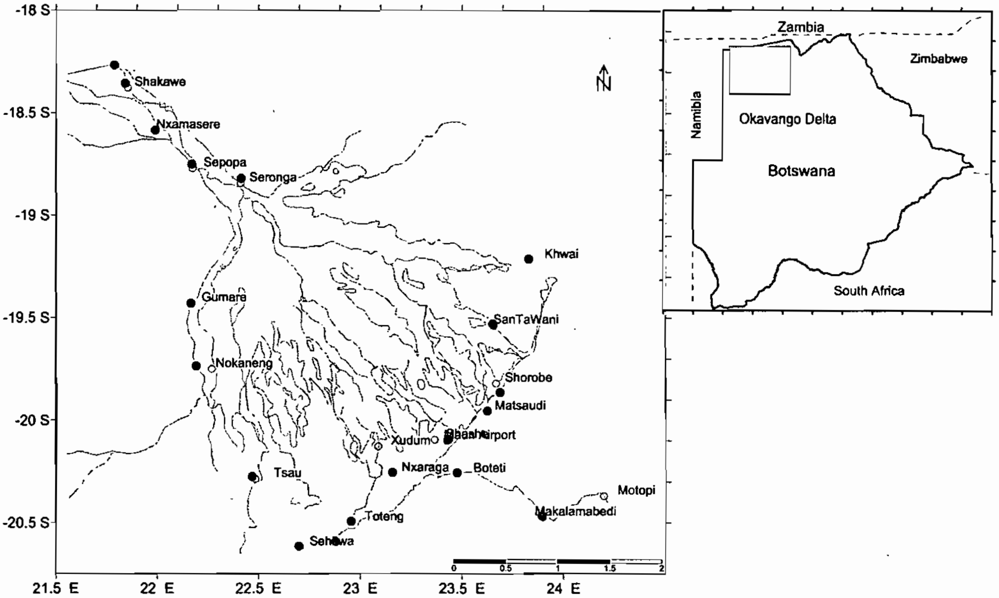

Fig. 1 Map of the Okavango Delta and surrounding localities in Botswana.

Okavango River, which originates in the tropical highlands of southern Angola, supplies the Okavango Delta with approximately 11×10^9 m^3/year (SMEC, 1987). During flood events, the extent of the inundated area increases from about 5000 km^2 to 6000–12 000 km^2, depending on the size of the flood (Bauer, 2004). Rainfall supplies about 500×10^6 m^3/year extra water (inflow) (Wolski & Murray-Hudson, 2005). Approximately 200×10^6 m^3/year leaves the Delta as surface runoff, and it is estimated that another 200×10^6 m^3/year flows through groundwater pathways. About 97% of the total input is ultimately lost through evapotranspiration (Andersson *et al.*, 2003). Currently there are three countries that share water from the Okavango basin: Angola, Botswana and Namibia. In all these countries, water demands are growing rapidly, which may lead to a reduction in inflow received into the Okavango Delta. Angola has old plans for hydroelectric power generation on the Cuito and Cubango rivers, while Namibia plans to increase irrigation (Andersson *et al.*, 2003, 2006).

3. METHODOLOGY AND DATA USED

3.1. Methodology

The hydrological model employed in the study was the Integrated Surface–Groundwater Model (ISGM) of the Okavango Delta (Chiyapo, 2006). This model was selected because it was designed as a spatially-enabled water balance model that worked well for representing the flooding pattern of the delta for the current or observed climatology. It was also easily interfaced with climate change scenarios available for the region to simulate post-climate change flooding patterns of the delta. The modelling philosophy of the ISGM is not fully described here. The model employs a spatially-distributed hydrological modelling approach in order to achieve a quantitative understanding of the spatial extent and temporal dynamics of the flood plains as a function of climate change. The ISGM has a surface water balance and a groundwater component. The surface water balance component of the ISGM solves the water and energy balance in the Okavango Wetland System, in terms of an integrated variable, called net vertical flux (F_{nv}) for any given climatic scenario. The groundwater balance component of the ISGM is based on subroutines of the well-known groundwater model, MODFLOW (McDonald & Harbaugh, 1988). It was possible to take advantage of the presence of sub-functions for wells, drains, recharge zones and river components in MODFLOW and these were introduced into the groundwater model component, which calculates the groundwater fluxes (McDonald & Harbaugh, 1988).

Calibration of the ISGM of the Okavango Delta (Chiyapo, 2006) for the baseline period (1961–1990), as well as post climate change simulations, were conducted using various error indices that were used to evaluate model performance. For the Relative Flooding Index (R_{FI}), maps for both the observed and the modelled flooding patterns for the baseline period were computed and compared. The R_{FI} is a relative measure of the number of months a pixel is inundated in relation to the overall period of simulation, for 30 years during the baseline period, and 60 years for the period spanning from 1990–2050s for the post climate change simulation case. An alternative second approach is the Standardized Residual Index (S_{RI}) of flooding patterns. The S_{RI} is a measure of model performance in terms of error at a pixel in simulating an observed pixel's status of flooding or non-flooding. The S_{RI} is used for calibration of the model through comparison of the differences between the modelled and observed flooding patterns. The error index, S_{RI}, is calculated as follows:

$$S_{RI}(x,y) = \sum_{t=1}^{N} \frac{F_{mod}(x,y,t) - F_{obs}(x,y,t)}{N} \tag{1}$$

where N is the number of months of the simulation period; $F(x, y, t)$ is the flooding pattern, a status of flooding assigned a value of one, or non-flooding state with a value of zero, with subscripts obs and mod referring to the observed and modelled situations, respectively, for simulation periods/time steps considered, and F is the two-dimensional flooding pattern in each pixel located at (x, y) at each simulated period. The value of N is taken as 360 months (for simulated months of the 30-year baseline period, 1961–1990) and 720 (for simulated months of the 60-year projected post-climate change horizon of the 1990s to 2050s). Expressed as dimensionless values or in percentages, S_{RI} depicts the relative comparison of the modelled flooding patters to those of observed ones.

3.2. Data used

The data used for the study involved information on: (a) time series of rainfall and other meteorological variables, and Okavango River inflow to the delta measured at Mohembo; (b) climate change scenarios from global climate models (GCMs) that generated regional climate change scenarios of monthly climatology for southern Africa from which the corresponding data for the Okavango Delta was retrieved; and (c) spatial flooding patters of the delta digitized from previous studies, which present average seasonal flooding areas (e.g. McCarthy, 2002; Bauer, 2004, Mendelsohn & Obeid, 2004). The Okavango River inflow to the delta recorded at Mohembo (18.29°S; 21.79°E) and rainfall and other meteorological data for Shakawe (18.38°S; 21.85°E) and Maun (19.98°S; 23.42°E) were used for the study. The data were obtained from the Department of Water Affairs, Botswana, through previous studies conducted by Alemaw *et al.* (2003).

Climate change experiments that have been conducted with GCMs show how the IPCC's mid-range scenario translates into recognizable patterns of climatic change over southern Africa (Hulme *et al.*, 1996). Possible changes in climate over southern Africa, which includes Botswana, have been retrieved from a variety of models put together in the scenario generator software (SCENGEN; Wigley, *et al.*, 2000). Precipitation and temperature during the baseline period (1961–1990) and the early 2050s for the two IPCC scenarios—UKTR and HadCM2 (UK) alternative scenarios under the IS92a emission—in 0.5° latitude/longitude grid cells over the Okavango Delta (18.0–21.0°S; 21.5–24.5°E) were retrieved. These data were coupled with the regional temperature and rainfall patterns. We resampled the climate pattern of the Okavango Delta's sub-regional climate over 1-km square grids and coupled it with the spatial extent seasonal maps. These flooding seasonal maps were reconstructed from identified pixels with the criteria that a pixel is flooded based on historical flooding of the main channels of the delta and the distance/proximity of individual pixels to the main channels, the historical seasonal rainfall and inflow into the delta measured at Mohembo – all constrained with the delta flooded area records available from McCarthy (2002) and McCarthy *et al.* (2003) and also digitized from Mendelsohn & Obeid (2004). The time series of observed monthly rainfall over the delta (represented by records at Maun and Shakawe) and the river discharge data of Mohembo for 1961–1990 were used to reconstruct the spatial identity or the state of each pixel in terms of being flooded or not by superimposing with the seasonal average monthly flooding areas presented in Mendelsohn & Obeid (2004).

4. RESULTS AND DISCUSSION

4.1. Climate scenarios

Climatic variables of interest (temperature and rainfall) are simulated on a monthly basis for the baseline period of 1961–1990. Due to the different philosophies used in the GCMs, the results of climate change simulations are not always the same between two or more models. In this report, results of the UKTR and HadCM2 are considered to be the primary scenario for making studies of potential impact and adaptive measures for climate change in the Okavango Delta of Botswana. The HadCM2 and UKTR simulated changes in the 2050s in relation to the baseline climate result in reductions in precipitation of 5.6 and 2%, and increases in temperature of 2.5 and 1.7°C, respectively. Among the 14 GCMs available in the SCENGEN software (Wigley *et al.*, 2000), the UKTR and HadCM2 scenarios were considered as the primary scenarios for the study as they both reproduced the current climate well in terms of rainfall and temperature. A comparison of observed and the UKTR mean annual rainfall for the baseline period, for Shakawe near the panhandle area is shown in Fig. 2. The observed annual rainfall at Shakawe for the hydrological years 1961–1990 is shown with the 95% upper and lower confidence limits.

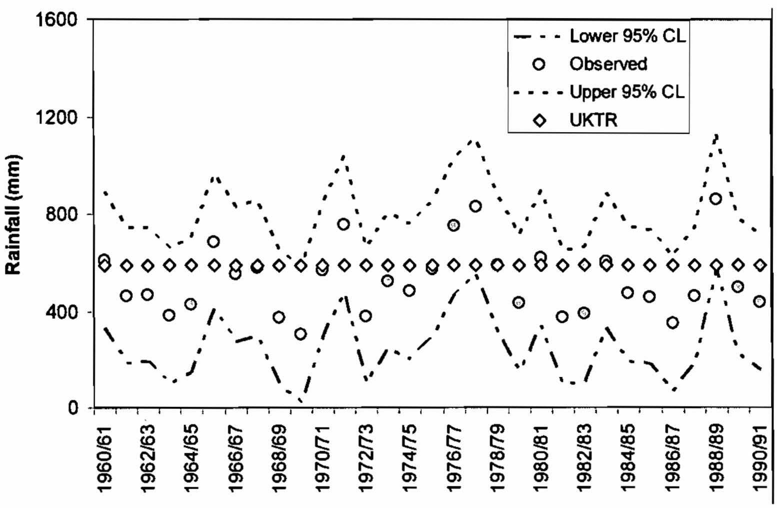

Fig. 2 A comparison of observed mean annual rainfall with the 95% upper and lower confidence limits, and UKTR scenario at Shakawe (close to panhandle area of the Okavango Delta) for the baseline period (1961–1990).

4.2. Calibration and simulation of spatial flooding patters

Results of calibration of the ISGM of the Okavango Delta for the 1961–1990 period (Chiyapo, 2006) are shown in Fig. 3 in terms of the S_{RI} maps. The S_{RI} values are presented as percentages from –100 to 100%, spatially over the Okavango Delta. It can be noted that the ISGM has adequately reproduced the historical flooding patterns looking at its performance for the baseline

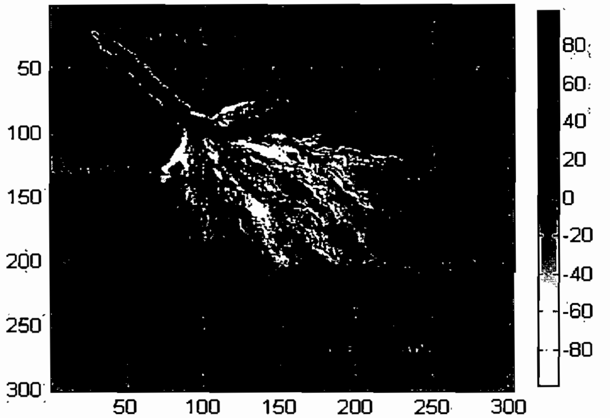

Fig. 3 Standardized Residual Index (S_{RI}) map of modelled and observed flooding patterns of the Okavango Delta for the baseline period (1961–1990).

period (Fig. 3). However, it should also be noted in the S_{RI} map that the model has overestimated the flooding patterns mostly on the fringes of the delta, especially the southern side, which can be attributed to the under-representation of the Kunyere and Thamakalane faults through which a lot of water drains away. This could be a problem area for future improvement on the modelling attempt of the current study. Also, some parts where the model has underestimated the flooding patterns can be observed in some areas of the delta along the main stream (around Mohembo station near Shakawe), which is known as the panhandle area, and other distributaries along southern fringes of the delta. Lake Ngami, further to the south of the delta has also been underestimated to some extent.

4.3. Flooding patters and recharge

Results of simulation of the ISGM of the Okavango Delta, for the GCM-induced scenarios, as well as baseline period, are available in terms of flooding extent of the delta, net vertical flux over the delta, actual evaporation, and recharge, among others. The UKTR was selected for a typical flooding month of May, for both the baseline and the core scenarios, and their flooding pattern maps, in terms of relative flooding patters, are reproduced and compared in Fig. 4.

Apart from flooding and surface water flow patterns in the delta, groundwater recharge from the delta is an important component that can be affected by possible climate change. Figure 5 shows the expected recharge of the 2050s under the UKTR climate change scenario. The average annual recharge of the delta for the 1961–1990 period was found to be 6.3 mm/year, which is 12% of the mean annual rainfall of 485 mm. From a regional regression or relationship between annual recharge and precipitation from study of large areas (40–374 200 km^2) in the Kalahari environment (Scanlon *et al.*, 2006), where mean annual recharge (in mm) is given as 0.03 times mean annual rainfall (in mm) minus 2.6, a recharge value of 11.95 mm/year can be obtained. This is quite reasonable and in good agreement for the baseline mean annual precipitation of 485 mm/year. The GCM simulations with HadCM2 and UKTR resulted in recharge of 6.2 and 5.8 mm/year, respectively. A comparison of mean annual recharge and the 95% upper and lower confidence limits for the baseline period and that expected in the 2050s among the 1-km square pixels in the Okavango Delta is summarised in Fig. 5. In general, the recharge pattern across the delta varies from 4 mm/year in the southwest to 8 mm/year in the northeastern part of the delta. It can also be noted that the recharge pattern generally follows the regional precipitation pattern, as well as the net vertical flux in the delta system, which is affected by precipitation, evapotranspiration and surface water storage and flow in the delta. The above fluxes over the delta are all simulated by the ISGM model (Chiyapo, 2006), for both the baseline and climate change scenarios.

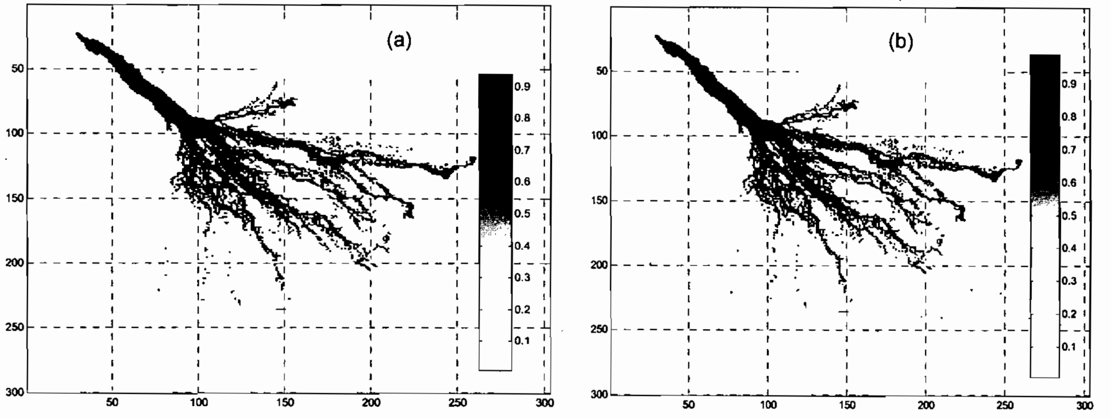

Fig. 4 The flooding pattern maps in terms of dimensionless R_{FI} values for the (a) modelled baseline period and (b) climate change scenario (UKTR) in the 2050s for the month of May.

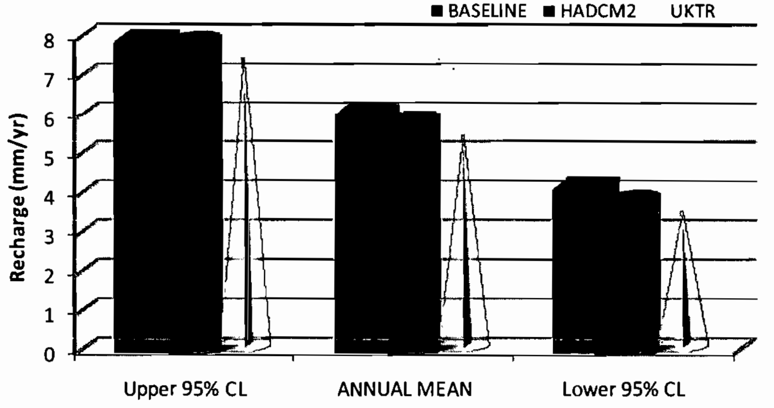

Fig. 5 Comparison of mean annual recharge and the 95% upper and lower confidence limits (CL) for the baseline period and that expected in the 2050s among the 1-km square pixels in the Okavango Delta for the baseline and the post-climate change HadCM2 and UKTR scenarios.

5. CONCLUSION

The results of the study of the impact of possible climate change on the flooding pattern of the Okavango Delta reveal the challenges of water resources availability under regional climate changes. Given the uncertainties in predicted climate change scenarios used in this study, climate change is likely to affect water allocation which depends on the water availability in the Okavango Delta. This calls for prioritized and integrated water resources management in this ecologically interesting wetland system of the delta and its drainage basin system.

The recharge pattern across the delta for the 2050s under the UKTR climate change scenario is expected to vary from 4 mm/year in the southwestern part to 8 mm/year in the northeastern part of the delta. It can also be noted that the recharge pattern generally follows the regional precipitation pattern for both the baseline period and the UKTR climate change scenario.

Climate change scenario results as presented above using the UKTR scenario may be realized earlier or later depending on the sensitivity of the climate system. If the system is not extremely sensitive to the regional and/or global climate pattern, changes may be delayed until much later than the simulation period (2050s). If the climate system is very sensitive, then climate change under the UKTR scenario could occur earlier than 2050. The timing of the UKTR scenario can thus be represented as alternative patterns of climate change. Thus, in the case of a delayed response, this may be represented by other models such as the OSU (USA) GCM, in which annual rainfall increases of up to 20% or more may be expected across northern Botswana where the Okavango Delta is located.

The flooding pattern and recharge simulated in the current study, and the climatic change scenarios discussed above, do not address land cover changes, channeling or blockage of the channels, flow redistribution and real-time geological faulting/folds of the complex wetland system of the delta, some or all of which could come about due to natural or human activities. This realization complicates further the climate change scenario interpretation and use in vulnerability assessments. In spite of these difficulties, the approach we used to model the surface–groundwater interaction, together with coupling externally with GCMs remains a recommended vulnerability assessment tool. It is a preliminary attempt to provide the best source of information to study the possible impacts of regional climate change on the flooding pattern and behaviour of the Okavango Delta.

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Examples of possible water management measures for elimination of climate change impacts on water resources

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Abstract The paper describes the results of a study that aimed to propose adaptation measures suitable for eliminating or reducing water scarcity problems in two basins that are administered by Ohře River Basin authority (of the Czech Republic), which initiated and also sponsored the implementation of the project. The basins are located in a low precipitation region in the Czech Republic and their hydrological conditions have deteriorated during the last two decades, and are probably already affected by impacts of the initial stages of climate change. The main problems occur during drought periods, when river discharges drop below minimum ecological flows the river water is excessively polluted by wastewater discharges from urban areas and the water resources are insufficient to meet water use requirements for the purposes of agriculture, which is a very important economic sector in this region. The results of the study also show controversial problems associated with implementation of the EU Water Framework Directive.

Key words water scarcity; drought; climate change; adaptation measures; Bilan model; EU Water Framework Directive

INTRODUCTION

Since 2000, the water protection and integrated surface and groundwater resources management in member countries of the European Union have been governed mainly by the implementation of the EU Water Framework Directive (WFD – Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000 establishing a framework for Community action in the field of water policy). During its implementation, it was increasingly evident that water management planning will also have to take into account projected impacts of climate warming on the water cycle and water resources. In addition to the predicted impacts, the results of meteorological and hydrological monitoring indicate that climate change has already been reflected in hydrometeorological variables. It is also projected that the changing conditions will be associated with an increasing occurrence and severity of droughts and floods. Hydrological extremes are addressed in subsequent legislation activities of the European Union.

On 18 July 2007, the European Commission adopted a document on Communication from the Commission to the Council and the European Parliament addressing the challenge of water scarcity and droughts in the European Union. Subsequently, the Environment Council adopted a drought management plan report, and in 2008, the European Parliament adopted a report on addressing the challenge of water scarcity and droughts in the European Union (2008/2074(INI)).

These documents discuss a number of challenges associated with drought and water scarcity and classify adaptation measures according to several aspects, such as those to rationalize water demand (infrastructure improvement and modernization, foster saving, re-use and recycling), which should be the priority, and those that address water demand with infrastructures (regulation, intake, desalination, transport, interconnection, etc.), which should be considered as an option when the previous measures have been exhausted, including effective water pricing policy and cost-effective alternatives. These remain subject to EU legislation, in particular to all requirements of the Water Framework Directive.

The implementation of the Water Framework Directive, and particularly preparation of river basin management plans according to the Directive also initiated discussions concerning the priority problems in the Czech Republic. At the level of the national administration authorities, the main problem discussed concerns preservation or revocation of protected areas that were delimited previously to protect localities where rivers could be dammed to secure water for future water supply purposes.

These discussions initiated implementation of a number of research projects, which have been focused mainly on future availability of water resources to meet water demands in individual regions and specific localities. This paper describes the results of one of these studies, which shows the reality in two small neighbouring basins, whose inhabitants are already facing increasing problems associated with water scarcity in the basins. The knowledge resulting from the study also contributes to rational decision-making in the above priority discussions.

DESCRIPTION OF THE BASINS AND THEIR WATER SCARCITY PROBLEMS

The two neighbouring basins, the Blšanka and Liboc river basins (Fig. 1), are located in the north-western part of Bohemia (the Czech Republic). Their basin areas are 482.5 km^2 and 339.3 km^2 and mean altitudes 404 m and 428 m, respectively. Mean annual precipitation is predominantly in the range 400 mm to 500 mm (locally 650 mm), depending on the altitude and exposure of the slopes. Mean annual air temperature is between 8 and 9°C.

Low precipitation and high potential evapotranspiration (relatively high air temperature) are the main causal factors of very low runoff. Mean annual discharge of the Blšanka River at Holedeč (water gauging station) is 0.711 $m^3 s^{-1}$ (runoff of 59 mm/year) and the mean discharge of the Liboc River at Libočany station is 0.928 $m^3 s^{-1}$ (86 mm/year). During drought events, the Blšanka River dries up fully (Fig. 2). Between the periods 1969–1990 and 1991–2006, the basin air temperature increased by 0.7°C, consequently increasing potential evapotranspiration, while the mean annual runoff decreased from 75 mm during 1969–1990 to 51 mm in 1991–2006. Agriculture, including hop production, dominates the land use of the basins (Fig. 3).

However, this is associated with high water use requirements for irrigation purposes. The main problems occur during dry years in the Blšanka River basin. Dry years were recently very frequent (mainly in 2001, 2004 and 2007) and consequently the water use for irrigation purposes increased by a factor of three as compared to that in 1998. In addition, high quantities of groundwater are abstracted for drinking water supply purposes. It was reported in Kněžek (2008) that as a result of excessive groundwater abstractions in the 1970s, the groundwater level in Holedeč locality (Blšanka River basin) decreased by 13 m.

In addition, water quality is greatly affected by discharges of wastewaters from urban areas, and therefore the combined impact of decreasing flows and water pollution pose high risks to the ecological status of the rivers.

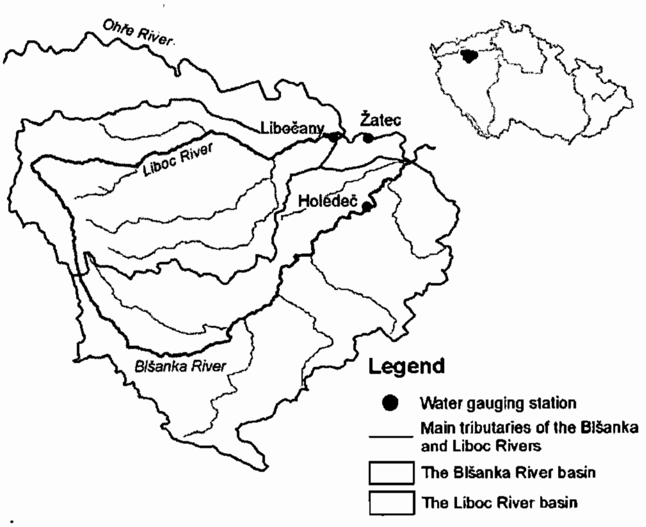

Fig. 1 Blšanka and Liboc river basins.

Fig. 2 The Blšanka River in July 2007.

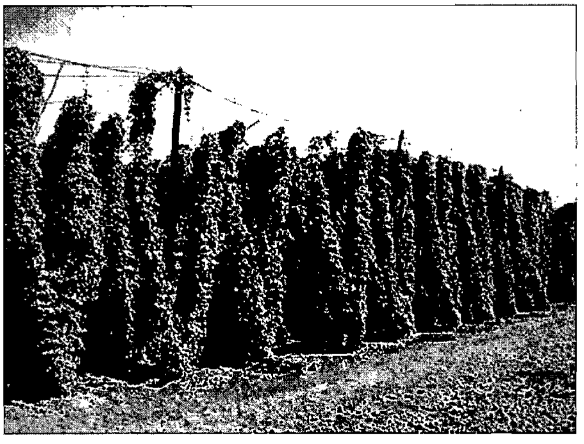

Fig. 3 Hop garden in the Blšanka River basin.

The main objective of the study was therefore to find measures whose implementation would ensure sufficient water for maintaining minimum ecological flows, mainly in the Blšanka River, and meet the water use requirements for the irrigation purposes.

DATA, METHODS AND TOOLS

For the purposes of the study, river flow data were available for the period 1968–2007 from Holedeč station, and for 2001–2008 from the Libočany station. Basin precipitation was calculated by using observations from 11 precipitation stations located in the basins. Air temperature series and relative air humidity series were derived from several climatological stations located in the basins and their vicinity.

Hydrological series for the purposes of the study were simulated by using the Bilan model, which is a water balance model developed by the T. G. Masaryk Water Research Institute for simulation of daily or monthly time series of water cycle components. The Bilan hydrological model is described in Tallaksen & Lannen (2004) and its executable version is available, together with example data for its application, on a CD, which is attached to the textbook. Input data for the Bilan model include primarily time series of monthly precipitation, temperature and relative air humidity. The model simulates time series of monthly potential evapotranspiration, actual evapotranspiration, infiltration across the land surface and recharge from the soil to the aquifer.

The output of the model includes monthly series of water storages in the snow pack, soil and aquifer. All these hydrological variables apply to the whole catchment. Furthermore, three runoff components, i.e. surface runoff, interflow and base flow (groundwater discharge) are calculated at the outlet of the catchment. The eight free parameters in the Bilan model are calibrated on the basis of minimising the differences between simulated and observed outflow from the basin.

The Bilan model is used for simulation of the series of water cycle components for current climate conditions and also those affected by climate change by applying climate change scenarios. The parameters of the model are first calibrated by using observed (or naturalised) series of basin precipitation, air temperature, relative air humidity and river flows, and they are subsequently used for simulation of water cycle components for the current conditions and conditions (the meteorological series) that are modified by using the climate change scenarios (2085 as reference year).

The T. G. M. Water Research Institute and other institutions in the Czech Republic at present use the climate change scenarios that have been derived from the results of the PRUDENCE (Prediction of regional scenarios and uncertainties for defining European climate change risks and effects) project. The scenarios are based on simulations by the HIRHAM regional atmospheric climate model (Undén *et al.*, 2002) and RCAO Atmosphere–Ocean model (Döscher *et al.*, 2002) and SRES emission scenarios A2 and B2 (Nakicenovic *et al.*, 2000). Scenario A2 ("pessimistic") assumes higher temperatures than the B2 scenario ("optimistic").

Water retention capacities necessary to be secured in the basins for maintaining the minimum ecological flows and water use requirements were calculated using the EXDEV (Experiments with Deficit Volumes) computer program that has been developed by Czech Hydrometeorological Institute (Řičica & Novický, 1994). Statistical analysis of the deficit volumes and approximation of their frequency curves using the Weibull probability distribution was performed using Matlab software.

RESULTS

In the description of the results, the main attention is paid to Blšanka River basin, whose water scarcity problems initiated implementation of this project. In order to meet current requirements for maintaining minimum ecological flows and water use demands, it was specified that the river flow should not drop below 0.115 $m^3 s^{-1}$ (0.8 mm/month). For the future, it was predicted that the water use demands for irrigation purposes could increase due to climate change impacts and therefore an additional requirement was to ensure river flows in the Blšanka in the growing season of 0.189 $m^3 s^{-1}$ (1.3 mm/month).

The results of statistical analysis of the flow series are illustrated in Fig. 4, which shows probability distributions (in the form of probability fields) of monthly flows in individual months of a year and minimum required flows for several alternative flow series. During August, which is the driest month, the probability of ensuring the flow requirement is 90–95% for conditions unaffected by climate change (derived from the period 1969–1990), 80–90% for conditions probably affected by initial stages of climate change (period 1991–2006), 70–80% for the optimistic scenario (HIRHAM B2) and 60–70% for the pessimistic scenario (HIRHAM A2). If we include the additional requirement for future water supply demands for irrigation purposes, the probabilities would drop to 40–50% for HIRHAM B2 and to 30–40% for HIRHAM A2.

For making decisions about which measures would be sufficiently effective, it was necessary to derive volumes of water, which are missing during drought events (deficit volumes) for maintaining the required flows. Results of the application of the deficit volume method (Tallaksen & Lannen, 2004) by using EXDEV showed that the storage capacity for ensuring ecological flows during the period 1969–2008 would be $0.51 \times 10^6 m^3$ (deficit in 2007). During the period fully affected by climate change, the ecological flow would be secured with 95% probability by a retention capacity of $1.89 \times 10^6 m^3$ for HIRHAM B2 climate change scenario and $2.47 \times 10^6 m^3$ for HIRHAM A2. For the alternative, which includes water irrigation requirements, the necessary storage capacity would be in the range from $3.26 \times 10^6 m^3$ to $5.75 \times 10^6 m^3$.

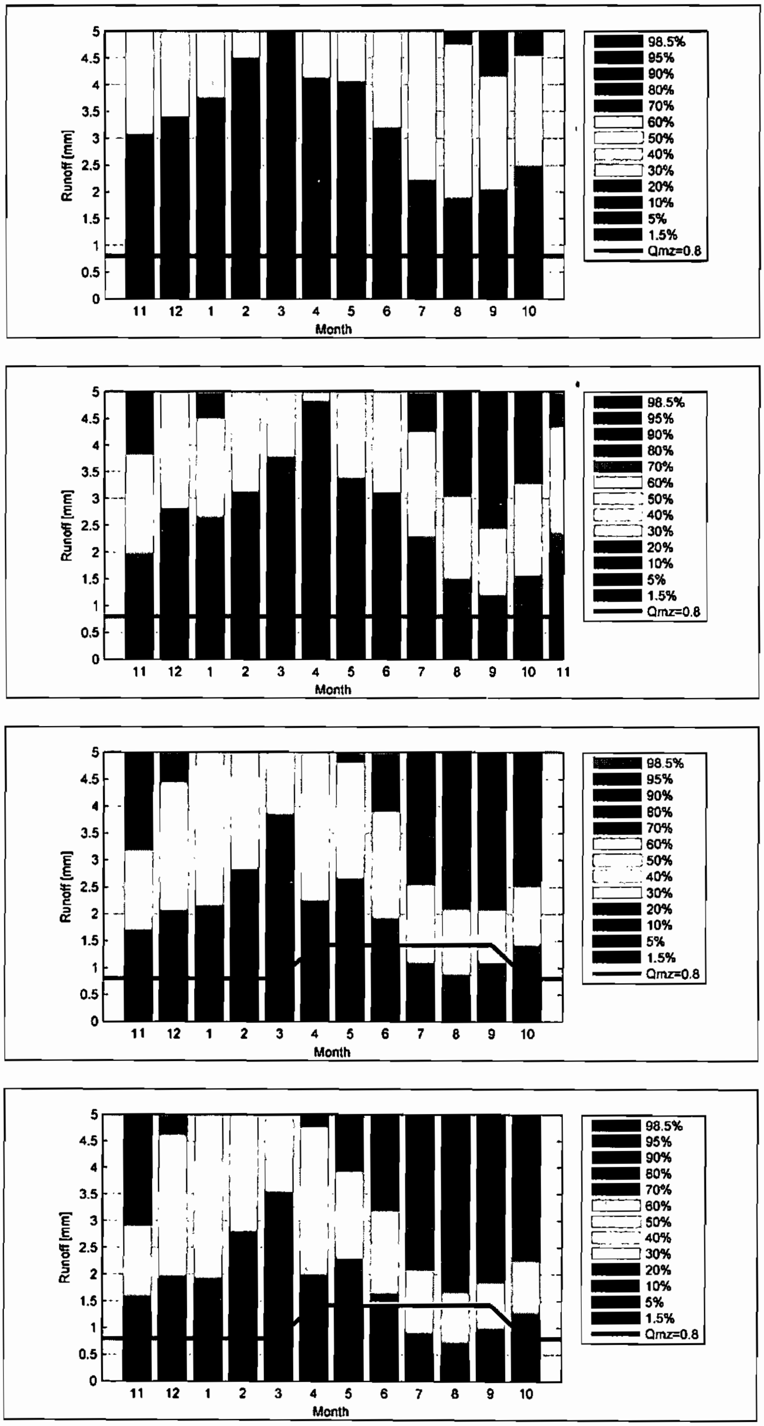

Fig. 4 Probability distributions of monthly flows in individual months of a year and minimum required flows, from the top: 1969–1990, 1991–2006, HIRHAM B2, HIRHAM A2.

POSSIBLE MEASURES

According to new policies adopted in the European Union and also in the Czech Republic, river basin management planners should preferably apply measures that do not disturb natural conditions, or even those which contribute to the naturalisation of rivers and their basins. These political trends initiated implementation of projects, which aim to assess different types of measures in terms of their potential (effectiveness). Potentials of the individual types of measures are discussed, e.g. in Novický (2008), and the results of a valuable assessment of biotechnical measures in the landscape are reported in methodology developed by Soukup *et al.* (2008). The methodology includes a catalogue of measures that can be implemented in order to reduce surface flow, to regulate outflow from drainage systems, to increase water retention in a landscape or to improve water quality, and the individual measures or their types are discussed in terms of their primary and secondary effects. There is no doubt that the individual measures, and particularly their suitable combination, can improve hydrological and particularly ecological conditions of rivers and their basins. However, the effectiveness of these measures in terms of river flow regulation is very low compared to the water scarcity problems discussed in this paper. The main problems of these measures are the relatively low water retention capacity stemming from their implementation and the mainly short-term effects of water flow regulation.

It was therefore evident that measures based on land and landscape improvements are not able to ensure flow requirements for meeting the water demands, nor for maintaining the minimum ecological flows, and therefore structural and organisational measures will have to be implemented.

In schemes implemented previously, possible augmentation of water resources in the Blšanka and Liboc basins was resolved by constructing several small reservoirs, whose purpose was to store water to be used for irrigation purposes during drought periods. However, these reservoirs were lately used for fish farming and therefore re-introduction of their original function would require reduction of their fish farming function. In addition, the capacity of the reservoirs is insufficient for meeting the water requirements, particularly in future.

The list of protected localities suitable for construction of new reservoirs includes four areas located in the Blšanka and Liboc basins. After reviewing present conditions for construction of a new reservoir in one of these localities, it was concluded that Hlubocká Pila on the Liboc River could be the most appropriate site, mainly because of the relatively high precipitation and therefore runoff in the basin of the Upper Liboc, and the fact that the potentially flooded area is not urbanised. The retention capacity would be probably sufficient to meet the water requirements, but an additional problem was the fact that the water from the reservoir would have to be transported for a distance of 20 km into the Blšanka River. In addition, the transported water could improve the flow conditions only in its lower reach. Subsequent project activities were therefore aimed at finding suitable localities for construction of additional reservoirs in the Blšanka basin and assessing these localities in terms of ecological and economical aspects. These localities are shown together with a final solution in Fig. 5.

Attention was also given to interactions between river flow and base flow, which is an important component of total flow in a river, particularly during drought periods when river flow is almost exclusively fed from groundwater. In was shown in Kněžek (2008) that groundwater level around the Lower Blšanka River dropped significantly during the 1970s and 1980s due to high groundwater abstractions for drinking water supply purposes. As a consequence, the water in the Blšanka River infiltrates into the alluvial plains, where it contributes mainly to evapotranspiration. This fact reduces the flow conditions in the Blšanka, particularly during low flow periods. These unfavourable conditions could be improved by flooding a former gravel pit located in the vicinity of the river with excess water during flood events. The accumulated water would gradually infiltrate and augment groundwater in the alluvial plains.

CONCLUSION

Measures that are appropriate for improving ecological conditions or problems associated with possible water scarcity in individual basins depend on a number of factors, the most important

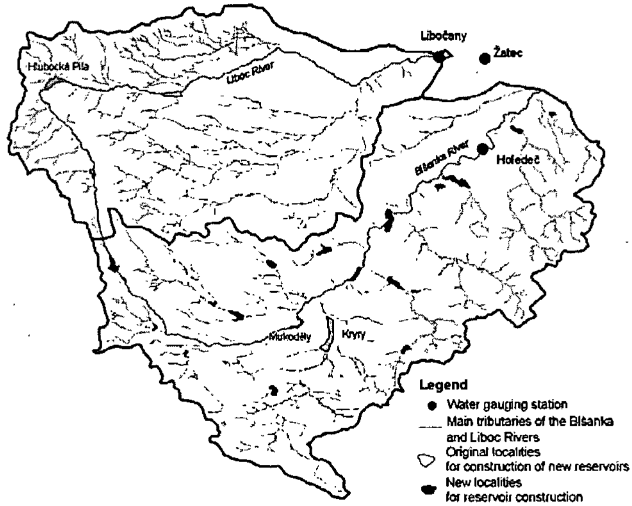

Fig. 5 Potential localities for construction of a small water reservoir in Blšanka and Liboc River basins and location of finally proposed new reservoirs (Hlubocká Pila, Mukoděly, Kryry).

of which will include the severity of the problem. In the basins of the Blšanka and Liboc rivers, the problems associated with water scarcity probably cannot be resolved without ensuring new water storage capacities. In terms of the Water Framework Directive, recently implemented in member states of the European Union, the construction of new reservoirs is undesirable because it will highly modify the upstream reaches of the rivers. However, one of the main objectives of the Directive is to maintain or reach good ecological status of surface water bodies. These facts will have to be carefully considered in decision making and river basin management planning in the basins of the Blšanka and Liboc rivers. New developments in this policy in the Czech Republic are reflected in the fact that new amendments to the Water Act will incorporate adoption of areas protected for possible construction of water reservoirs and a list of these areas will be specified in secondary legislation after implementation of relevant feasibility, ecological and economic studies.

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Use of high resolution climate change data to simulate hydrological change at the small catchment scale

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Abstract The spatial resolution of common output of global change models is too coarse to be used for impact studies in small catchments. To overcome this limitation, the German Environmental Agency (Umweltbundesamt) has published a data set with a grid size of 10 km (REMO data set), which is suitable for small scale simulations. We used this data set to analyse the effects of global change on the small Kielstau catchment (50 km^2), situated in northern Germany. In a first step we analysed the differences between measured values and the REMO base scenario (C20) of temperature, precipitation and discharge. Compared to measured values, the simulated winter temperatures, annual precipitation and the discharge values were much higher. In a second step, simulations were carried out with the eco-hydrological model SWAT, which was calibrated with measured runoff data. A comparison between the C20 and three different IPCC scenarios (B1, A2, A1B) showed a decrease of the summer discharge. However, the interpretation of these results is difficult because of the high precipitation in all REMO climate data sets, all simulated values with scenario data are still higher than the measured discharge.

Key words climate change; modelling; SWAT; REMO; IPCC scenarios

INTRODUCTION

It is an accepted fact that the global climate is changing and that the world is getting warmer and wetter (IPCC, 2007). The main tools of climate change research are the so-called "Global Circulation Models" (GCMs), see e.g. Roeckner *et al.* (1996) and McGuffie & Henderson-Sellers (2004), which simulate the global climate with a minimal grid size of 250 km for Germany. With growing public awareness there is also a growing demand for information about the consequences of global change at the local and regional scale. These studies are still very rare for two reasons: first, grid size of climate models is only approx. 250 km – thus neglecting many local climatic effects caused by topography or the small patches of typical agricultural land use. The second reason is that the tools and the data base used for regional modelling are completely different from the global ones. To make local analyses easier, the German Ministry of the Environment has funded the creation of a downscaled climate data set at a 10 km resolution for Germany and the Alps, the so-called REMO data set (Jacob *et al.*, 2007).

The aim of this study is: (1) to compare the REMO base scenario data (C20, 20th century) with measured climatic data, and (2) to use the SWAT model to compare the discharge simulations based on measured data and present and future REMO data set for the Kielstau catchment, a small lowland catchment in northern Germany.

METHODS

Kielstau catchment

The study was carried out in the Kielstau watershed. It is located in the region of Schleswig-Holstein, northern Germany, and covers an area of about 50 km^2. The catchment has a rather flat relief, with a maximum elevation difference of 50 m. Soils mainly consist of Gleysol, Podsol and Luvisol, among which Gleysol belongs to the major wetland soil types. Most of the land in this catchment is used for agriculture (87%), forest and urban land use share the remaining area. Average annual precipitation is around 860 mm, and evaporation approx. 400 mm (Schmidtke, 1999). For a complete description see Schmalz *et al.* (2008a,b).

REMO Data Base and Scenarios

The REMO model (MPI, 2009) is based on the European climate model of the German Weather Service (DWD) (Majewski, 1991) used for weather prediction with a spatial resolution of about 10 km. The regional model is driven by climate input data for the scenarios from the ECHAM5/MPI-OM models (Jungclaus *et al.*, 2006) and produces a data set in hourly resolution of several climate parameters. For this study, we used daily mean or sum values of precipitation, temperature, wind speed, humidity and sunshine duration as input for the SWAT model.

The data set is available for the following ECHAM IPCC-scenarios: (1) the climate of the 20th century (C20); (2) the sustainable development scenario (B1); (3) constant population, fast economic growth with new technologies (A1B); and (4) increasing population and slow technological development (A2) (see Nakicenovic *et al.*, 2000, for a detailed description of the IPCC scenarios). The temperature rise (Fig. 1) of the B1 scenario is the lowest value with 2°C followed by 2.6°C for A2 and nearly 2.8°C for A1B. The increase is higher in winter and autumn and lower during the summer months. The days with frost and ice decrease for all scenarios from 40 to approx. 5 days.

The REMO base scenario (named 1950–2000, C20) cannot be compared directly to the real data from this period, but at least the long term mean values should be similar. We will test this assumption with the precipitation, temperature and discharge values. In addition, we also check the spatial distribution of the climate variables.

The SWAT Model

The SWAT model (Soil and Water Assessment Tool, version SWAT2005 from 2005, Arnold *et al.*, 1998) is an eco-hydrological model, developed to assess the long-term impact of management practices on water, sediment and nutrients of meso- and macroscale catchments (Arnold & Fohrer, 2005; Neitsch *et al.* 2005). The parameterisation, calibration and validation of the model for the Kielstau catchment are already published in Fohrer *et al.* (2007) and Schmalz *et al.* (2008b).

RESULTS AND DISCUSSION

This section is divided into two parts: first we compare the measured climate and discharge data to the REMO base scenario of the 20th century (C20), second we analyse the differences between the C20 and the three IPCC scenarios for the discharge of the Kielstau catchment.

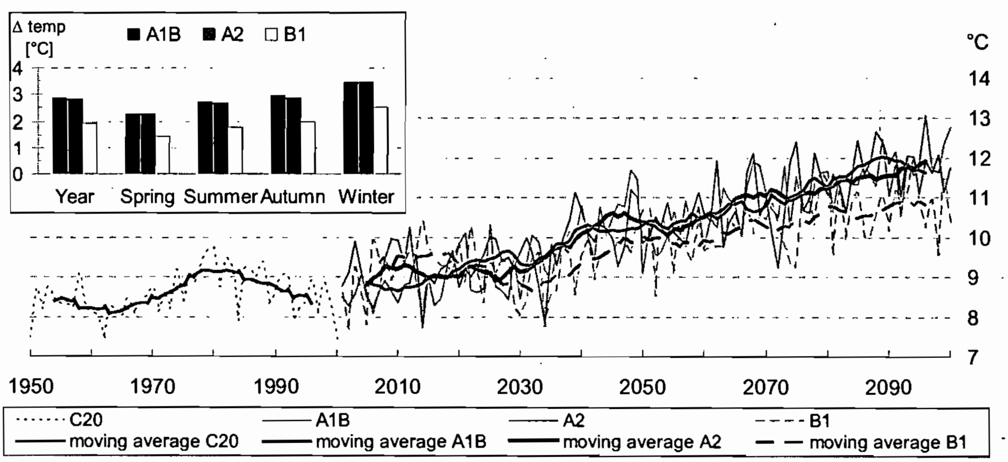

Fig. 1 Air temperature of the REMO Data set (original data (thin lines), 9 year moving average (thick lines). Inset: Changes in seasonal and annual mean values of the last 30 years (2071–2100).

Temperature

Seasonal distribution Figure 2 shows the seasonal distribution of the measured and simulated temperatures. The difference between measured and the C20 values are low, but there is a systematic deviation in the winter months: the C20 values from November to April are approx. 1°C higher than the measured values. The differences between the IPCC scenarios and the C20 values are more or less constant (Fig. 2(a)). In winter, the modelled temperatures of the scenarios are up to 5°C higher than the measured values, in summer only 2°C.

Spatial distribution The spatial distribution of modelled temperature (Fig. 3) is similar to the measured one at station Meierwik, the north–south and east–west gradients were met during the whole year. However, the temperatures in winter at the coast of the North Sea are up to 3°C higher.

Precipitation

Precipitation is one of the main inputs into the hydrological system, it is therefore very important for an eco-hydrological simulation that amount and seasonal distribution are similar to the measured present one.

Seasonal distribution Figure 3 shows the seasonal distribution of measured and modelled precipitation. The measured values are nearly always lower than the C20 scenario and the seasonal distribution is also different: the amount during the summer months are much higher. The scenario values are lower in summer and higher in winter and thus approach the trend of the measured values. This makes it very difficult to model ecological effects of climate change, because the deviating pattern of precipitation is probably much higher than the expected changes due to climate change.

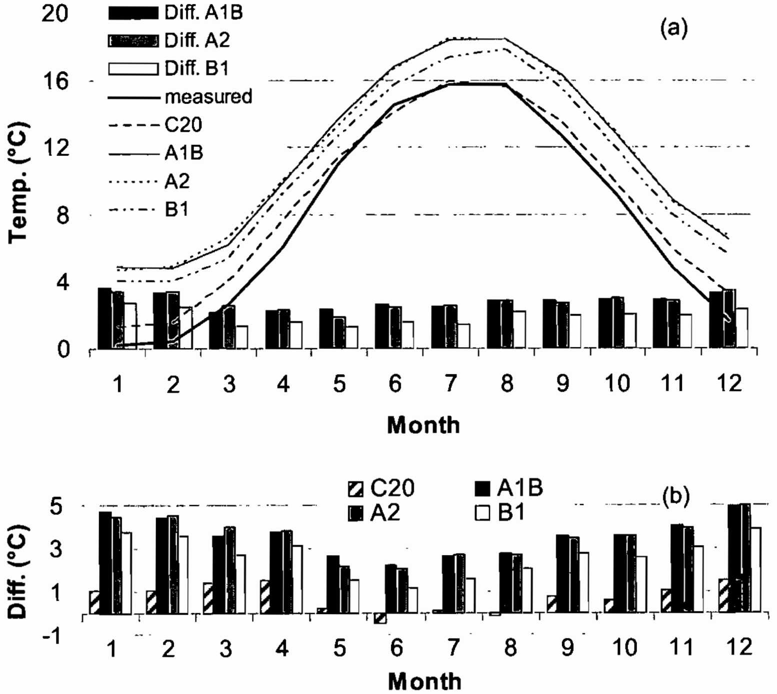

Fig. 2 Seasonal distribution of measured air temperature and the REMO scenarios (last 30 years) for the Kielstau catchment and deviations from the C20 (a) and measured data (b).

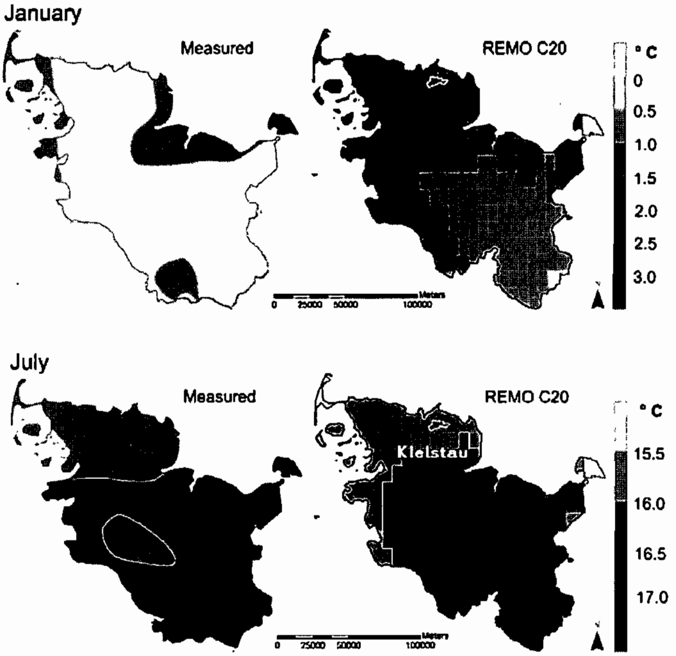

Fig. 3 Spatial distribution of measured and REMO C20 temperatures.

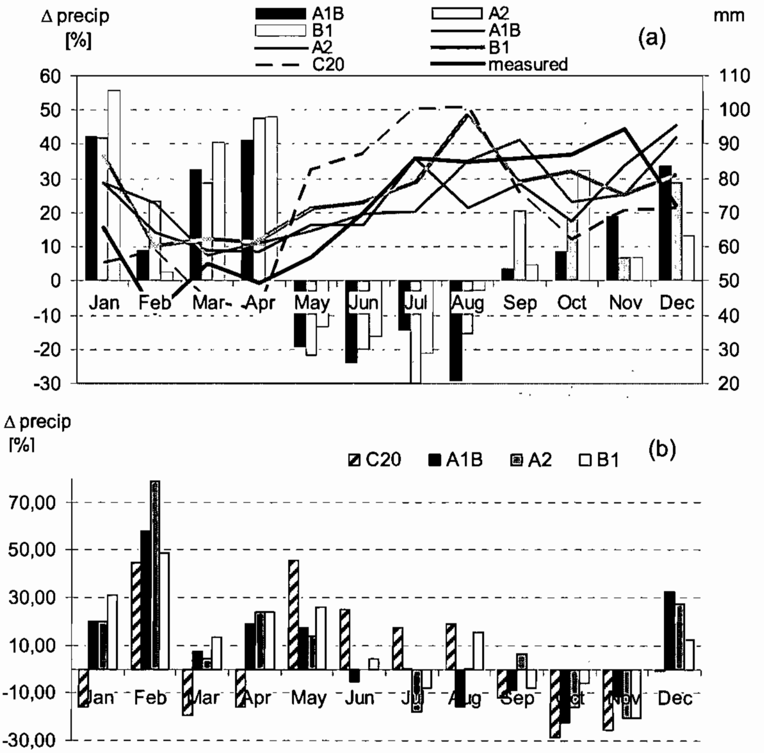

Fig. 4 Seasonal distribution of measured precipitation and the REMO-C20 scenario for the Kielstau catchment with: (a) percentage of C20 value, and (b) percentage of measured values.

Spatial distribution A second question is whether not only the absolute values, but also the REMO spatial distribution, match the reality. Figure 5 compares the precipitation for Schleswig-Holstein. The position between the North Sea and the Baltic Sea makes it rather difficult to model the climate; it is a challenge for the downscaling procedures. Despite its flat topography, there are distinct gradients of temperature and precipitation north–south and in an east–west direction. Especially at the North Sea coast the modelled precipitation is neàrly 1000 mm higher than the measured values. The differences get lower to the east, but the values for the Kielstau catchment are still higher than the measured ones. This is the reason why a comparison between measured data and the scenarios is not possible.

Climate change impact on the discharge of the Kielstau River

Assessing the effects of climate change on the discharge (simulated with the SWAT model) of the Kielstau River was the final goal of this project. First, we present the changes in the seasonal distribution of discharge (Fig. 6), second we analyse the changes in the extreme values and finally the flow duration curve based on daily values.

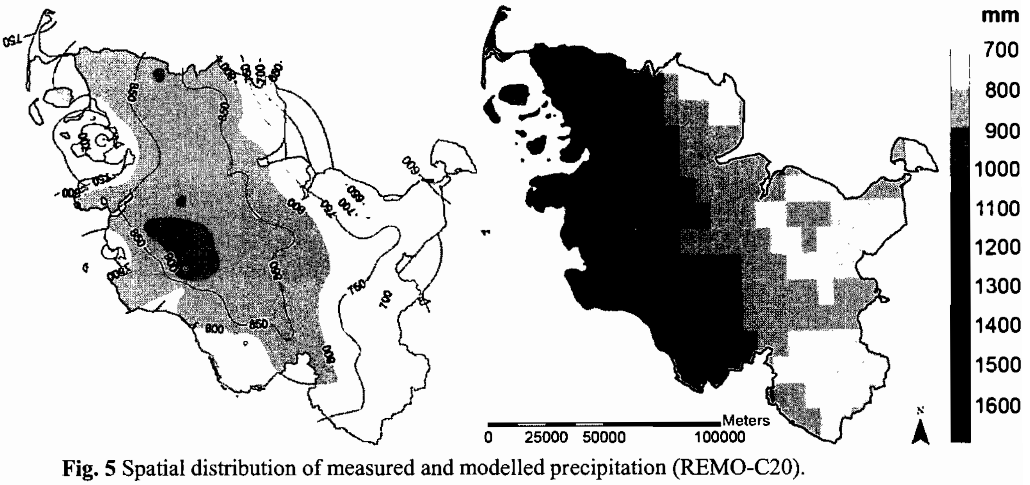

Fig. 5 Spatial distribution of measured and modelled precipitation (REMO-C20).

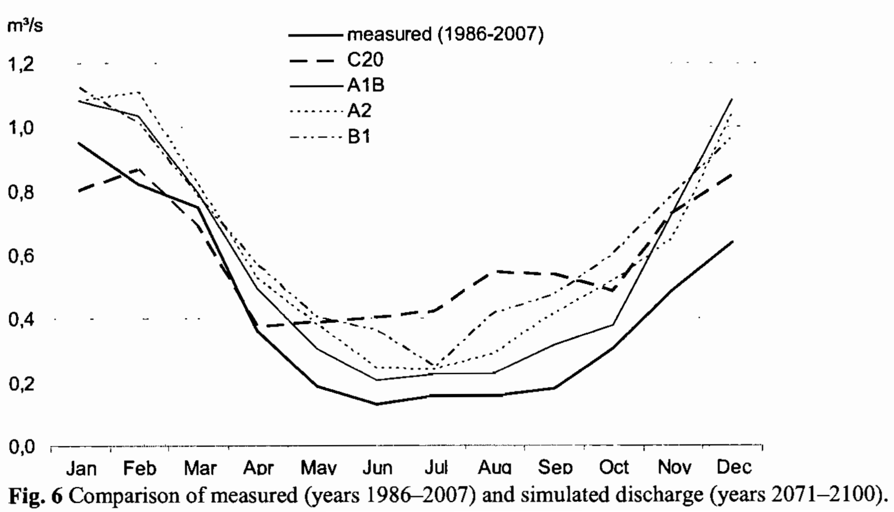

Fig. 6 Comparison of measured (years 1986–2007) and simulated discharge (years 2071–2100).

Seasonal distribution: In Fig. 4 we have shown that summer precipitation has increased and winter precipitation has decreased in the REMO base scenario C20 compared to observed values. These changes have a direct effect on the seasonal distribution of simulated discharge (Fig. 4): in the summer months (May–September), discharge of the C20 scenario has nearly doubled compared to the measured values, only from March to April are there are no differences. The change in precipitation in the IPCC scenarios causes an increase of discharge in winter (November–April) and a decrease during summer, but even the discharge values based on the extreme scenario A1B are still higher than the measured discharge.

Extreme values A comparison of the frequency distribution of discharge (Fig. 7) reveals the increase of the mean discharge values between 0.3 and 1.5 m^3/s. The mean discharge (Fig. 8) decreases in summer and increases in winter; the annual average values do not change significantly. The mean low water level in summer decreases by app. 30% for all scenarios and increases in winter between 40 (A2, A1B) and 70% (B1).

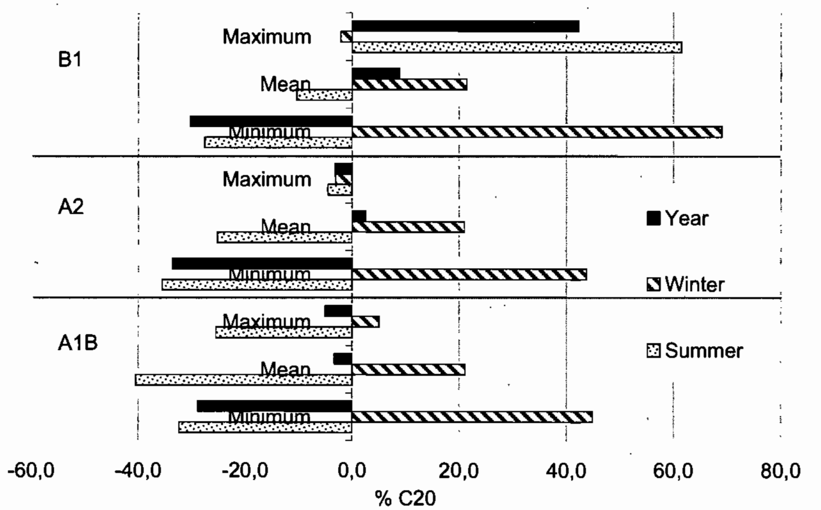

Fig. 7 Difference of annual maximum, mean and minimum discharge values between the C20 and the IPCC scenarios (mean values of the annual extreme values of the last 30 scenario years).

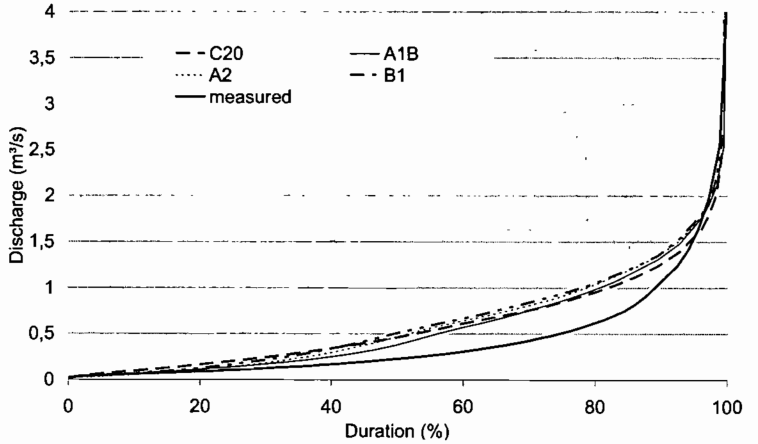

Fig. 8 Comparison of the observed and simulated flow frequency distribution (C20, A2, A1B, B1: IPCC scenarios of REMO climate data).

CONCLUSIONS

The REMO data base is a unique opportunity to get climate data with high resolution as input, but there are a few drawbacks, at least for Schleswig-Holstein which is situated between the North Sea and the Baltic Sea. The seasonal distribution of temperature and the absolute height of precipitation are not well met. The high precipitation in summer in the REMO simulations causes much higher discharge than observed in reality. Even the simulated values of the dry scenario are still higher than the measured values. It is possible to analyse the differences between the C20 and the IPCC scenarios, but the interpretation of these results remains difficult because they are still higher than measured values.

Nevertheless, the REMO data base is an extremely helpful database. It creates a common base for scenarios at a spatial scale suitable for regional analyses.

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6 Monitoring and Optimisation

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Monitoring groundwater variability from space: the GRACE satellite gravity mission

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Abstract NASA and the German Space Agency (DLR) launched the GRACE satellite mission in 2002. GRACE provides accurate monthly solutions for Earth's global gravity field. Differences between solutions provide information about monthly variations in Earth's mass distribution. The results can be used to monitor changes in the total volume of water stored in regions with scales of a few hundred kilometres and larger. Total water volume includes contributions from groundwater, soil moisture, surface water, and snow. If soil moisture and surface water+snow contributions are subtracted, using observations or model output, the residuals provide groundwater variability.

Key words groundwater monitoring; water storage; soil moisture; GRACE satellite; remote sensing; time-variable gravity

INTRODUCTION

Water resources can be managed more effectively if changes in the volume of stored water can be monitored. Land water storage in a large region is a difficult quantity to monitor using *in situ* methods. Groundwater is often inferred by monitoring water table depths in wells and converting to water volume using an assumed value of porosity. Soil moisture can be monitored using soil moisture probes. An inherent difficulty with these data types is that the distribution of available wells and soil moisture probes in a region is often far from satisfactory. Measurements must be interpolated between instruments and extrapolated to the edges of the region, a task complicated by the fact that water storage can vary significantly over small scales. For some terrestrial properties where *in situ* observations are subject to sampling problems such as these, remote sensing satellites can be used to obtain dense coverage over large regions. The water content in the upper few centimetres of soil, for example, can be mapped using satellite-based passive microwave measurements. Using satellites to monitor water beneath this thin surface veneer might seem improbable. But there is a satellite presently in orbit that is able to sense water variability at depth, and to recover spatial averages of that variability over large regions.

The GRACE (Gravity Recovery And Climate Experiment) satellite mission was launched in March 2002, and is projected to last through to 2013. GRACE, which consists of a pair of satellites 220 km apart in the same orbit, provides monthly, global gravity field solutions at scales of a few hundred kilometres and greater, in the form of spherical harmonic coefficients (Tapley *et al.*, 2004). Models are used to remove atmospheric and oceanic contributions from the satellite measurements before solving for the gravity coefficients. The coefficients can be used to estimate mass variability in the spatial domain. Here, we use coefficients truncated to degree 60, computed by the University of Texas Center for Space Research (CSR) to estimate monthly water storage variability, focusing on the United States of America to demonstrate the applicability of GRACE for groundwater recovery.

The gravity field results are equally sensitive to water at all depths: soil moisture, surface water+snow, and groundwater. For some applications (e.g. validation of land surface models, or combining with river discharge measurements to estimate precipitation minus evaporation), total water storage is of direct use. For other applications it can be useful to combine GRACE with independent estimates of specific water storage components. For example, monitoring groundwater variability requires that independent soil moisture and surface water+snow estimates, either from observations or from models, be subtracted from the GRACE total water results. Two studies illustrating this approach are presented here.

ANALYSIS METHODS

Global gravity fields are typically provided in the form of spherical harmonic (Stokes) coefficients. Let ΔC_{lm} and ΔS_{lm} represent the time-variability of the Stokes coefficients about their long-term means, and let $N(\theta,\phi)$ be the time-varying component of the geoid height at co-latitude θ and eastward longitude ϕ. It is usual to expand N as a sum of Legendre functions, P_{lm} (see e.g. Wahr *et al.*, 1998):

$$N(\theta,\phi) = a\sum_{l=2}^{\infty}\sum_{m=0}^{l} P_{lm}(\cos\theta)(\Delta C_{lm}\cos m\phi + \Delta S_{lm}\sin m\phi) \tag{1}$$

where a is Earth's radius. The horizontal scale of any term in (1) is inversely proportional to l.

Variability in the Stokes coefficients can be used to determine mass changes near Earth's surface. The largest such variations are caused by changes in the distribution of water, snow, and ice stored on the surface and in the ground. Let $\sigma(\theta,\phi)$ be the change in mass/area at (θ,ϕ), integrated vertically through the water column; σ is related to the Stokes coefficients as (Wahr *et al.*, 1998):

$$\sigma(\theta,\phi) = \frac{a\rho_{ave}}{3}\sum_{l=2}^{\infty}\sum_{m=0}^{l} \frac{2l+1}{1+k_l} P_{lm}(\cos\theta)(\Delta C_{lm}\cos m\phi + \Delta S_{lm}\sin m\phi) \tag{2}$$

where ρ_{ave} is Earth's average density, and the k_l are Love numbers representing perturbations in Earth's gravitational potential due to deformation caused by $\sigma(\theta,\phi)$.

Equation (2) is the starting point for recovering surface mass density from GRACE. Errors in the GRACE results increase as l becomes large (i.e. at short scales). Because large l terms have important contributions to (2), the use of (2) as written can lead to inaccurate results. To obtain more accurate estimates a multiplicative factor that is small for large values of l can be inserted into the sum. The result will then not be an exact representation of the surface mass at (θ,ϕ). But it is possible to choose multiplicative factors so that the sum still has a meaningful connection to the surface mass in a localized region. Such methods have been used for improving the GRACE mass estimates. These methods fall into one of two categories: smoothing the surface mass, or averaging over a specific region.

Smoothing

The simplest way of modifying (2) to obtain accurate results, is to introduce smoothing factors W_l into the sum, so that:

$$\bar{\sigma}(\theta,\phi) = \frac{a\rho_{ave}}{3}\sum_{l=2}^{\infty}\sum_{m=0}^{l} \frac{2l+1}{1+k_l} W_l P_{lm}(\cos\theta)(\Delta C_{lm}\cos m\phi + \Delta S_{lm}\sin m\phi) \tag{3}$$

$\bar{\sigma}$ represents a smoothed version of the surface mass anomaly, equivalent to:

$$\bar{\sigma}(\theta,\phi) = \int \sigma(\theta',\phi')W(\alpha)\sin\theta'\,\mathrm{d}\theta'\mathrm{d}\phi' \tag{4}$$

where α is the angle between (θ,ϕ) and (θ',ϕ'), and the W_l are Legendre coefficients of the smoothing function $W(\alpha)$. A common choice for $W(\alpha)$ is a Gaussian function (Wahr *et al.*, 1998), where the radius (the distance between the centre and half-amplitude of the Gaussian) is chosen to match the quality of the data and the spatial extent of the signal.

Regional averaging

Many applications require estimates of mass variability for a specific region, e.g. estimating water storage changes in a river basin. This problem is best addressed by constructing averaging functions optimized for that region. An exact basin average of the surface mass density has the form:

$$\sigma_{region} = \frac{1}{\Omega_{region}} \int \sigma(\theta,\phi)\eta(\theta,\phi)\sin\theta' \mathrm{d}\theta' \mathrm{d}\phi' \tag{5}$$

where the basin function $\eta(\theta,\phi)$ equals 1 when (θ,ϕ) is inside the basin and equals 0 when it is outside; and Ω_{region} is the angular area of the basin. In the harmonic domain, (5) is:

$$\sigma_{region} = \frac{a\rho_{ave}}{3\Omega_{region}} \sum_{l=2}^{\infty}\sum_{m=0}^{l} \frac{2l+1}{1+k_l}(W_{lm}^{c}\Delta C_{lm} + W_{lm}^{s}\Delta S_{lm}) \tag{6}$$

where W_{lm}^{c} and W_{lm}^{s} are the spherical harmonic expansion coefficients of $\eta(\theta,\phi)$.

GRACE results that use an exact regional average tend to be inaccurate unless the size of the region is large. For more accurate results, a smooth averaging function can be constructed that is nearly 1 inside the region and nearly 0 outside, but that varies smoothly between 0 and 1 along the edges (e.g. Swenson & Wahr, 2002). The expression for σ_{region} still has the form (6), but W_{lm}^{c} and W_{lm}^{s} now decrease more rapidly with increasing l.

APPLICATIONS TO THE UNITED STATES OF AMERICA

To illustrate the capabilities of GRACE we apply these methods to the USA using CSR's monthly solutions for April 2002–November 2008. We replace the GRACE C_{20} coefficients with coefficients determined from Satellite Laser Ranging (Cheng & Tapley, 2004); and we include degree-one coefficients determined from the other GRACE coefficients and an ocean model (Swenson *et al.*, 2008a). We remove post-glacial-rebound contributions computed by Paulson *et al.* (2007) for Peltier's (2004) ICE-5G ice deglaciation model and VM2 mantle viscosity profile. All Stokes coefficients are filtered to reduce correlated noise (Swenson & Wahr, 2006).

We simultaneously fit a constant, a trend, and annual and semi-annual terms to each Stokes coefficient. We use the annually varying components in equation (3), choosing the W_l to correspond to 250-km Gaussian smoothing, to compute the annual cycle in mass on an evenly-spaced grid. The amplitude of the North American annual cycle is shown in Fig. 1(a). The most prominent features are: (1) a large-amplitude signal running southeastward from Alaska through the northwestern USA, associated with seasonal snowpack; (2) large-amplitude signals in Central America, the lower Mississippi valley, and southeastern Canada, all corresponding to high precipitation regions; and (3) a low-amplitude band across the arid southwestern USA.

The mid-section of the USA, about 40% of the total USA land area, lies within the Mississippi River drainage basin, outlined in white in Fig. 1(a). Figure 1(b) shows the GRACE time series for total water storage in this drainage basin, computed using equation (6). The averaging coefficients, W_{lm}^{c} and W_{lm}^{s}, are for a smoothed version of the Mississippi basin function: a convolution of the basin function with a Gaussian filter of 250-km half width (equation (29) of Swenson & Wahr, 2002). Also shown is a similarly constructed time series for total water storage predicted by the GLDAS/Noah land surface model (Rodell *et al.*, 2004). The agreement between GRACE and GLDAS/Noah is excellent.

Groundwater estimates

GRACE is not able to separate groundwater from soil moisture and surface water+snow. But if the latter two components can be independently estimated, their contributions can be subtracted from the GRACE results to determine groundwater variability (e.g. Rodell & Famiglietti, 2002; Rodell *et al.*, 2007). The feasibility of this approach has been verified for two USA regions, centred on Illinois (Swenson *et al.*, 2006) and Oklahoma (Swenson *et al.*, 2008b), shown as plus signs in Fig. 1(a). Both Illinois and Oklahoma host a network of automated soil moisture sensors which can be used to model and remove soil moisture contributions from the GRACE results. Neither region experiences significant snowpack. Government agencies in both regions have also

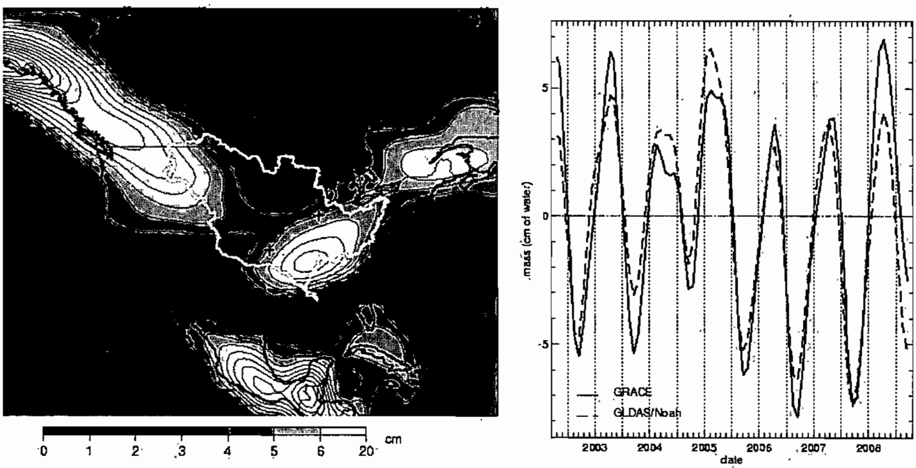

Fig. 1 (a) Amplitude of the annual cycle in mass, in cm of water thickness, inferred from GRACE. Results are smoothed with a 250-km Gaussian. The Mississippi River basin is outlined in white. Plus signs denote the centres of Illinois (northeastern plus sign) and Oklahoma (southwestern). (b) Total water storage variability in the Mississippi River basin. The black line shows monthly results from GRACE; the dashed line from the GLDAS/Noah land surface model (Rodell *et al.*, 2004).

established a network of wells used to monitor water table levels and so to estimate groundwater fluctuations. These well data can be used to help assess the GRACE groundwater results.

The difficulty in using *in situ* data to remove soil moisture from GRACE depends not only on the accuracy and spatial coverage of the soil moisture measurements, but also on the depth to the saturated zone and on how completely the soil moisture measurements sample that depth. The Illinois and Oklahoma studies represent different end-members of that problem.

Illinois The Illinois State Water Survey maintains a network of 19 automated soil moisture probes that record soil moisture in 11 layers through the top 2 metres of soil, and 16 wells used to monitor water table levels (Fig. 2(a)). For most of the wells the depth to the saturated zone is 2 m or less. Thus, we expect the soil moisture data to provide a reasonably complete record of total soil moisture above the saturated zone.

The GRACE Stokes coefficients are used in equation (3) with 300-km Gaussian smoothing coefficients (W_l) to recover a time series of total stored water centred on Illinois. The smoothing function (Fig. 2(a)) extends outside the region covered by the soil moisture and well level sites. But the most highly weighted GRACE region is sampled by the *in situ* data. The soil moisture data are averaged into monthly values, integrated vertically to obtain total-column soil moisture, and spatially smoothed using the GRACE smoothing function. The groundwater storage estimates are obtained by multiplying the well level data by a specific yield of 0.08, averaging into monthly means, and spatially smoothing. The resulting soil moisture and groundwater time series are shown in Fig. 2(b). Both time series are dominated by seasonal signals of about the same amplitude, though the groundwater tends to lag the soil moisture by 1–2 months. Figure 2(c) compares the sum of these soil moisture and groundwater signals with the GRACE results. The two results agree well in phase, with the *in situ* results having a slightly larger seasonal amplitude.

Oklahoma The analysis of the Oklahoma results is complicated by the fact that the soil moisture observations do not extend all the way down to the water table, which is typically deeper than 4 m. The GRACE Stokes coefficients are used in equation (3) with smoothing coefficients W_l for a 300-km Gaussian, to estimate total water storage variability centred on Oklahoma. Values of the smoothing function are shown in Fig. 3(a); it shows the Oklahoma Mesonetwork (OM) of soil moisture sites; each records soil moisture at depths of 5, 25, 60, and 75 cm. The GRACE smoothing function extends well outside the soil moisture network.

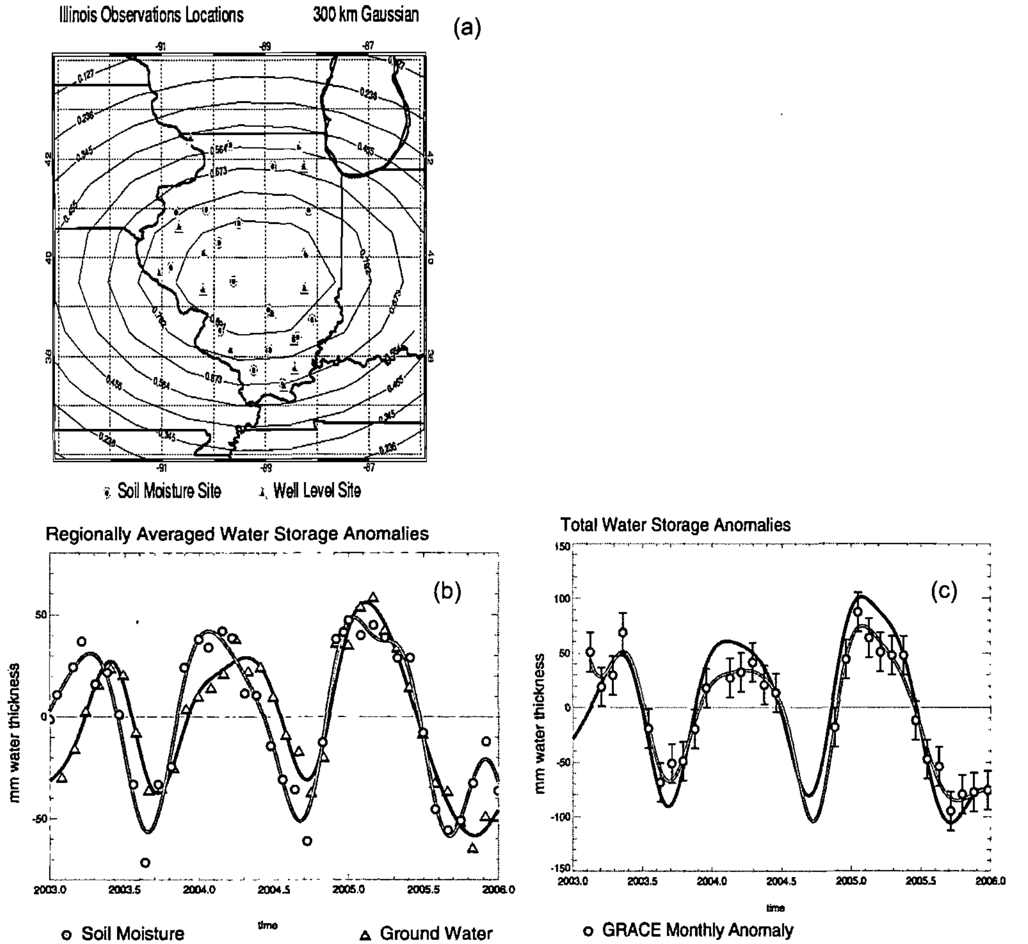

Fig. 2 (a) Contours denote the values of the GRACE smoothing function for Illinois. The symbols denote the locations of soil moisture probes and monitoring wells (from Swenson *et al.*, 2006). (b) Monthly values of groundwater and vertically integrated soil moisture, from the *in situ* Illinois data. (c) Comparison between GRACE total water storage, and the sum of the *in situ* soil moisture and groundwater components (dark line, without symbols). (From Swenson *et al.*, 2006).

The OM observations show that soil moisture variability across the network has about the same amplitude through the entire 5–75 cm depth range. Thus, there is no guarantee that these soil moisture results capture all the water storage variability above the saturated zone. Vertical extrapolation of the OM results is necessary before vertically integrating to obtain the total soil moisture component.

To devise a vertical extrapolation method, soil moisture measurements are used from the ARM (Atmospheric Radiation Measurement) network of soil moisture probes shown in Fig. 3(b). These probes sample soil moisture down to 1.75 m depth and so provide increased coverage with depth, though their spatial distribution is not dense enough to construct useful GRACE corrections by themselves. The ARM results show that significant soil moisture variability extends at least as deep as 1.75 m, and probably to well below that depth. The time variability tends to keep the same phase as the depth increases, but to decrease in amplitude. Exponentially decreasing functions of depth are fitted to the ARM measurements, and the results are extended down to 4 m depth, where all extrapolated amplitudes are nearly zero. Vertical integration of the ARM results, first over just the upper 75 cm and then over the entire 4-m column, shows that a good approximation to the full 4-m integration can be obtained by multiplying the 75-cm vertical integration results by 1.75. Consequently, the 75-cm monthly integrations of the original OM soil moisture results are multiplied by 1.75 to obtain a total soil moisture component.

Figure 3(c) shows the total water storage results from GRACE, and the OM-based soil moisture component. The phases agree well. Both are maximum in February/March and minimum in August/September. The GRACE amplitudes range from 1.2 to 1.7 times the soil moisture values. That difference presumably represents groundwater storage. Figure 3(d) compares the GRACE-minus-soil moisture time series with a rough estimate of groundwater variability inferred using water level measurements from wells scattered around the region (Fig. 3(b)), many well outside the GRACE footprint. This independent estimate is highly uncertain, not only because of the spatial sampling problems, but also because of uncertainties in the specific yield, required to convert water levels to water storage. Even so, the two groundwater time series are in general agreement, both dominated by seasonal cycles that are in phase with one another. The correlation coefficient between the series is 0.89 and the RMS difference is 9.1 mm.

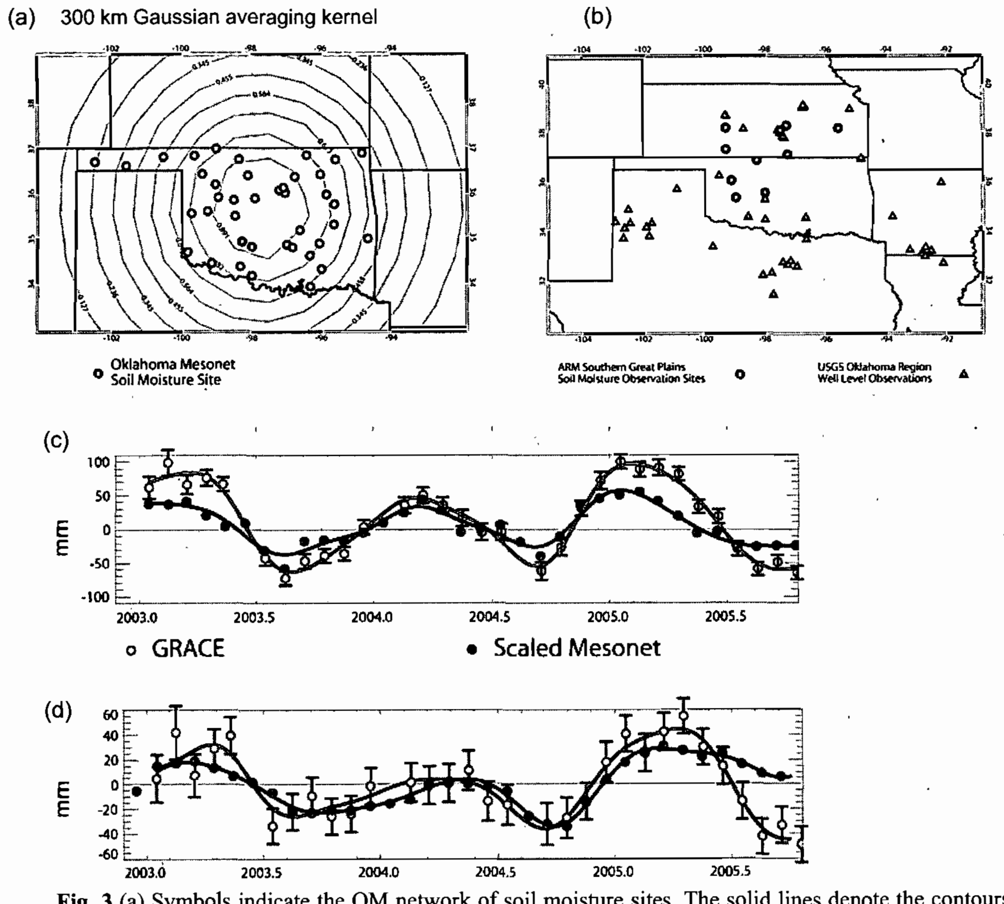

Fig. 3 (a) Symbols indicate the OM network of soil moisture sites. The solid lines denote the contours of the GRACE smoothing function for Oklahoma. (b) The locations of the soil moisture probes and monitoring wells of the ARM network (from Swenson *et al.*, 2008b). (c) GRACE results for total water storage, and OM estimate of soil moisture, for Oklahoma. (d) A comparison between the GRACE groundwater estimates (light circles with error bars) computed as GRACE-minus-OM soil moisture, and a groundwater estimate for the surrounding region from the ARM well levels (dark circles). (From Swenson *et al.*, 2008b.)

DISCUSSION

GRACE allows users to recover monthly changes in total water storage averaged over regions with scales of a few hundred kilometres and greater. To estimate groundwater from the GRACE results, estimates of vertically integrated soil moisture and surface water+snow must be subtracted from the GRACE results. However, with the exception of the Amazon River flood plain and a few very large lakes (e.g. the Great Lakes, Lake Victoria, etc.), there is rarely enough liquid water variability on the surface to significantly impact the GRACE results.

The Illinois and Oklahoma studies described above are optimal, in that both regions have soil moisture monitoring networks. In each case there is good agreement between the GRACE-minus-soil moisture results and *in situ* estimates of groundwater variability. GRACE-based groundwater estimates have the inherent advantage that GRACE is sensitive to all the groundwater within a region. There is no interpolation or extrapolation of point measurements. Knowledge of subsurface soil parameters such as specific yield is also unnecessary. The GRACE groundwater results do require soil moisture estimates. If those estimates are based on *in situ* measurements, then they are likely to require interpolation and extrapolation. Still, it is usually easier and cheaper to measure soil moisture than to dig wells.

There are other ways to estimate soil moisture. Moisture in the upper few centimetres of soil is monitored, for example, by AMSR-E on the AQUA satellite. If those results could be extrapolated vertically through the entire soil layer, it would be possible to combine them with GRACE to determine groundwater. There have, as yet, been no attempts to do this. Output from land surface models offers another means of estimating soil moisture. The Fig. 1(b) results for the Mississippi River basin suggest that state-of-the-art models are capable of predicting total water storage variability with some confidence, though whether those models accurately partition the water storage between soil moisture and groundwater cannot be inferred from comparisons such as these.

The combination of GRACE and land surface models can be particularly useful for estimating anthropogenic groundwater signals. Few operational models include anthropogenic sources, particularly at the scales sampled by GRACE. But those models do a credible job predicting total water storage changes caused by natural variability. By subtracting the modelled natural water storage from GRACE total water storage, it is possible to estimate anthropogenic contributions. In regions where there is significant groundwater extraction, long-term changes in anthropogenic water storage are likely to reflect groundwater depletion. This method has been applied to northern India, where non-anthropogenic water storage predicted by land surface models has been subtracted from GRACE to estimate the ongoing depletion rate of groundwater due to excessive groundwater extraction (Tiwari *et al.*, 2009). This method is applicable to any region with dimensions of a few hundred kilometres or larger, where there is likely to be significant groundwater depletion.

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Vegetation phenology to partition groundwater from surface water-irrigated areas using MODIS 250-m time series data for the Krishna River basin, India

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Abstract This paper describes a remote sensing based vegetation-phenology approach to accurately separate out and quantify groundwater irrigated areas from surface-water irrigated areas in the Krishna River basin (265 752 km^2), India, using MODIS 250-m every 8-day near continuous time series for 2000–2001. Temporal variations in the Normalized Difference Vegetation Index (NDVI) pattern, depicting phenology, obtained for the irrigated classes enabled demarcation between: (a) irrigated surface-water double crop, (b) irrigated surface-water continuous crop, and (c) irrigated groundwater mixed crops. The NDVI patterns were found to be more consistent in areas irrigated with groundwater due to the continuity of water supply. Surface water availability, however, was dependent on canal water release that affected time of crop sowing and growth stages, which was in turn reflected in the NDVI pattern. Double-cropped (IDBL) and light irrigation (IL) have relatively late onset of greenness, because they use canal water from reservoirs that drain large catchments and take weeks to fill. Minor irrigation and groundwater-irrigated areas have early onset of greenness because they drain smaller catchments where aquifers and reservoirs fill more quickly. Vegetation phonologies of nine distinct classes consisting of irrigated, rainfed, and other land-use classes were derived using MODIS 250-m near continuous time-series data that were tested and verified using groundtruth data, Google Earth very high resolution (sub-metre to 4 m) imagery, and state-level census data. Fuzzy classification accuracies for most classes were around 80% with class mixing mainly between various irrigated classes. The areas estimated from MODIS were highly correlated with census data (R-squared value of 0.86).

Key words groundwater irrigated areas; surface-water irrigated areas; phenology; MODIS; NDVI; irrigated areas; Krishna basin, India

INTRODUCTION

In the Krishna River basin, the fourth largest basin in India (265 752 km^2), changes in the irrigated areas are frequent. Major canal irrigation schemes often suffer from inequitable distribution of water due to over-use in head reaches which is partly caused by farmer preferences to water intensive crops like rice and sugarcane (Bhutta & Velde, 1992; Gaur *et al.*, 2008). However, farmers often supplement with and/or exclusively use groundwater irrigation.

Groundwater irrigation, in particular, has become increasingly common globally. In India, the total groundwater irrigated area even exceeds the surface-water irrigated area (Velpuri *et al.*, 2009; Shah *et al.*, 2000). Most groundwater-irrigated plots are small (<1 ha), which complicates use of traditional satellite image classification techniques (Biggs *et al.*, 2006). However, quite often these small plots adjoin each other, one after the other, with each plot having a well providing contiguity. Nevertheless, identification of training sites for supervised classifications is particularly problematic in areas with patchy irrigated systems, where plot sizes are often small relative to satellite pixels.

Due to multi-cropping and varying cropping calendars, single image snap-shots often do not adequately characterize irrigated areas (Sellers & Schimal, 1993). Many crops have relatively short and staggered growth, development, and senescent phases, making accurate mapping difficult using satellite images from a single or even a series of overpasses. Satellite data with short return intervals, such as daily Moderate Resolution Imaging Spectrometer (MODIS) imagery and their 8-day processed cloud-free composites, e.g. MOD09Q1, data products (King *et al.*, 2003), have problems distinguishing the full variety of cropping patterns and irrigation intensities (including groundwater and surface water).

This paper presents maps of land cover and irrigation in a heterogeneous landscape using an irrigated fraction approach. The method fuses and compares multiple data sources, including a time series of MODIS imagery, groundtruth data, and agricultural census data. We first generate generalized classes of land cover using unsupervised classification of a time series of MODIS NDVI images. We then estimate irrigated fractions and total irrigated area for each remote sensing class using groundtruth data and agricultural census data. Accuracy assessment is performed by comparing the irrigated fraction statistics determined by each data source, and by fuzzy accuracy assessment based on groundtruth data (Thenkabail *et al.*, 2005). The main innovations of the method include use of NDVI time series to differentiate areas irrigated by surface water and groundwater, and the use of multiple data sources for aggregate accuracy assessment.

STUDY AREA

The Krishna basin (Fig. 1) is India's fourth largest river basin and covers 265 752 km^2 of southern India, traversing the states of Karnataka (116 247 km^2), Andhra Pradesh (78 256 km^2) and Maharashtra (71 249 km^2). The basin is relatively flat, except for the Western Ghats and some forested hills in the centre and northeast.

The River Krishna originates in the Western Ghat mountains, flows east across the Deccan Plateau, and discharges into the Bay of Bengal. It has three main tributaries that drain from the northwest, west and southwest (Fig. 1). The climate is semi-arid, with some dry, sub-humid areas in the eastern delta and humid areas in the Western Ghats. Annual precipitation averages 780 mm and decreases gradually from 850–1000 mm in the Krishna Delta to 300–400 mm in the northwest, then increases to >1000 mm in the Western Ghats (Fig. 1). In the extreme western parts of the basin, the Western Ghats have high annual precipitation (1500–2500 mm). Most of the rainfall occurs during the monsoon from June to October.

DATA

The MODIS data for the Krishna River basin were downloaded from calibrated global continuous time series mega data sets (see http://www.iwmidsp.org) composed from the individual files from

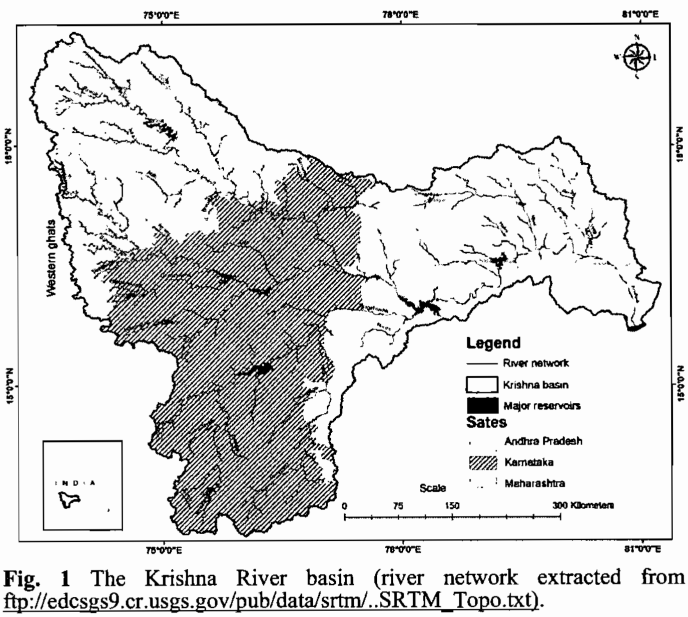

Fig. 1 The Krishna River basin (river network extracted from the SRTM 90-m DEM, ftp://edcsgs9.cr.usgs.gov/pub/data/srtm/..SRTM_Topo.txt).

Table 1 MODIS Terra[2]-band reflectance data characteristics used in this study[1].

Band	Band width (nm)[3]	Band centre (nm)[3]	Potential application[4]
1	620–670	648	Absolute land cover transformation, vegetation chlorophyll
2	841–876	858	Vegetation land cover transformation

1 Of the 36 MODIS bands, the seven bands reported here are specially processed for land studies.
2 MODIS bands are re-arranged to follow the electromagnetic spectrum (e.g. blue band 3 followed by green band 4).
3 Nanometres.
4 Taken from MODIS web site (http://modis-land.gsfc.nasa.gov/)

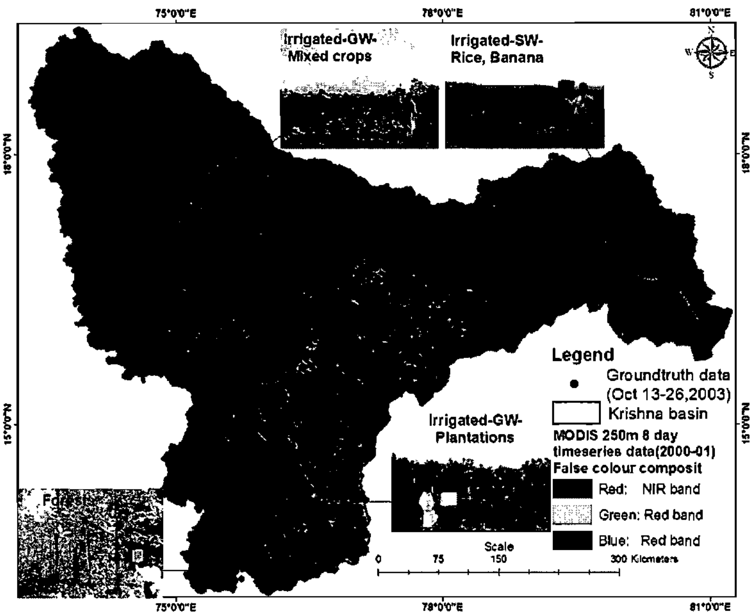

Fig. 2 Spatial distribution of groundtruth data points in the Krishna River basin.

the NASA website (http://www.modis.land.gsfc.nasa.gov). MODIS 2001 every 8-day (Table 1) Terra sensor data in two specific bands: Band 2 (near infrared), and Band 1 (red)) are processed for land applications as a MODIS surface reflectance product (MOD09Q1). The MOD09Q1 is computed from MODIS level 3 bands 1–2 (centred at 648 and 858 nm). The product is an estimate of the surface reflectance for each band as it would have been measured at ground level if there were no atmospheric scattering or absorption. Original MODIS data is acquired in 12-bit (0 to 4096 levels), and stretched to 16-bit (0 to 65536 levels).

Groundtruth data were collected during 13–26 October 2003 for 144 sample sites covering about 6500 km of road travel in the Krishna River basin (Fig. 2). The Geocover 2000, SOI-toposheets were also used as additional groundtruth information in class identification.

Point specific data was collected from 90 m × 90 m plots and consisted of GPS locations, land-use categories, land cover percentages, cropping pattern during different seasons (through farmer interviews), crop types, and watering method (irrigated, rainfed). Samples were obtained within large contiguous areas of a particular land use/land cover (LULC). A stratified-systematic sample design was adopted. The framework was stratified by motorable road network or footpath access where possible, made systematic by locating sites every 5 or 10 km along the road network by vehicle or on foot (see Thenkabail *et al.*, 2004a, 2005) for detailed description on the ground-truth methodological approaches).

METHODS

MODIS NDVI time series classification

A time series of MODIS 8-day composite reflectance images at 250-m resolution, was obtained for 1 June 2000 to 31 May 2001 (MOD09Q1 data product). The 8-day composite images in the MOD09A1 data set are free of cost and pre-calibrated (King *et al.*, 2003; http://modis-sr.1tdri.org/html). The large scene size and daily overpass rate of MODIS makes it attractive for large area crop mapping, and NDVI derived from MODIS has high fidelity with biophysical parameters (Huete *et al.*, 2002). The composites are created using the maximum NDVI method on the daily MODIS data to minimize cloud effects (Holben, 1986). The 8-day composite images were downloaded from June 2005 to July 2006. There were three to four 8-day composites per month for a total of 45 8-day composites. The 8-day NDVI were stacked into a 46-band NDVI mega-file image (MFI).

Unsupervised classification followed by progressive generalization (Cihlar *et al.*, 1998) was used to classify the MODIS time series and generate generalized classes. The unsupervised ISOCLASS cluster algorithm (ISODATA in ERDAS Imagine 9.2™) run on the MFI generated an initial 40 classes, with a maximum of 40 iterations and convergence threshold of 0.99. Though groundtruth data were available at the time of image classification, unsupervised classification was used in order to capture the full range of NDVI time series over a large area. Use of unsupervised techniques is recommended for large areas that cover a wide and unknown range of vegetation types, and where landscape heterogeneity complicates identification of homogeneous training sites (Achard *et al.*, 1995; Cihlar, 2000; Biggs *et al.*, 2006). Identification of training sites is particularly problematic for small, heterogeneous irrigated areas.

The original 40 classes from the unsupervised classification were merged by user-controlled progressive generalization (Cihlar *et al.*, 1998) using the class-average MODIS NDVI time series, groundtruth data (described below), and GeoCover mosaics of Landsat imagery from 1990 and 2000 (Tucker *et al.*, 2004; Thenkabail *et al.*, 2005). Classes with similar NDVI time series and land cover were merged into a single class, and classes showing significant mixing, e.g. continuous irrigated areas and forest, were masked and reclassified using the same ISOCLASS algorithm. Some continuous irrigated areas mixed with forests in the Western Ghats were separated using a 90-m digital elevation model (DEM) from the Shuttle Radar Topography mission (SRTM) and an elevation threshold of 630 m, determined using the DEM, Landsat imagery and groundtruth data. The merging procedure, also known as progressive generalization (Cihlar *et al.*, 1998) resulted in a 40-class map and a 9-class map.

Class signatures and NDVI-reflectivity thresholds

The class signatures of NDVI are unique spectral properties of a class that can be mapped using NDVI time series of a class. It is not possible to have "spectral signatures" when single date or a few date images are used as is often the case with most LULC studies. Since near continuous MODIS data has been used in this study, a unique set of LULC class signatures were possible (e.g. Fig. 3).

The threshold NDVIs and NDVI signatures over time help us determine the land-use type including forests, surface irrigation areas, groundwater irrigation areas, rainfed, and rangelands:

1. onset of a cropping seasons (e.g. Rabi and khariff);
2. duration of the cropping seasons such as for khariff and Rabi;
3. magnitude of the crops during different seasons and years (e.g. drought *vs* normal years);
4. end of cropping season (senescence).

In order to illustrate these possibilities, the MODIS CS-NDVI signatures are presented and discussed for a set of distinct classes (Fig. 3) and thematically similar classes (Fig. 4). The NDVI of forest class 39 and 40 never falls below 0.5 on any date throughout a year and across years (Fig. 4), and clearly separates the surface irrigated areas and groundwater irrigated areas as seen in Fig. 4.

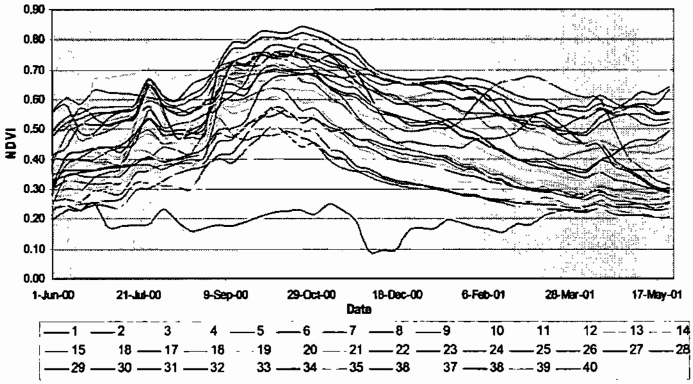

Fig. 3 Class spectral signatures for unsupervised 40 classes (MODIS time series data).

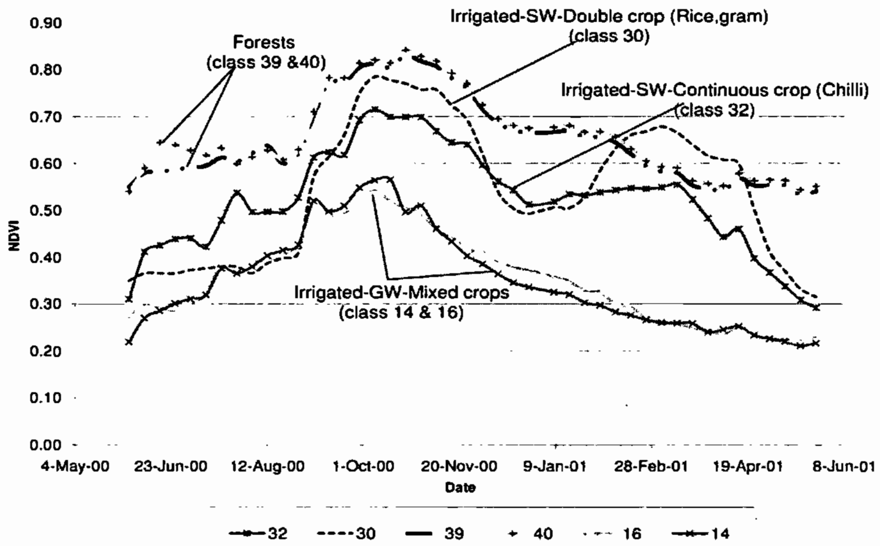

Fig. 4 MODIS NDVI signatures for six spectrally-close classes (MODIS time series data).

Irrigated fractions

The MODIS NDVI pixels are 250 m × 250 m and larger than many irrigated plots. Due to this sub-pixel heterogeneity, each class in the unsupervised classification has both a name and an irrigated fraction. The area irrigated by surface or groundwater for a given area of interest is computed from the MODIS classification as:

$$I_k = \sum_{i=1}^{n} \alpha_{i,k} A_i \qquad (1)$$

where I_k is the net area irrigated by source k (surface or groundwater), i is the MODIS class number, n is the number of MODIS classes, $\alpha_{i,k}$ is the fraction of class i irrigated by source k, and A_i is the area covered by MODIS class i. This definition is for net irrigated area, which includes all areas irrigated at some point during the annual cropping cycle, and counts double-cropped areas once. The irrigated fractions for each class ($\alpha_{i,k}$) were calculated using the average of the ground-truth estimates (Table 2).

Table 2 Land use/Land cover area fraction, 2000–2001.

LULC	Area (km^2)	N	Vegetation cover percent (mean): Tree	Shrubs	Grass	Others	Open	Crop	Major crops
Water bodies	5 176.8								
Shrub lands mix with rangelands	65 223.6	15	4.3	4.7	6.9	10.2	8.5	39.3	Grains, oilseeds
Rangelands mix with rainfed	10 441.3	33	0.7	1.0	2.0	1.8	0.0	42.2	Grains, oilseeds, pulses
Rainfed agriculture	59 122.4	17	4.8	5.0	9.9	8.7	4.8	66.8	Rice, grains, oilseeds, pulses
Rainfed + groundwater	30 146.1	25	2.1	1.3	3.4	7.1	10.8	75.6	Rice, oilseeds, pulses, grains, cotton, chilli
Minor irrigated (light/tank)	21 208.2	6	1.5	1.1	2.9	6.3	3.7	84.5	Cotton, grains, oilseeds, rice
Irrigated, conjunctive	27 187.4	10	6.7	2.0	1.7	11.3	2.8	88.9	Sugarcane, fodder grass, chilli, cotton
Irrigated, double crop rice, gram.	24 884.7	22	1.7	3.2	1.6	2.8	1.8	87.6	Rice, grains, pulses
Forests	22 361.4	12	60.2	12.2	3.0	2.7	10.2	21.3	Teak, coffee, aracanut, rice
Basin total	265 752.0	140	10.2	3.8	3.9	6.4	5.3	63.3	

Table 3 Irrigated, rainfed, and other land use/ land cover (LULC) areas.

LULC	%	Land use/ land cover area within the classes (km^2): Water	Tree	Shrubs	Grass	Others	Crop	Basin totals
Class1: Water bodies	1.9	5177.82	0.00	0.00	0.00	0.00	0.00	5 177.82
Class2: Shrublands mix with rangelands	24.5	0.00	4 370.83	15 852.40	4 488.25	14 873.85	25 631.04	65 216.37
Class3: Rangelands mix with rainfed	3.9	0.00	69.62	104.43	2 297.53	3 571.62	4 404.67	10 447.88
Class4: Rainfed agriculture	22.2	0.00	2 826.60	2 933.04	5 830.59	8 014.60	39 501.37	59 106.20
Class5: Rainfed + groundwater	11.3	0.00	626.23	388.50	1 020.53	5 315.05	22 788.84	30 139.15
Class6: Minor irrigated (light/tank)	8.0	0.00	327.83	231.41	617.09	2 121.23	17 924.41	21 221.96
Class7: Irrigated, conjunctive	10.2	0.00	740.24	547.63	453.21	1 282.59	24 182.39	27 206.06
Class8: Irrigated, double crop rice, gram	9.4	0.00	423.12	920.91	472.90	1 258.16	21 803.18	24 878.27
Class9: Forests	8.4	0.00	13 464.18	2 497.51	670.97	954.27	4 771.36	22 358.30
Basin-totals	100.0	5177.82	22 848.65	23 475.82	15 851.07	37 391.38	161 007.26	265 752.00

RESULTS AND DISCUSSION

LULC fractions

Land-use/land cover fractions calculated from groundtruth data points fall into classes (Thenkabail *et al.*, 2005; Gumma *et al.*, 2009). Each LULC class is combination of several land cover types (Table 2). For example in Table 2, in Class 6, cultivable areas (84.4%) dominate but there are other land cover types including 1.5% trees, 1.1%, shrubs, 2.9% grass, 3.7% others which include fallow, weeds, rock and built-up areas (Table 2). In these cultivable areas, cotton was the dominant crop, whilst rice and grains were the next most dominant crops. The areal estimate of various thematic joint classes was calculated as follows (see Table 2):

Cultivable land in Class 6 = LULC class area for class × LU % of cultivable lands
= 21 208 × (84.4/100) = 17 900 km^2

Using the same approach, there were 86 653 km^2 net of irrigated areas in 2000–2001 which includes surface water and groundwater irrigation in the basin.

Land-use/land cover maps and area statistics

The process of LULC classification, identification and labelling for river basins using MODIS time series is described in detail in Thenkabail *et al.* (2005) and Gumma (2009). This process led to nine distinct classes (Fig. 5(a)–(d)) which are spectrally well separated in MODIS data (Fig. 6).

Classes were identified based on groundtruth data including GPS referenced digital images and field observations. The LULC percentages in the Krishna River basin for 2000–2001 (Table 3, Fig. 5(a)–(d)) are: water bodies, 1.9% (Fig. 5(a)) of the total basin; shrub land mixed with range land, fallow, 28.4% (Fig. 5(b)); rainfed agriculture, 22.2%; rainfed+groundwater irrigation, 11.3% (Fig. 5(c)); minor irrigation which includes the tank and small reservoir classes, 8%; and surface irrigation by canal, 19.6% (Fig. 5(d)); and forest, 8.4% (Fig. 5(b)). By using spectral signatures (Fig. 6), this study identified major changes in groundwater-irrigated areas and rainfed areas in the major command areas, demonstrating the usefulness of the spectral matching technique. The results show that there is a marginal decrease in the total irrigated area of the Krishna basin which is 86 653 km^2 (groundwater-irrigated area is 22 784 km^2 and surface-water irrigated area is 63 868 km^2; Fig. 5(d)).

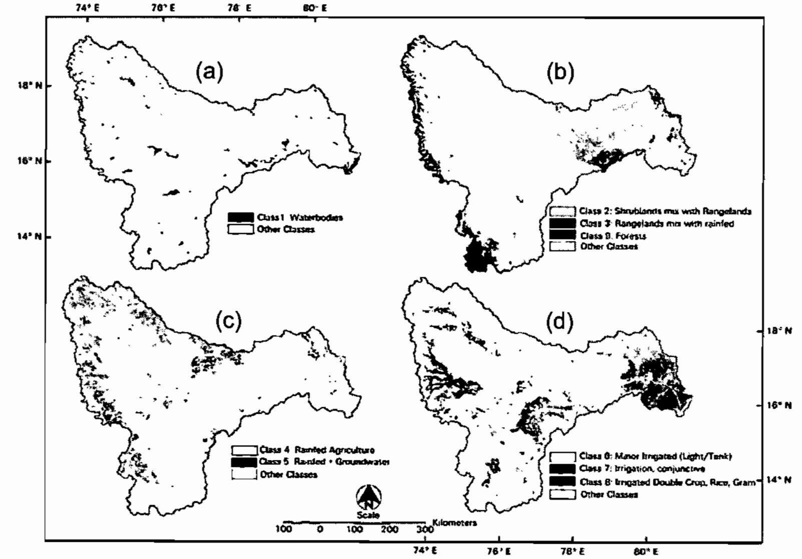

Fig. 5 Irrigated and other land-use classes of the Krishna basin (India) using MODIS 2000–01 data. The spatial distribution of the nine classes is shown: (a) water bodies, (b) natural vegetation, (c) rainfed croplands, and (d) irrigated areas.

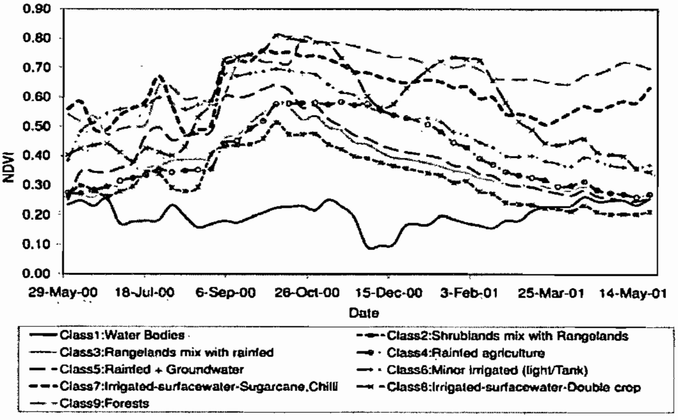

Fig. 6 MODIS NDVI signatures over time (for classes in Fig. 6) for the irrigated, rainfed, and other LULC classes.

Accuracy assessment

A qualitative accuracy assessment was performed to check if the irrigated area is classified as irrigated or not, without checking for crop type or type of irrigation. The accuracy assessment was performed using groundtruth data, to derive robust understanding of the accuracies of the data sets used in this study. The groundtruth data were based on an extensive field campaign conducted throughout the Krishna basin during the kharif season by International Water Management Institute researchers and consisted of 144 points.

Accuracy assessment provides realistic class accuracies (see equations below) where land cover is heterogeneous and pixel sizes exceed the size of uniform land cover units (see Gopal *et al.*, 1994; Thenkabail *et al.*, 2005; Biggs *et al.*, 2006). For this study we assigned 3 × 3 cells of MODIS pixels around each of the groundtruth points to one of six categories: absolutely correct (100% correct), largely correct (75% or more correct), correct (50% or more correct), incorrect (50% or more incorrect), mostly incorrect (75% or more incorrect), and absolutely incorrect (100% incorrect). Class areas were tabulated for a 3 × 3-pixel (9 pixels) window around each groundtruth point. If nine out of nine MODIS classes matched with the groundtruth data, then it was labelled absolutely correct, and so on (Table 4).

The accuracy assessment was carried out using:

$$\text{Accuracy of irrigated area class} = \frac{\text{Groundtruthed irrigated points classified as irrigated area}}{\text{Total number of groundtruthed points of irrigated area class}} \times 100$$

$$\text{Errors of commission for irrigated area class} = \frac{\text{Non-irrigated groundtruth points classified as irrigated area}}{\text{Total number of non-irrigated groundtruth points}} \times 100$$

$$\text{Errors of ommission for irrigated area class} = \frac{\text{Irrigated groundtruth points falling on non-irrigated area class}}{\text{Total number of groundtruthed points of irrigated area class}} \times 100$$

The accuracies and errors of the map of LULC are assessed based on intensive groundtruth data (Table 4). First, the 144 groundtruth data points reserved for accuracy assessment from Krishna basin field campaigns were pooled, and the accuracy was assessed. The accuracy of the rainfed croplands varied between 59% and 61%. However, the errors of omission were 2–8% and of commission 19–36%. The pooled data from the two sources provided a rainfed cropland accuracy of 94% with errors of omission of 5% and errors of commission of 27%.

Table 4 Fuzzy accuracy assessment using groundtruth data. Numbers in parentheses indicate the fuzzy correctness percentage. Values in the table indicate the percent of groundtruth windows in each class with a given correctness percentage.

MODIS LULC class	Sample size	Total correct	Total incorrect	Fuzzy classification accuracy Absolutely correct	Mostly correct	Correct	Incorrect	Mostly incorrect	Absolutely incorrect
		(%)	(%)	100 % correct	≥75% correct	≥51% correct	≥51 % incorrect	≥75% correct	100% incorrect
Water bodies	0	100	0	100	0	0	0	0	0
Shrublands mix with rangelands	15	81	19	47	10	23	19	0	0
Rangelands mix with rainfed	33	76	24	15	31	30	20	3	1
Rainfed agriculture	17	59	41	45	9	5	3	15	23
Rainfed + groundwater	25	63	37	41	5	17	18	1	18
Minor irrigated (light/Tank)	6	80	20	0	62	18	17	0	4
Irrigated, conjunctive	10	68	32	46	8	14	14	0	18
Irrigated, double crop	22	87	13	64	16	7	4	5	5
Forests	12	87	13	86	1	0	0	0	13
Total	140	78	22	49	14	12	12	3	9

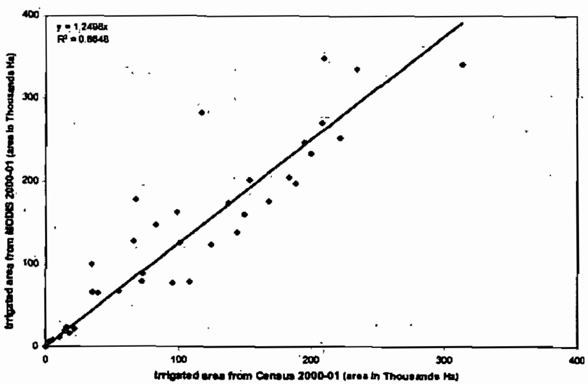

Fig. 7 District-wise irrigated area from the MODIS classification compared with the agricultural census.

Comparisons with census data

The LULC area statistics of the Krishna basin districts were obtained from the Bureau of Economics and Statistics, Andhra Pradesh, Directorate of Economics and Statistics, Karnataka and Department of Agriculture, Maharashtra. The data were obtained at district level from the respective states, and were fractionalized based on the district-wise area covered in the Krishna basin, for comparison with the MODIS data. The fractionized statistics data were compared with the MODIS data for the year 2001. Most of the districts data match with the MODIS data for the year 2001; few districts have difference in the statistical data and MODIS data varying between –30% and +30% (Fig. 7).

Vegetation phenology of groundwater and surface-water irrigated areas

Groundwater and surface-water irrigation show good separation in the classification (Fig. 6), and the district-wise irrigated area from the classification compares well with district-level census data (Fig. 7). Two properties of the NDVI time series allow separation of groundwater and surface-water irrigation: annual average NDVI, which is a function of the irrigated fraction, and timing on onset of greenness, which is a function of the timing of water availability for vegetation. Annual NDVI in both continuous and double irrigated systems exceeds annual NDVI in groundwater systems, reflecting the higher irrigated fraction in areas irrigated with surface water. Double cropped (IDBL) and light irrigation (IL) have relatively late onset of greenness, because they use canal water from reservoirs that drain large catchments and take weeks to fill. Minor irrigation and groundwater irrigated areas have early onset of greenness because they drain smaller catchments where aquifers and reservoirs fill more quickly (Fig. 6). This may not be the case in all years, depending on the relative timing of reservoir filling. In 2001, for instance, the onset of greenness in groundwater and surface-water systems was similar, which might make discrimination of the two more difficult.

Groundwater and surface-water irrigated areas also tend to be spatially segregated, which enhances their separability. Groundwater irrigation occurs along valley bottoms of second- and third-order streams or below small reservoirs, while surface irrigation occurs below larger reservoirs draining large catchment areas. Though conjunctive use of both surface water and groundwater may occur in some areas, such as in minor schemes near small reservoirs, in sugarcane irrigated areas during the dry season, or at the tail end of canals in surface-water command areas, clear separation of the two sources in major canal command areas is difficult, even in the field.

CONCLUSIONS

The paper presents a vegetation phenological approach, derived using time series MODIS 250-m data, in separating groundwater irrigation from surface-water irrigation based on a study conducted in a large river basin (Krishna, India). Annual average NDVI and timing of onset of greenness allowed the separation of groundwater from surface water. The specific conditions separated were: (a) double cropped (IDBL) and light irrigation (IL) have relatively late onset of greenness, because they use canal water from reservoirs that drain large catchments and take weeks to fill, and (b) minor irrigation and groundwater-irrigated areas have early onset of greenness because they drain smaller catchments where aquifers and reservoirs fill more quickly. The time series NDVI phenological signatures were distinctly different in the Krishna basin for: (a) irrigated surface-water double crop, (b) irrigated surface-water continuous crop, and (c) irrigated groundwater mixed crops. Of the basin area of 26 575 200 ha, the percentage distributions of various classes were (see spatial distribution of areas in Fig. 5(a)–(d)): water bodies (1.9%), shrub lands mixed with rangeland fallow (28.4%), rainfed agriculture (22.2%), groundwater dominant irrigated areas (11.3%), minor irrigation including tank and small reservoirs (8%), and surface irrigation by canal (19.6%), and forests (8.4%). The total minor irrigation (small reservoirs, tanks, groundwater) was 19.3% of the basin area and is about the same as the major irrigation from surface water (19.6%)

of the basin area. The total irrigated area was 38.9% (10 073 077 ha) of the basin area. However the groundwater irrigation is at times mixed with rainfed irrigation bringing its area slightly lower than 11.3%. So, the overall irrigated area from the MODIS 250-m time series reported here is close to that reported by Velpuri *et al.* (2009) for the Krishna basin, which was 9 356 160 ha, derived using Landsat 30-m data in combination with MODIS 500-m data.

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New perspectives in monitoring water resources in large tropical transboundary basins based on satellite imagery and radar altimetry

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Abstract The combined use of satellite imagery and radar altimetry offers entirely new perspectives for the monitoring of water resources in large tropical transboundary basins. We illustrate this point of view with results from a study conducted mostly in the region of the Llanos de Mojos, a large complex of wetlands located within the southernmost extension of the Amazon Basin, at the Brazilian border with Bolivia and Peru, and also from previous studies. First, despite the current limitations of the radar altimetry missions, which were designed primarily for ocean level or ice cap studies (essentially the revisit time, the size of the water bodies that can be monitored, and the lack of reliable data in the presence of relief), the data processing and the tools we developed to select the data appropriately, allow us to retrieve quite accurately the seasonal variability of the water elevation within the selected basin. For instance, the common altitudinal reference of the radar altimetry missions is offering new modelling opportunities, as the river slope is a key parameter for hydrodynamic studies. Second, the results emphasize the benefit of coupling these data with remote sensing images, to obtain information on surface water storage in this very complex system. Lastly, the spatial distribution that can be obtained nowadays, and the perspectives offered by future sensors, are moving towards a detailed global capability for monitoring wetlands and flood plains, as well as their relationship with the river flow. The application of these monitoring tools is of primary importance for tropical poorly-gauged basins in terms of infrastructure monitoring and planning, flood and drought monitoring and forecasting, fluvial waterway monitoring and transport planning, and the fluvial dynamics of the riverbed and discharge modelling.

Key words water resources; remote sensing; radar altimetry; transboundary basins; Amazon Basin

INTRODUCTION

Radar altimetry has recently demonstrated a strong potential for hydrological studies. A review of the different applications can be found in Calmant *et al.* (2008). We summarize the major ones:

surface water resource monitoring in relation to climate and agriculture. Many studies have been conducted over inland areas, mostly over lakes (see Creteaux & Birkett, 2006, for a review). The previous studies mostly used the water levels derived from the measurements of the 10-day repeat cycle of the CNES/NASA missions: T/P (1992–2005), followed by Jason-1 since 2002, and now Jason-2 since July 2008. Only a few studies were based on water levels derived from ESA altimetry missions (i.e. the ERS series started in 1991, and now ENVISAT, both with a 35-day repeat period) over lakes (Medina *et al.*, 2008) and rivers (Berry *et al.*, 2005; Frappart *et al.*, 2006a,b).

(b) sampling of all kind of water bodies for estimating the spatio-temporal variations of surface water volume over the main stream jointly with the flood plains in the Negro River basin (Frappart *et al.*, 2005, 2008) and the Mekong basin (Frappart *et al.*, 2006b), examining the relationship between river and flood plain through the differences in water levels was conducted by Cauhope (2004), and modelling the transfer of water between river and flood plain partly based on altimetric water levels time series (Bonnet *et al.*, 2008).
(c) Estimates of river slope and implications for hydrodynamics with both T/P and ENVISAT data Leon *et al.* (2006a,b).

The present study presents an application of radar altimetry for estimating slope and bankfull discharge.

DATA AND METHODS

The study area is located at the Brazilian border with Bolivia and Peru. In Brazil, this region is known as the upper Madeira region. The Madeira River has four main tributaries, two flowing through Bolivia: Beni and Mamore rivers; one flowing through Peru and Bolivia: the Madre de Dios River; and one forming the border between Bolivia and Brasil: the Guapore River. The Mamore and Beni are meandering rivers, flowing mostly from south to north, west of an extensive inundation plain, called Llanos de Mojos. The Llanos de Mojos region is a large flood plain of variable extent related to the alternating dry and rainy tropical seasons (Ronchail *et al.*, 2005); it is partly dry during the Austral winter and reaches 150 000 km^2 at the end of the rainy season (Roche & Fernandez, 1988).

Radar altimetry data

The altimetry data used in this study are the along-track measurements from the ENVISAT mission, made available by the CTOH, specifically the range data retracked by the ICE 1 algorithm. In order to extract the value of water surface elevation, we used a manual selection of the altimetry ranges projected in the plane perpendicular to the flow direction (Roux *et al.*, 2009). Virtual gauges are defined at the intersection between a water body and the satellite ground track. Each cycle, the water level is obtained by computing the median of all the data included in the selection window and the associated L1 norm dispersion. This process, repeated each cycle, allows the construction of the time series of water level associated with a virtual station (more details can be found in Frappart *et al.*, 2006a; Santos Da Silva *et al.*, 2008). Water levels are referenced to geoid EIGEN-GRACE02C, complete to order 200 (Tapley *et al.*, 2005).

For the purpose of validation, the data at six virtual stations have been compared with the water levels at six conventional gauges located on the Madeira and the Guapore rivers; gauges Porto Velho, Principe da Beira, Pimenteiras, Abuna, Pedras Negras, Vila Bela de Santissima Trindade, data distributed by ANA, http://www.ana.gov.br. Figure 1 presents the location of the 31 virtual gauges included in this study.

Satellite image data

The satellite images used were the JERS-1 (L Band SAR launched by the National Space Development Agency of Japan, NASDA, in February 1992) images mosaic from the Global Rain Forest Mapping (GRFM) project at 100-m resolution (Siqueira *et al.*, 2000).

Geomorphological parameters and discharge estimation

Three main geomorphological parameters have been estimated at the virtual stations, and are presented in Table 2. Each virtual station is identified by a sequential number, the satellite track number, the river it crosses, and the latitude-longitude of the virtual station's centre given as the mean of the longitude and latitude for the points constituting the station. The width (*L*) was measured from the JERS-1 images mosaic by the distance measuring tool of ARCGIS, and is

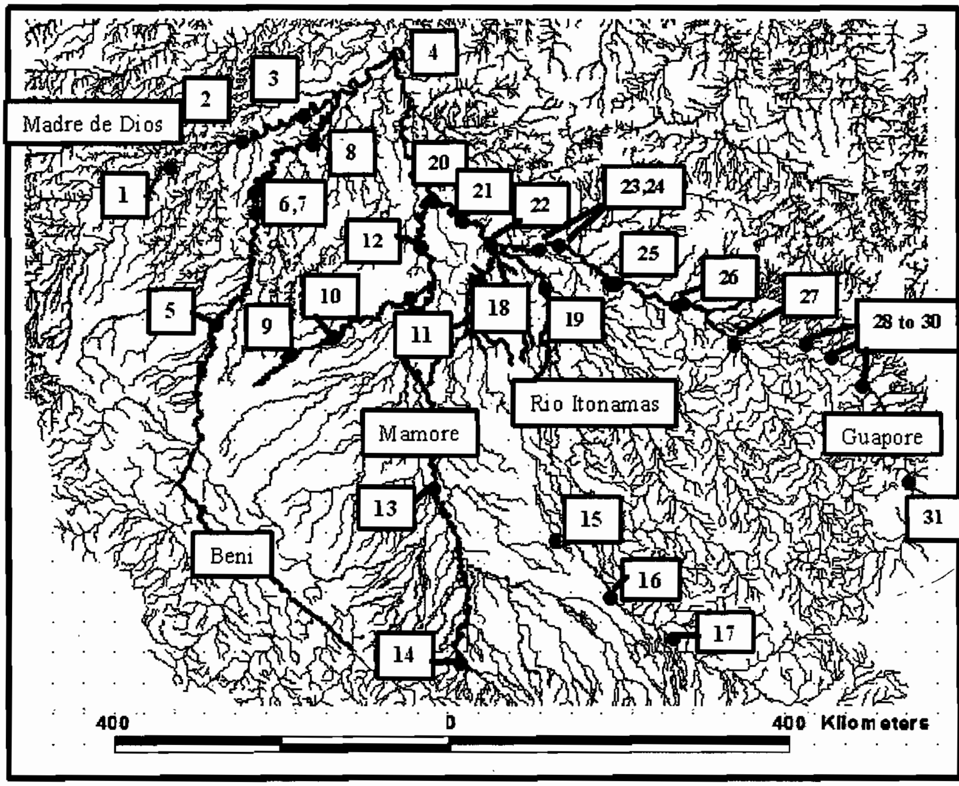

Fig. 1 Location of the virtual gauges.

therefore given with an uncertainty of 100 m, being the resolution of the image. The maximum level difference H (m) has been estimated by the following formula:

$$H = \max(h_i) - \min(h_i) \quad (1)$$

where h_i is the median of the water surface altitude at the virtual station for each ENVISAT cycle (m). Slope (m/m) has been estimated by two methods. The first is given by:

$$S1 = ((\min(h_i) - \min(h_j)) / \mathrm{dist}(SV_i \text{ to } SV_j) \quad (2)$$

SV_j being located downstream of SV_i and dist(SV_i to SV_j) being the distance between SV_i and SV_j in m, and measured by the distance measuring tool of ArcGIS.

$$S2 = 1/2[(\min(h_i) - \min(h_j) / \mathrm{dist}(SV_i \text{ to } SV_j)) + (\min(h_h) - \min(h_i) / \mathrm{dist}(SV_h \text{ to } SV_i))] \quad (3)$$

SV_h being located upstream and SV_j downstream of SV_i.

Bankfull discharge $Q1^*$ (m^3/s) was estimated for each virtual station by:

$$Q^* = 4.00 \times A^{*1.21} \times S^{0.28} \quad (4)$$

given by Williams (1978) where Q^* is the bankfull discharge, A^* cross-sectional area at bankfull and S the slope. A^* has been approximated by:

$$A^* = L \times H \quad (5)$$

Another formula is given by Bjerklie *et al.* (2003):

$$Q2^* = 0.1676 \times L^{*1.86} \quad (6)$$

The estimation of discharge was evaluated by comparing with monthly discharges estimated at two conventional gauges (Table 1).

Table 1 Minimum (Min *Q*), Mean (Mean *Q*), and Maximum (Max *Q*) of monthly discharges (m^3/s) estimated between, respectively, September 1970–November 2007, at Guaraja-Mirim, on Mamore River upstream of the confluence with the Guapore River, and May 1967–June 2008 at Porto Velho, on the Madeira River, 331 km downstream of the Beni-Mamore confluence.

Conventional gauge	Min *Q*	Mean *Q*	Max *Q*
Guaraja-Mirim	1039	7688	21280
Porto Velho	2220	18394	47400

RESULTS AND DISCUSSION

Validation

We can analyse the precision of the altimetric data by estimating the discrepancy between water levels at conventional gauges and at virtual gauges. Table 2 presents the RMS standing for the discrepancy at six couples of conventional gauges/altimetric virtual stations The RMS is calculated with respect to the best fitting regression line, the coefficient of which is provided.

As we have seen, it is difficult to assess the precision of the altimetry-derived water levels by comparison with conventional gauges, as the first is averaging the surface elevation across the channel of the river, and whereas the second is measuring at one single point located on one bank of the river. First, the precision of the conventional gauge with respect to the spatial variability of the river surface is never stated. Koblinsky *et al.* (1993) attributed part of the error in altimetric measurements to the error at *in situ* gauges, having estimated the discrepancy between two conventional gauges located on the opposite banks of the Tapajós River. But most of the time, the *in situ* gauge is given as the absolute reference of the river surface plane. Second, it is very uncommon to have an altimeter track right above the conventional gauge chosen as reference. When located at some distance from each other, the river section can change, or unknown amounts of water can reach the river between the two points from an ungauged tributary, or other unknown amounts can be temporarily lost in a derivation or an inundation plain. Analysing Table 2, we can summarize the validation situation in three categories:

(a) the regression coefficient is very different from 1, which is the case when the river sections (conventional and altimetric) are not similar: Principe da Beira (regression coefficient of 0.75; the track partly crosses over a braided part of the river), Pedras Negras (regression coefficient of 0.74; the track crosses at an angle 30° from the river's longitudinal profile, allowing it to take into account part of the longitudinal slope of the river in the altimetric measurement). The last case is that of Vila Bela de Santissima Trindade. Track 393 of ENVISAT crosses the Guapore River right above the conventional gauge, but is averaging the 100-m wide channel at this point plus the 17-km of flood plain, which gives a regression coefficient of 0.62.

(b) the regression coefficient is near 1, which could indicate a similar river section. It is the case for the comparison made at Porto Velho and Abuna. The Madeira River has an equivalent width at both the conventional and virtual stations, and the track crosses in a near perpendicular direction to the river flow. But the distance between the two measurements points is large, 18 km for Porto Velho and 30 km for Abuna. In both cases, there are some

Table 2 Comparison between *in situ* gauge and virtual gauge.

Conventional gauge	Altimeter track	Distance (km)	Regression coeff. (m/m)	Rms (m)
Porto Velho	951	18	1.06	0.395
Principe da Beira	192	5	0.75	1.349
Pimenteiras	478	0	1.02	0.170
Abuna	278	30	0.99	1.496
Pedras Negras	106	29	0.74	0.689
Vila Bela de Santissima Trindade	392	0	0.62	0.342

derivations by flood plains. For Abuna, the derivations seem very important, as seen on JERS images, and a tributary joins the river between the conventional gauge and the satellite track.

(c) the regression coefficient is near 1, and the distance between the two measurement points is null. This is the case at Pimenteiras. This is the only case where the precision of the altimetric data can be assessed: 17 cm Rms. This is within the range given in Calmant & Seyler (2006) for the ENVISAT data (decimetric accuracy), taking into account the spatial variability of the river across the channel.

Morphological parameters at virtual stations and estimation of discharge

The bankfull discharges estimated from the slope downstream of the virtual station ($S1$) and from the mean slope upstream and downstream of the station ($S2$) are very close to each other. Comparing the bankfull discharge of the Mamore upstream of the confluence with the Guapore (Md), with the maximum discharge at Guajara-Mirim, the closest value is that of D^*1, calculated only with the downstream slope. The value of D^*1 summed for the four tributaries (Sum 4tr, Table 3) is also closest to the values of maximum discharge in Porto Velho. Bankfull discharges calculated with equation (6) (only the width of the river) seem highly overestimated. Retaining the values of D^*1 as the more realistic, it can be assessed that at bankfull, the Madre de Dios contributes 33% of the discharge of the Madeira River, the Beni only 11%, the Mamore before the confluence with the Guapore 21%, the Guapore 20%, and the sum of the two at the confluence with the Beni River contributes 56% of the Madeira discharge (Table 4). These are rough estimates, but it could be useful in an ungauged basin, to have estimates of discharge obtained only from satellite data.

Table 3 Bankfull discharge by rivers, MD: Madre de Dios; B: Beni; M: Mamore; G: Guapore; Sum M-G: sum of value for Mamore and for Guapore; Md: Mamore after the confluence with Guapore; Sum 4tr, sum of the values for the four tributaries; Q^*1 bankfull discharge (m^3/s) estimated by equation (4) with $S1$ (2); Q^*2 bankfull discharge (m^3/s) estimated by equation (4) with $S2$ (3); Q^*3 bankfull discharge (m^3/s) estimated with equation (6).

	MD	B	M	G	Sum M-G	Md	Sum 4tr
Q^*1	12 912	4 161	8 183	7 765	15 918	21 812	38 885
Q^*2	13 355	4 045	7 395	8 695	15 090	19 490	36 890
Q^*3	27 782	11 057	16 270	23 881	40 151	42 075	80 914

Table 4 Contribution (%) of the four tributaries at the bankfull discharge of Madeira River.

	MD	B	M	G	Md
%total Q^*1	33	11	21	20	36
%total Q^*2	36	11	20	24	29
%total Q^*3	34	14	20	30	22

CONCLUSION

Altimetric data, as they are today, are not well adapted to continental hydrology. Nevertheless, we can assess that a radar altimeter can correctly monitor the seasonal fluctuations of water stage in great transboundary basins. From previous studies, we can state that the width of the river is not the only criterion to take into account to predict the reliability of the time series. Rivers less than 100 metres wide can be sampled, provided that they are surrounded by a flood plain, even if covered by flooded forest. Steep relief near the river, steep longitudinal slope, islands, and flow direction along track are major impediments for obtaining a reliable time series. But, in other situations, the time series obtained allow monitoring of the seasonal variation of water stage. Although it is very difficult to assess the precision of the altimetric data by comparison with conventional gauges, in cross-track situations at the exact location of a conventional gauge, discrepancies between virtual and conventional gauges can be analysed, and in all cases, they do not exceed 20 cm. In this study, bankfull discharges have been estimated for the four main tributaries of the Rio Madeira.

Of course, precision, revisit time, size of the water bodies monitored, and conditions of shallow relief impose restrictions on the use of satellite radar altimetry in hydrology. Actually, they limit its use to a range of applications: levelling of conventional unlevelled water gauges; study of the relationship between the river and its flood plain and between the river and the swamps and wetlands within the watershed; study of the elevation profile and river slope. These two last applications have opened up entirely new perspectives in the hydrological field: for example, infrastructure monitoring and planning (in particular monitoring of remote dams for producing hydropower), flood and drought monitoring and forecasting, fluvial waterway monitoring and transport planning, and fluvial dynamics of the riverbed and discharge modelling. All these applications can be conducted with joint spatial and conventional monitoring, but in remote, tropical forested areas, or in vast wetlands, unreachable most of the time, as the region in the present study, or in transboundary basins where conventional data are unevenly distributed, or lastly in a politically troubled region, radar altimetry monitoring is of prime importance. From this perspective, the SWOT mission, which will be launched around 2015, and will provide a global, quasi-continuous measurement of water surface area and elevation, will provide very valuable new data sets. A lot of improvements are yet to be achieved in processing the existing data, and it is necessary to continue other validation works in distinct environments.

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Representation of uncertainties in spatial modelling of decision processes in integrated water resources management

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Abstract This work presents a spatial modelling of a decision-making process in water management, which considers the associated uncertainties. The case under study is a coastal area in a rapid process of urbanization in Northeast Brazil. Groundwater is exploited for municipal supply and for controlling aquifer levels, so that flooding is minimized. The decision variables are the location and discharge of extraction wells, subject to flow restrictions related to contamination sources from surface and saline intrusion from the ocean. Subjective data and associated uncertainties were handled using Fuzzy Logic: public opinion on preferred areas in which to minimize flood risk, and expert opinion on contamination of groundwater. A groundwater flow numerical model simulated the impact of management alternatives on the aquifer, and Monte Carlo simulation was applied to the characterization of its parameter uncertainty. The approach was successfully implemented in a geographical information system.

Key words uncertainty; decision making; flooding; groundwater; Monte Carlo; Fuzzy Logic; GIS; Brazil

INTRODUCTION

Decision-making processes in water resources management are affected by uncertainties related to many factors: the variety of knowledge involved, natural complexity of information, incomplete or unavailable data and model errors. Identifying and estimating the uncertainties involved in the processes – from the data acquisition until the final information is generated – are difficult tasks. Moreover, when the decision making depends on spatial data or spatial analysis, it is necessary to consider the uncertainty related to their spatial characteristics, such as location, scale and resolution (Malczewski, 1999). This paper presents a spatial modelling of a decision-making process in a specific problem of water management, considering the associated uncertainties, which are modelled and visualized using geographical information systems (GIS). The tools available in a desktop GIS are explored, incorporating Fuzzy Set concepts in the spatial modelling process. The resultant maps represent the uncertainty associated with the decision alternatives. A groundwater flow model was utilized to simulate the impact of management alternatives on the aquifer. Subjective data and associated uncertainties were handled using Fuzzy Logic: public and expert opinion on preferred areas at which to minimize flood risk, and expert opinion on contamination of groundwater. Monte Carlo simulation was applied to uncertainty characterization in groundwater numerical modelling.

CASE STUDY

The case under study is a small coastal area (6.4 km^2), the Bessa District, located in João Pessoa City, State of Paraíba, in Northeast Brazil (Fig. 1). The district is undergoing a rapid process of urbanization, and appropriate urban planning is not fully implemented, nor drainage infrastructure. As a consequence, the area suffers frequent floods in the rainy season, caused by, among other factors, the high level of the shallow aquifer influenced by the proximity of a river and the ocean (Fig. 1). Thus, surface and groundwater have to be managed jointly, given their close interaction. Groundwater exploitation is necessary for municipal supply and can also minimize the risk of floods by controlling aquifer levels. However, contamination sources from surface and saline intrusion from the sea must be considered when extracting groundwater. The decision problem is the development of a groundwater exploitation plan that provides the number and location of wells

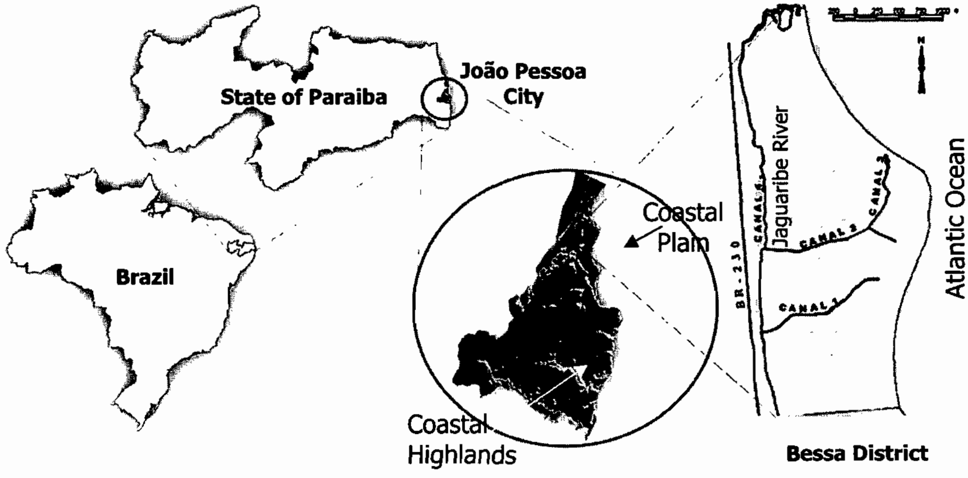

Fig. 1 Location of Bessa District.

and the discharge of each of them, so that the aquifer level is lowered at prescribed positions within the area.

The information available for the development of the groundwater exploitation plan is not plentiful. A geological map of the region and a one-year field monitoring in several observation wells provided data for preliminary calibration of a groundwater flow model. A detailed map of urban occupation with the location of the main facilities, surface topography and river network is also available. However, data on the actual soil and water contamination by surface sources (rivers, wastewater collection network, service stations, etc.) are not available and potential contamination possibilities had to be estimated by an expert. Finally, target areas for lowering the aquifer levels, and thus minimizing flood risks, were selected based on a public opinion survey and an expert consultation on the importance of urban facilities and transportation corridors in the area.

THE DECISION PROCESS AND UNCERTAINTY REPRESENTATION

Target areas for minimizing inundation risk

The selection of target areas for minimization of inundation risk was performed using two kinds of information: opinion from stakeholders, collected through interviews, and expert evaluation, based on the importance of urban facilities and transportation corridors within the area.

The opinions of stakeholders were collected by showing a map of the district and asking them to draw regions of high, medium and low priority for minimizing flood risks. They could also refine their evaluations by assigning weights (valued between 0 and 1) to each category. These weights were assumed as fuzzy membership values to the categories and the set of individual evaluations was averaged using a fuzzy T-Norm operator (Zadeh, 1973; Zimmermann, 2001).

Complete membership functions for the same categories were built by expert consultation, considering legislation and the relative importance of urban facilities for that particular district. The functions assign values of membership to the pixels in the map to each category, according to the distance of each pixel to a certain facility. For example, the *high priority* category presents highest values of membership (close to 1) for the pixels on an important facility, with values decreasing as the pixels are more distant from the facility.

Then, also using a T-Norm, the two sets of maps were combined (Fig. 2).

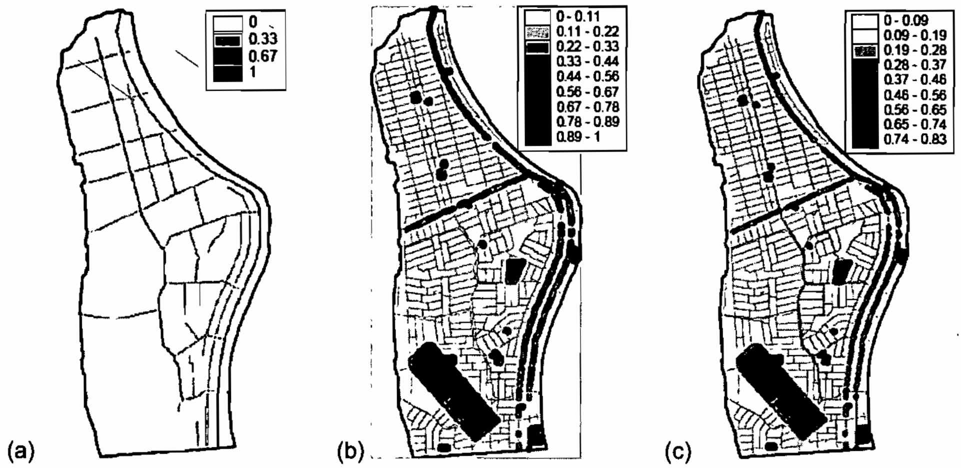

Fig. 2 Uncertainty maps for *high priority* category for (a) stakeholder opinion, (b) expert recommendation and (c) resultant average map. Scale indicates membership values to this category for each pixel.

Possibility of groundwater contamination

For the particular case of Bessa district, three potential sources of contamination were considered: hydrocarbon percolation from service stations, saline intrusion from the sea and polluted waters from rivers and channels. Since no data on actual contamination were available for the area, again expert consultation was used to assess potential contamination possibility. Using distance functions as membership functions, the estimated possibility and associated uncertainties were represented, as was done with the previous criterion. The three resultant sets of maps were aggregated, for each category (high, medium and low), using a fuzzy S-Norm operator. This operator "sums-up" the effect of all sources of contamination in the resultant map (Fig. 3).

Aggregation of subjective information

The maps (Fig. 3) were generated on a desktop GIS using its map algebra module, including the representation and evaluation of the fuzzy membership functions for each variable and the operation between different variables. The two resultant maps for target areas and contamination possibility were superimposed into a single overlay (Fig. 4(a)) for supporting the next step of the decision process.

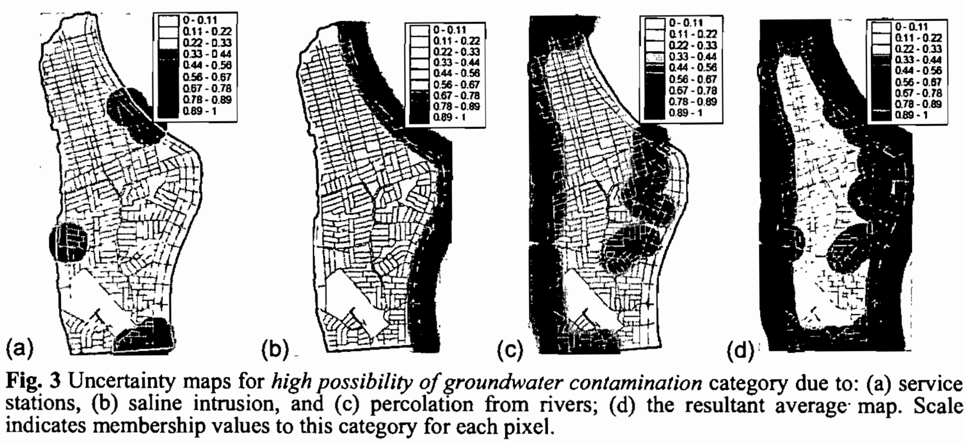

Fig. 3 Uncertainty maps for *high possibility of groundwater contamination* category due to: (a) service stations, (b) saline intrusion, and (c) percolation from rivers; (d) the resultant average map. Scale indicates membership values to this category for each pixel.

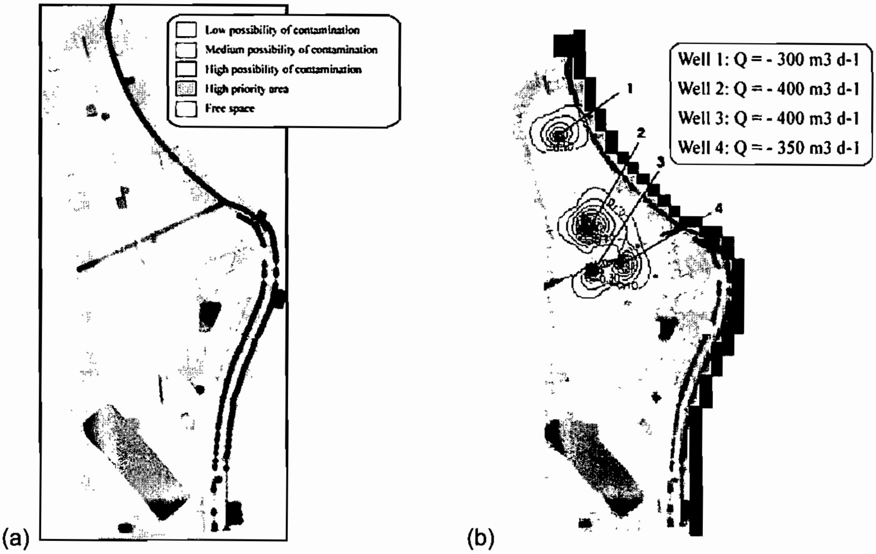

Fig. 4 (a) Overlay showing areas of high priority for lowering aquifer levels and zones of low, medium and high estimated groundwater contamination possibility; (b) location and discharge of water extraction wells and aquifer levels for the "best" management alternative.

Decision among management options

A groundwater flow numerical model (Chiang *et al.*, 1998), based on the US Geological Survey's MODFLOW (Harbaugh *et al.*, 2000), was used to simulate the impact of the management alternatives (number, location and discharge of wells) on flow trajectories and on aquifer water levels. Flow information was necessary to assess whether groundwater flows from a source of contamination into the wells, thus eventually contaminating the extracted water. The aquifer levels were used as the reference to evaluate the main objective of the management: reduce flood risk by lowering water table. In addition, municipal water demands for the district were taken as basis for extraction discharge values.

In the case reported in this paper, the "best" decision was reached by a trial-and-error procedure (Fig. 4(b)). The presence of gradual boundaries between categories – representing the uncertainty in their evaluation – make such a procedure appropriate for permitting the decision makers to use their experience to interpret the model results against the objectives and restrictions represented on the map. However, automatic optimization or search procedures can be developed considering such rules-of-thumb to take uncertainty into account in the decision.

Decision evaluation based on modelling uncertainties

Since model calibration is naturally influenced by several uncertainties, particularly when field data are scarce, it is mandatory to consider the impact of these uncertainties on the consequences of the selected decision. One approach to perform this robustness analysis of the decision is through Monte Carlo simulation (Morgan & Henrion, 1990): groundwater model parameters and/or input data are perturbed so that a large set of simulations is generated, all of them subject to the same prescribed management decision (number, location and discharge of wells). The outcome of the simulations – the map of water levels – will indicate the range of variability of water levels that can be expected to actually occur and whether the management decision is robust to the model uncertainties.

In the case reported here, few data were available for model calibration and the main uncertainty was considered to be related to the values of the hydraulic conductivity of the aquifer.

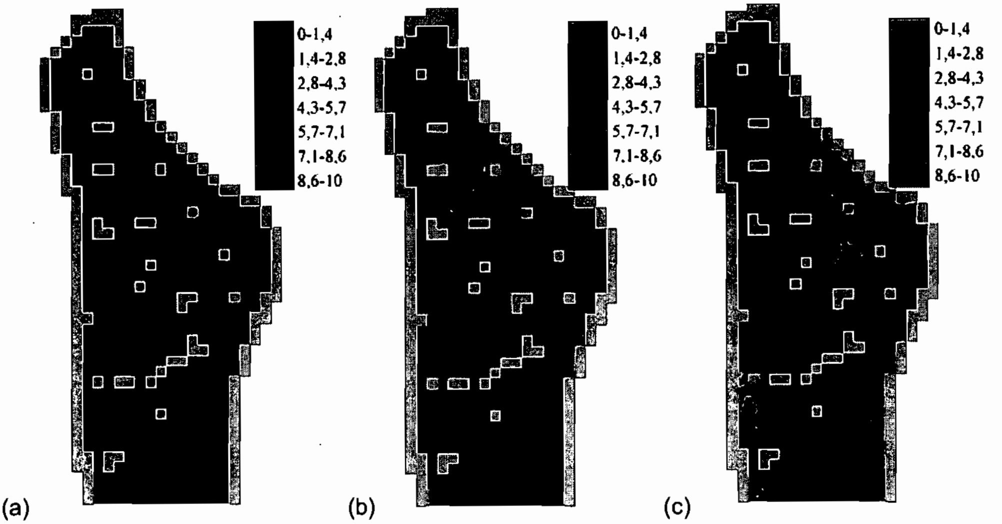

Fig. 5 Maps of spatial distribution of hydraulic conductivity: (a) values used in model calibration; (b) and (c) two of the maps generated by the Monte Carlo algorithm.

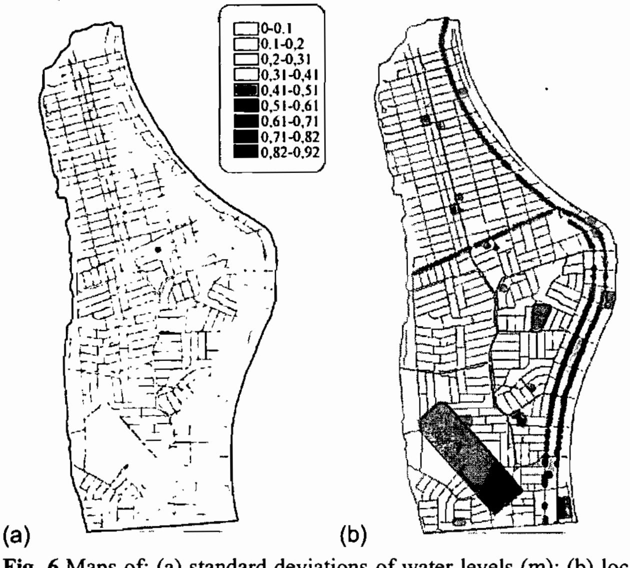

Fig. 6 Maps of: (a) standard deviations of water levels (m); (b) location of these deviations in relation to *high priority* areas for flood risk minimization.

Therefore, this variable was perturbed, for each cell of the numerical model, by the Monte Carlo algorithm. Figure 5 shows the maps of hydraulic conductivity used for model calibration and examples of maps perturbed by Monte Carlo. The results of the model simulations using the perturbed values of hydraulic conductivity show relatively high deviations among resultant water level values (Fig. 6(a)). The location of the higher deviations largely coincides with the *high priority* flood risk minimization areas selected in the decision process (Fig. 6(b)).

CONCLUSIONS

The proposed approach for considering uncertainties of several sources and types occurring in typical decision processes in integrated water resources management was tested in a real case study. The problem of lack of data and information for complete characterization of the problem and for reliable modelling was dealt with using subjective information from experts and by recognizing numerical model fragility. Stakeholder participation in the process could also be modelled. Fuzzy Logic was shown to be a suitable approach for representing uncertainty and classical Monte Carlo simulation was successfully used to evaluate the robustness of the proposed decision and expected errors in the estimated water table levels. A typical GIS tool was appropriate to represent the process and associated uncertainties.

Selecting management alternatives without considering, in an integrated way, the several uncertainties in the decision process, can lead to ineffective actions in the water management process. The general procedure illustrated here can be easily applied to several decision processes using standard tools, such as simulation models and GIS software.

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Optimal real-time management of an integrated water resources system by ANN-based hedging rules

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Abstract The main objective of this study is to develop a hydrological-economic (HE) procedure capable of incorporating long- and short-term hydrological variability for the real-time optimal operation of an integrated water resources system. The long-term information is represented by hedging rules derived by Artificial Neural Networks (ANNs) and Implicit Stochastic Optimization (ISO). The short-term information is assumed to be deterministic since meteorological forecasts are generally available. The procedure is used for the operation of a dam reservoir in the presence of groundwater supply with the purpose of minimizing economic costs and maximizing the system's sustainability. The HE procedure was applied to operate the system over a 10-year daily horizon. The results were shown to be equivalent to those obtained by a perfect-forecast deterministic model and superior to those found by the so-called Standard Operating Policy (SOP).

Key words real-time water management; artificial neural networks; hedging rules; sustainability; economic analysis; implicit stochastic optimization; integrated systems

INTRODUCTION

The social and economic development of any region is directly related to the quality and quantity of its water resources. Although the conjunctive use of surface water and groundwater is a reality for many cities around the world, only a few real-time operating procedures can be found in the literature (Philbrick & Kitanidis, 1998). Matsuyama City, which is the capital of Ehime Prefecture in Japan, is one of these cities and suffers from several problems related to water shortages. Such problems could be reduced by considering hydrological uncertainties in management models. Implicit stochastic optimization (ISO) procedures are techniques that implicitly consider this variability (Reddy, 1987; Farias *et al.*, 2006). This study consists of developing a hydrological-economic (HE) procedure capable of incorporating the long- and short-term hydrological variability for the real-time optimal operation of Matsuyama City's water resources system. The long-term information is represented by hedging rules derived by Artificial Neural Networks (ANNs) and ISO. The short-term information is assumed to be deterministic since accurate meteorological forecasts are generally available.

SYSTEM DESCRIPTION AND ECONOMIC VALUE OF WATER

System description

The integrated water supply system of Matsuyama City is composed of the Ishitegawa Dam reservoir and a set of 26 unconfined wells located around the Shigenobu River, which is the main river of its hydrographic basin. The variables of the system are shown in Fig. 1(a) and described as follows: (1) control variables R_D (release from the dam) and R_W (release from the wells), which are the release decisions to be made in order to meet demands (around 215 000 m^3/day) and economic objectives; (2) state variable S, that determines the current reservoir storage; and (3) stochastic variables I and H, which define the reservoir inflows and groundwater levels at the observation well, respectively. It is important to observe that the groundwater level at the observation well is going to be used as a basis to define the maximum groundwater release ($R_{W\max}$) that can be withdrawn from the set of unconfined wells at each daily interval, as seen in Fig. 1(b).

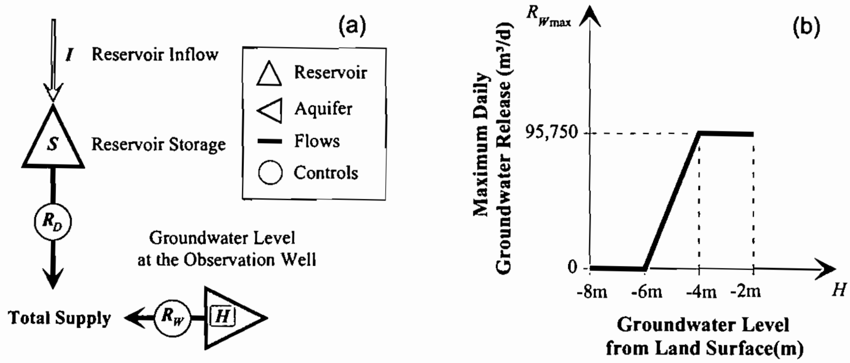

Fig. 1 System description: (a) system variables, and (b) maximum daily groundwater release as a function of groundwater level at the observation well.

Pumping cost

The marginal pumping cost (electricity, operation, and maintenance) is expected to rise with the increase of pumping rates (Philbrick & Kitanidis, 1998). In this study, the marginal pumping prices are assumed to increase linearly with daily groundwater deliveries as represented by:

$$p_p = \left(\frac{p_{p\max} - p_{p\min}}{R_{Wcap}}\right) Q_W + p_{p\min} \tag{1}$$

where p_p is the marginal pumping price; $p_{p\max}$ is the maximum marginal pumping price; $p_{p\min}$ is the minimum marginal pumping price; Q_W represents the daily groundwater deliveries; and R_{Wcap} is the maximum groundwater release capacity.

The value for pumping cost (PC) is obtained by integrating the daily demand curve from zero to the groundwater release R_W, as presented in Fig. 2(a):

$$PC = \int_0^{R_W} \left[\left(\frac{p_{p\max} - p_{p\min}}{R_{Wcap}}\right) Q_W + p_{p\min}\right] dQ_W = \left(\frac{p_{p\max} - p_{p\min}}{2R_{Wcap}}\right) R_W^{\ 2} + p_{p\min} R_W \tag{2}$$

Shortage cost

Water shortages economically affect not only industries, commerce and agriculture, but also the average consumer. The economic losses from shortages grow very quickly with the degree of rationing and one of the most used concepts to translate those losses is the users' willingness to pay (WTP) for the water (Philbrick & Kitanidis, 1998; Brouwer & Pearce, 2005). The value for an additional amount of water is related to the delivery reliability, as seen in Fig. 2(b). The producers' loss is the result of the reduced water sales during the scarcity period and the economic effects over the various users represent the consumers' loss. The demand curve can be defined by estimating the demand elasticity, which is the percent change in quantity demanded per percent change in price. The daily demand curve for this study is given by the following equation:

$$p_s = \left(\frac{Q}{\theta}\right)^{1/E} \tag{3}$$

where Q represents the water deliveries; θ is a scale factor greater than zero; p_s is the marginal price for supply; and E is the demand elasticity, which is inelastic and therefore less than zero. The value of the constant θ is estimated from an observed price and the observed water quantity at that price.

The estimation of the shortage cost (SC) can be found by integrating the daily demand curve between the water release and the demand, as shown in Fig. 2(b):

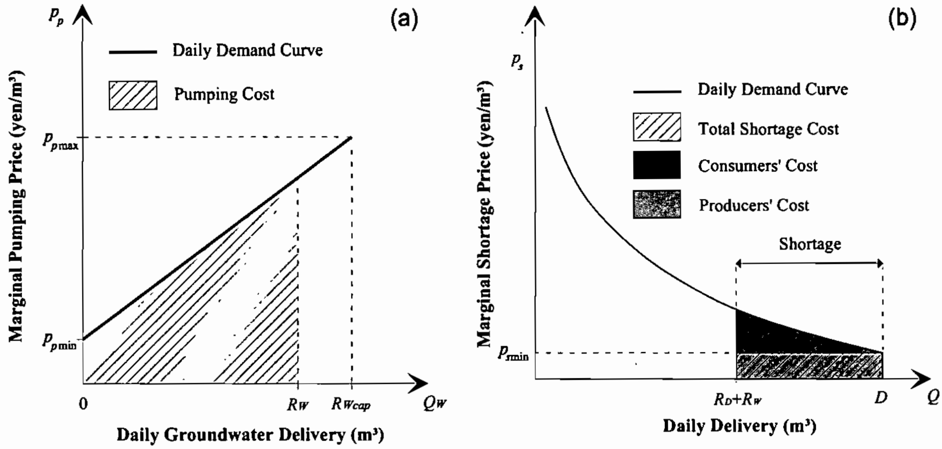

Fig. 2 Marginal price *versus* water delivery for: (a) pumping and (b) shortages.

$$SC = \int_{(R_D+R_W)}^{D} \left(\frac{Q}{\theta}\right)^{1/E} dQ = \left(\frac{1}{\theta}\right)^{1/E} \frac{(D)^{1+1/E} - (R_D + R_W)^{1+1/E}}{1+1/E} \tag{4}$$

where D is the demand and (R_D+R_W) is the total release from the water system.

HYDROLOGICAL-ECONOMIC (HE) PROCEDURE

Deterministic optimization model

This model assumes that the main objective of the operation is to find the allocations of water that best satisfy the demand without compromising the system. Another goal is to minimize the pumping cost whenever possible, i.e. every time there is an alternative optimal solution for the releases. The operation is derived for annual cycles and, therefore, the objective function of the optimization problem for year y ($y = 1, 2, \ldots, Y$) is written as follows:

$$minimize \sum_{t=1}^{365} \left\{ \alpha_1 \left[\frac{R_D^y(t) + R_W^y(t) - D(t)}{D(t)} \right]^2 + \alpha_2 \left[\frac{PC^y(t)}{PC_{\max}} \right] \right\} \tag{5}$$

where t is the day index; α_1 is the priority coefficient for the first term of the objective function; α_2 is the priority coefficient for the second term of the objective function; $R_D^y(t)$ is the reservoir release during day t of the year y; $R_W^y(t)$ is the groundwater release during day t of the year y; $D(t)$ is the demand at day t; and $PC^y(t)$ and $PC_{\max}$ are the pumping cost at day t of the year y and maximum pumping cost, respectively.

Releases and storage at each period are related to reservoir inflow and spill through the continuity equation:

$$\begin{aligned} S^y(1) &= S_0^y + I^y(1) - R_D^y(1) - Sp^y(1); \quad \forall y \\ S^y(t) &= S^y(t-1) + I^y(t) - R_D^y(t) - Sp^y(t); \quad \forall t = 2,\ldots,365; \quad \forall y \end{aligned} \tag{6}$$

in which $S^y(t)$ is the reservoir storage at the end of day t of year y; S_0^y is the initial reservoir storage at year y; $I^y(t)$ is the inflow during day t of year y; and $Sp^y(t)$ is the spill that eventually might occur during day t of year y.

The physical limitations of the system define lower and upper bounds for releases, storage and spill:

$$0 \le R_D^y(t) \le R_{D\max}(t); \quad \forall t, y \tag{7}$$

$$0 \le R_W^y(t) \le R_{W\ \max}^{\ y}(t); \quad \forall t, y \tag{8}$$

$$R_W^y(t) + R_D^y(t) \le D(t); \quad \forall t, y \tag{9}$$

$$S_{\text{dead}} \le S^y(t) \le S_{\max}; \quad \forall t, y \tag{10}$$

$$Sp^y(t) \ge 0; \quad \forall t, y \tag{11}$$

$$S_{365}{}^y \ge S_S; \quad \forall y \tag{12}$$

$$S_0{}^{y+1} = S_{365}{}^y; \quad \forall y \tag{13}$$

where $R_{D\max}(t)$ is the maximum reservoir release during day t; $R_{W\ \max}^{\ y}(t)$ is the maximum groundwater release, which depends on the groundwater level $H^y(t)$ at the observation well during day t of the year y; $S_{\max}$ is the maximum reservoir storage; S_{dead} is the reservoir dead storage; and S_S is a minimum storage for the beginning of each year y.

Implicit stochastic optimization procedure

The ISO procedure for this study has three basic steps described as follows:

(a) generate M synthetic daily sequences (365 days) of reservoir inflows and groundwater levels;
(b) optimize the system operation for all M sequences using the deterministic optimization model equations (5)–(13);
(c) use the ensemble of optimal outcomes to develop reservoir storage hedging rules.

The end-of-period reservoir storage $S(t)$ is related to reservoir storage at the previous time period $S(t-1)$, total demand during the current time period $D(t)$, reservoir inflow during the current time period $I(t)$, and maximum groundwater release during the current time period $RW_{\max}(t)$. Therefore, with information on the initial reservoir storage, total demand, inflow and maximum groundwater release for the current day, the end-of-period amount of water that should be kept in the reservoir can be defined by the hedging rules, which are established by an ANN.

ANN-based hedging rules

A multilayer feedforward ANN is responsible for deriving reservoir storage hedging rules based on the optimal results obtained by the application of the ISO procedure.

Architecture, topology and activation functions The architecture of the network is formed by the input layer, one hidden layer and the output layer. The input layer is composed of four neurons, which are the initial reservoir storage $S(t-1)$, current reservoir inflow $I(t)$, current total demand $D(t)$, and current maximum groundwater release $RW_{\max}(t)$. The number of four neurons in the hidden layer was determined based on a trial-and-error procedure. The end-of-period reservoir storage $S(t)$ is the single neuron of the output layer. The network topology is shown in Fig. 3. The tan-sigmoid and linear functions are chosen as the activation functions for the hidden and output neurons, respectively.

Training process The training is performed by the back-propagation algorithm which has been successfully applied to water resources systems (Haykin, 1999). In this approach, the Levenberg-Marquardt (LM) algorithm is used for the back-propagation training. A detailed explanation of LM algorithm is provided by Hagan & Menhaj (1994). In order to improve generalization, the ANN training is stopped by the Early Stopping Method (Demuth & Beale, 2005).

Short-term optimization using the ANN-based hedging rules

This model uses the same objective function of the deterministic optimization model and incorporates the ANN-based hedging rules into the constraints of the problem. In this procedure

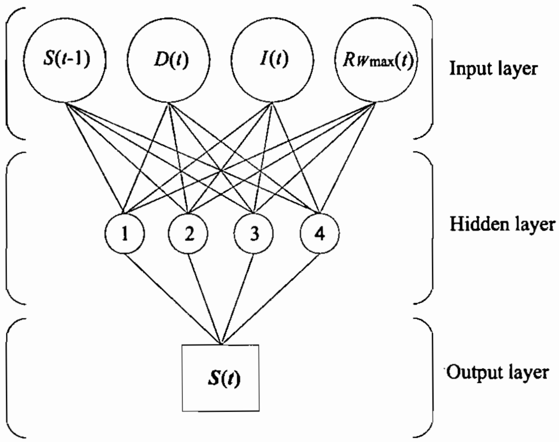

Fig. 3 Topology of the ANN model responsible for determining the hedging rules.

the optimization is not used for annual cycles anymore and the optimization problem is formulated as follows:

$$minimize \sum_{t=1}^{N} \left\{ \alpha_1 \left[\frac{R_D(t) + R_W(t) - D(t)}{D(t)} \right]^2 + \alpha_2 \left[\frac{PC(t)}{PC_{\max}} \right] \right\} \tag{14}$$

subject to:

$$S(1) = S_0 + I(1) - R_D(1) - Sp(1)$$
$$S(t) = S(t-1) + I(t) - R_D(t) - Sp(t); \quad \forall t = 2, \ldots, N \tag{15}$$

$$0 \leq R_D(t) \leq R_{D\max}(t); \quad \forall t \tag{16}$$

$$0 \leq R_W(t) \leq R_{W\max}(t); \quad \forall t \tag{17}$$

$$R_W(t) + R_D(t) \leq D(t); \quad \forall t \tag{18}$$

$$S_{\text{targ}}(t) \leq S(t) \leq S_{\max}; \quad \forall t \tag{19}$$

$$Sp(t) \geq 0; \quad \forall t \tag{20}$$

where N is the short-term operating horizon and $S_{\text{targ}}(t)$ is the target reservoir storage at period t.

The N-step-ahead target reservoir storages S_{targ} are obtained by the recursive application of the ANN-based hedging rules using short-term forecasts of $I(t)$ and $R_{W\max}(t)$ for the next N days. During the real-time operation, the short-term optimization model finds the optimal releases for N days ahead; however, only the allocations for the current day are used. The procedure is repeated for the next day and so forth until the final day of operation is reached. Like the deterministic optimization model, the HE procedure was constructed in the MATLAB platform.

APPLICATION AND RESULTS

The maximum storage ($S_{\max}$) of the Ishitegawa Dam reservoir was assumed to be 8 500 000 m^3 although the actual capacity is 12 800 000 m^3. The extra space is always empty so as to protect the region against floods.

The minimum and maximum marginal pumping prices ($p_{p\min}$ and $p_{p\max}$) were estimated as 20 yen/m^3 and 60 yen/m^3, respectively. The minimum marginal shortage price ($p_{s\min}$) was assumed to be the subsidized price paid by the consumers, which is 160 yen/m^3. The demand curve elasticity for city supply (E) was projected as –0.32, which is a reasonable value if compared to those found in the works of Martin & Thomas (1986) and Dandy (1992).

For the application of the HE procedure, the short-term operating horizon was set to 5 days (N = 5 d). The reservoir inflows and groundwater levels for this operating horizon were assumed to be reliable since accurate meteorological forecasts are generally available. In view of the fact that the objectives are very conflicting, the priority coefficient α_1 was kept superior to α_2 ($\alpha_1 = 1 \times 10^8$ and $\alpha_2 = 1$) in both deterministic and short-term optimization models. As a consequence, the releases were given priority over the option of reducing the pumping cost.

The ISO process was run under an operating horizon of 365 250 days (1000 years). The sequences of reservoir inflows and groundwater levels were generated by the first-order auto-regressive (AR-1) multivariate model suggested by Matalas (1967), which successfully incorporated the statistical features of the historical data into the generated values, as seen in Fig. 4. This provided 365 250 days of optimal daily reservoir storages. The data on initial reservoir storages, total demands, inflows, maximum groundwater releases and end-of-period storages were grouped and trained by the ANN model.

The HE procedure was applied to the operation of the system from 2001 to 2003 and compared to the results obtained from the utilization of the deterministic optimization model using perfect forecasts. The operation using the deterministic model gives us the "ideal" releases that should be employed since it has knowledge of future inflows and groundwater levels for one year ahead. Additionally, a simulation based on the so-called Standard Operating Policy (SOP) was used for comparison. The SOP states that the demands should be met whenever possible (Loucks *et al.*, 1981). In this study, the SOP prioritizes the utilization of the reservoir water over the groundwater. The accumulated costs due to pumping and shortages for all procedures were also calculated. The water allocation and cost results are displayed in Fig. 5.

Examination of Fig. 5(a)–(c) shows that the operation using the HE procedure tries to allocate water in a way very similar to the deterministic optimization model. This information indicates that the results from the derived policies were quite satisfactory, because they have information only on the previous reservoir storage and forecasts of reservoir inflows and groundwater levels for five days ahead, whereas the deterministic optimization model has knowledge of forecasts for one year ahead and thus better means to define superior policies. Comparing the results from the deterministic model with the SOP, it can be seen that the optimization model tries to mitigate the large concentrated deficits that happen with the simulation by decreasing the releases prior to shortages periods so that the overall deficit also diminishes. Figure 5(d) suggests that even though the SOP tried to reduce pumping costs by using the reservoir water, its application was more expensive than the other procedures due to the higher shortages in scarcity periods.

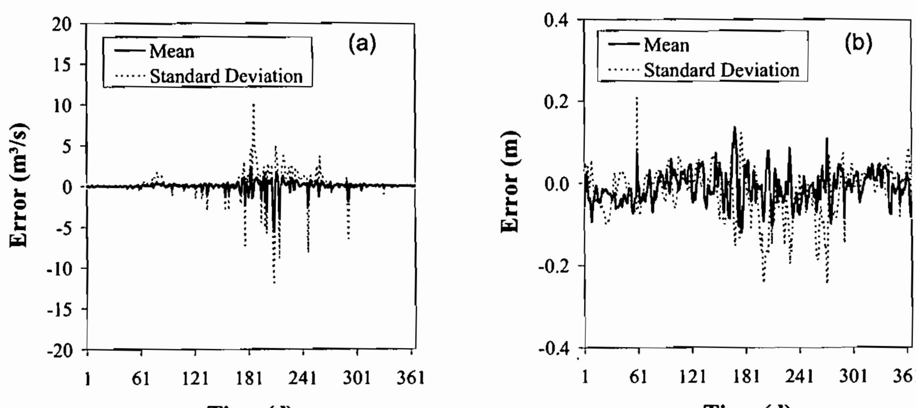

Fig. 4 Errors between generated and observed daily means and standard deviations for: (a) reservoir inflows and (b) groundwater levels.

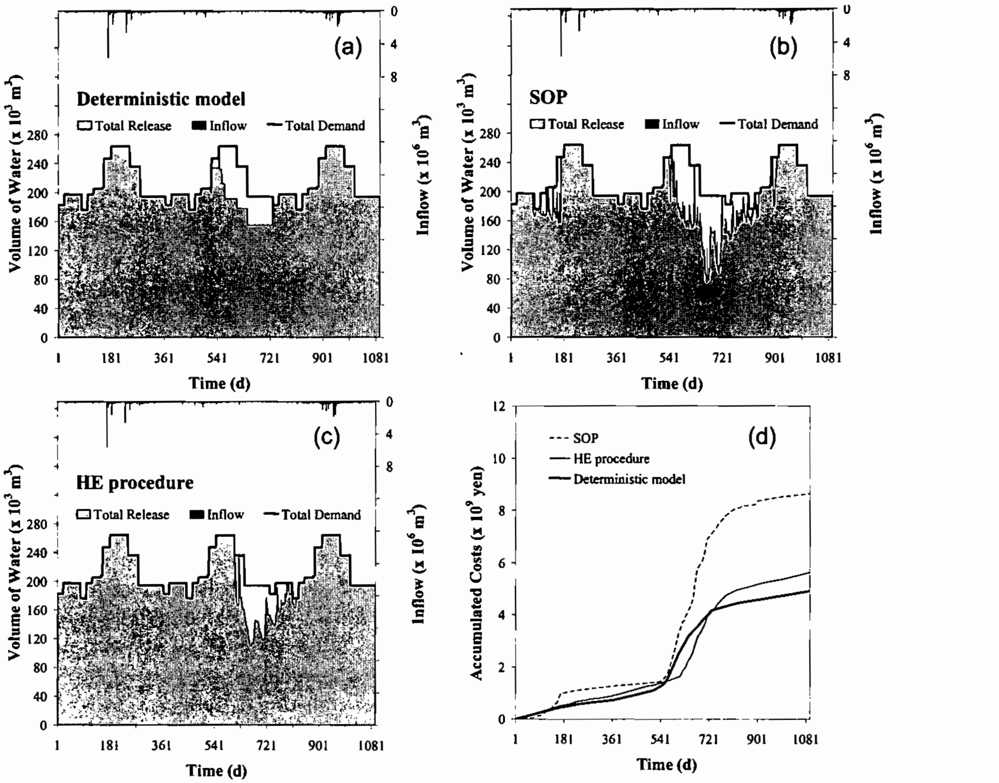

Fig. 5 Allocation of water by: (a) deterministic model, (b) SOP and (c) HE procedure; and (d) results of accumulated costs for pumping and shortages by all procedures.

CONCLUSION

A hydrological-economic (HE) procedure was defined in order to devise sustainable and economic policies for a dam reservoir in the presence of groundwater supply. The results showed that the HE procedure and the deterministic model generated more attractive hydrologic and economic policies than the standard rules of simulation.

The deterministic model produces the best policies for the system operation. However, since perfect forecasts for long periods are not available, the deterministic optimization model is not practical. On the other hand, the HE procedure presented results very similar to the ideal ones and needs information only on the initial reservoir storage and forecasts for five days ahead. As a result, the HE procedure may be a useful support for the sustainable and economic management of water supply systems with reservoirs and groundwater sources.

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Integrated approaches for water resources management: examples from France

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Abstract Integrated water resources management (IWRM) implies a multidisciplinary approache to consider all aspects of the water cycle, including surface water and groundwater, but also socio-economic activities and the cultural context. There is a variety of approaches, methodologies and tools that could be used to support IWRM. The choice must be made according to the context, the objectives, the scale of the problem to tackle and the degree of precision that is needed. This paper presents two different modelling approaches which were adopted to construct tools dedicated to integrated water resource management. In the first case study, located in the west Paris area, France, a model including an optimisation component was proposed to find the best operational exploitation of groundwater and surface water with minimal cost. In the second example, a numerical tool, SimAGE, was developed to explore water management scenarios at the catchment scale and to assess their environmental and economic impacts.

Key words integrated water resource management; hydro-economic assessment; numerical and mathematical modelling; decision support tools; SimAGE; La Réunion

INTRODUCTION

Nowadays, good water governance is essential to ensure the sustainable use of water resources in the context of global change and growing water demand. This requires that the socio-political and administrative levels adopt the concept of Integrated Water Resources Management (IWRM) with participatory processes (Third World Water Forum statement, Kyoto, March 2003). This concept of IWRM appeared in the early 1970s, but was only endorsed by the United Nations in the so-called "Dublin Statement" (Conference on Water and the Environment, Dublin, 1992). It has been defined as a combination of old areas of science and research to consider problems in new, more holistic ways (Parker *et al.*, 2002). Actually, IWRM today implies multidisciplinary approaches to consider all aspects of the water cycle, including surface water and groundwater, but also the socio-economic activities and cultural context. It relies on field studies and enquiries, data collection and analysis, and conceptual and mathematical or numerical modelling.

However, in the field, true IWRM is seldom achieved, notably because of a lack of strong political will, and of explicit legislation and barriers generated by the institutional set-ups (as separate responsibilities for surface water and groundwater). Yet the know-how and the tools exist: guidelines based on dedicated studies, multi-component numerical and mathematical models, and integrated decision support systems are among the various technical tools available to support water managers in their choices and implementation of IWRM. The scientific approaches adopted to develop these tools may be very different and depend both on the objectives of the end user (operational, medium- or long-term management and investment) and on the spatial and temporal scales of the application.

This paper presents two different analysis and modelling approaches that were adopted to construct tools dedicated to integrated water resource management in two very different settings: the first case study, located in the west Paris area, France, needed support for best operational exploitation of groundwater and surface water at a local scale; the second, located on La Réunion Island, had to face water quality issues at a catchment scale.

IWRM BASED ON AN OPTIMAL, INTEGRATED HYDRO-ECONOMIC MODELLING APPROACH—EXAMPLE OF THE WEST PARIS AREA

Background

In the Paris area, several large well fields abstract water from the chalk aquifer in the Seine valley both north and south of the city. Due to their position close to the Seine River, abstraction leads to induced recharge from the river, thereby increasing the natural yield of the aquifer. Sometimes, however, this is not enough and the high water demand forces introduction of artificial recharge in the well fields to avoid groundwater depletion and the rapid collapse of the groundwater levels. In some cases, without artificial recharge, the groundwater resources may dry out in less than two weeks.

Presentation of the case study—stakes and objectives

The hydro-economic approach for IWRM described below was first developed for the large Croissy well field, which extends over a distance exceeding 2 km along the Seine River in the Paris west area (Pennequin *et al.*, 1991, 2003). This well field included at the time over 40 high-yield wells tapping water from the Senonian chalk aquifer (and sometimes from the overlying alluvial layer as well), and nine artificial recharge basins used to avoid depletion of the groundwater resource and to maintain a raw water quality at an acceptable level in the aquifer system (Fig. 1). Average monthly abstraction exceeded 4×10^6 m^3 to provide water to several hundred thousand inhabitants in the Paris west area. The average monthly artificial recharge ranged between 35 and 50% of the monthly abstraction volume. Artificial recharge was performed using water from the nearby Seine River; this water was injected into the aquifer after undergoing an adapted pre-treatment. Today, this general situation still prevails.

Fig. 1 Schematic representation of the groundwater exploitation in the large Croissy well field (west Paris area).

The cost of the exploitation scheme with artificial recharge used at Croissy had the inconvenience of being relatively high compared to cost of the more classic groundwater exploitation scenario involving abstraction from wells only: about 30–50% of the cost per cubic metre of water produced at Croissy was due to artificial recharge. However, several hydro-economic studies showed that the Croissy exploitation scheme made sense, and turned out to be more efficient and, overall, economically more reasonable than alternative solutions elaborated then to alleviate the stress on the Croissy aquifer system and meet the local water demand (*development of new water resources – increase in exploitation of existing resources*).

In an attempt to lower the price per m^3 of water sold to its customers, the private company in charge of managing the Croissy aquifer agreed to develop an optimal management tool for the entire well field (wells and artificial recharge basins) and for the aquifer system (Chalk, alluvial layer, Seine River) that it abstracts water from. The objective set out was to: (1) minimize the exploitation cost; (2) optimize the couplet "abstraction" (from the wells)–"injection" (through the artificial recharge basins); and (3) ensure the sustainability of the local water resource.

Elaboration of the optimal integrated water resources management model

To reach the above objectives, an optimal integrated water resource management model was developed. It included several mathematical and numerical models, including: a quasi three-dimensional numerical groundwater flow model for the aquifer system, a Seine River–aquifer system water exchange model, an infiltration–runoff model, a prevision model for the evolution of the Seine–aquifer water exchange capacity and a hydro-economic optimization model. The reader is referred to Pennequin *et al.* (1991) for the basic mathematical expressions used for the main simulation and parameter estimation models developed in this case study.

The hydro-economic optimization for the functioning of the well field-aquifer system includes minimizing an economic base function subjected to a series of constraints, several of them being nonlinear. The base function, or governing equation, for the optimal integrated water resource management model can be summarized using the following mathematical expression:

$$\min_t (f(x) = [\sum_{i=1}^{n} C_{f_i} X_{f_i} + \sum_{j=1}^{m} C_{B_j} X_{B_j}]) \tag{1}$$

for $C_{f_i} > 0$; $C_{B_j} > 0$; $X_{f_i} \geq 0$; $X_{B_j} \geq 0$; $n > 0$; $m > 0$, $t \in T$ (T is the simulation time)

where C_{f_i} is the integrated unit cost for well exploitation (F/L^3); C_{B_j} is the integrated unit cost for artificial basin operation (F/L^3); X_{f_i} is the abstraction yield for phase t (L^3/T); and X_{B_j} is the injection yield for phase t (L^3/T).

The constraints take into consideration all significant parameters related to the context, the water needs, the aquifer system and the exploitation setting, which may impose limitations or unavoidable obligations in the course of the exploitation of the water resource. Among these are: (a) the evolution of the water demand; (b) the main parameters, operation rules and operational/limiting capacity of the exploitation system (infiltration capacity of the recharge basins, maximum well yield, technical breakdown, maintenance, etc.); (c) the economic parameters and their evolution through time (energy cost, treatment cost, etc.); (d) the status of the aquifer system and its interfaces (computed by the simulation models); (e) the possible interference mechanisms between wells and artificial recharge basins (computed by customized simulation models); (f) target groundwater levels (according to optimum annual evolution curves for the resource established to ensure in all climatic conditions the balance between its potential, water needs and the need to minimize exploitation costs); and (g) maximum induced recharge to avoid water quality degradation, etc. These models are built around a database and a user-friendly interface (Fig. 2).

The optimal integrated management model was built to operate on a monthly basis, whereby each month, the whole set of models is first updated with the current data and, next, forecasted computations for the exploitation are made based on expected data computed for the coming period, regarding both the aquifer system and the exploitation setting. Therefore, at the end of each month, an optimum exploitation scenario is computed for the month ahead, at minimum cost for the operator in charge of the water resource abstraction; the latter just has to apply the exploitation instructions given by the model (average monthly yield for each well, average monthly water volume per basin for artificial recharge, maintenance period for selected wells and basins, etc.).

Outcome and benefits of the approach

The benefits of this optimal IWRM approach are multiple and multiform: first, the direct exploitation costs can be decreased significantly. This was the case in Croissy, for example, where the exploitation costs for 1989 and 1990 (the model started functioning in 1988) actually decreased compared to the previous years, in spite of the severe drought which prevailed, and which should normally have led to a higher cost if past exploitation procedures had been maintained.

Indirect benefits are also numerous. In the case of Croissy, for example, repeated use of the model progressively allowed the resource managers at all levels to become aware of the aquifer system behaviour and of the relationship that existed between aquifer behaviours and economic aspects. This led to abandoning several anti-economic exploitation procedures and to optimizing the combined use of abstraction from wells, induced recharge from the Seine River and artificial

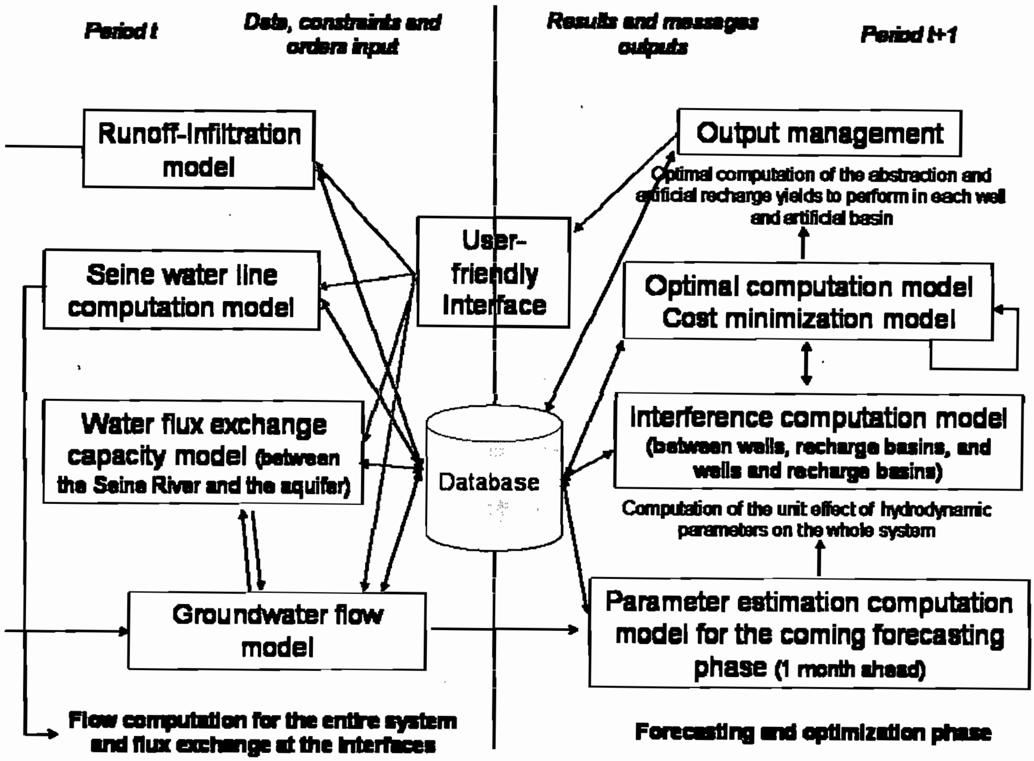

Fig. 2 Structure of the Croissy optimal hydro-economic model developed in 1988 (from Pennequin *et al.*, 1991, 2003).

recharge to meet the set targets in terms of quantity (groundwater levels) and quality (resulting groundwater quality). Furthermore, this fine tuning of the exploitation avoided making unnecessary costly investments for new water resources, and allowed for a better rationalisation of the water abstraction over the entire area (far beyond the well field area).

Finally, beside obvious multiform economic gains, this approach also allows great improvements to be made to the definition of the exploitation framework several weeks, and even (often) several months, ahead of time, as it provides the means to anticipate normal events, giving the water resource manager the opportunity to better control the evolution of his water resource, thereby securing water distribution in all situations.

IWRM BASED ON SCENARIO ASSESSMENT—EXAMPLE OF LA REUNION

La Réunion is a French mountainous volcanic island with a tropical climate. It has important water resources but they are not equitably distributed between east and west, or highlands and coast. Moreover, these resources are particularly vulnerable to turbidity (for surface water) and saline intrusion (for groundwater). Thus, water managers and stakeholders expressed the need for tools to inform and support their decisions in order to set up a sustainable and concerted water management on the island.

Presentation of the case study

The case study is located in the southern part of La Réunion and covers an area of 625 km^2 (Fig. 3). It includes seven small and medium towns (Saint-Louis, Entre-Deux, Les Avirons, Etang-Salé,

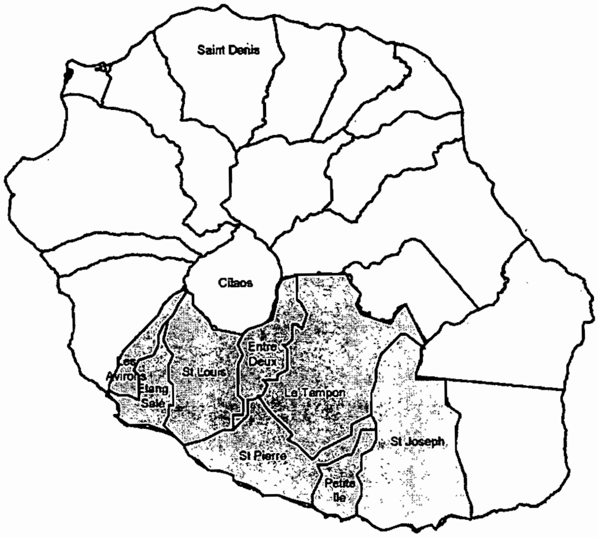

Fig. 3 Location of the study area on La Réunion island (the island is ~65 km across).

Saint-Pierre, Le Tampon and Petite-Ile) with a total of 207 000 inhabitants, two irrigated areas (7000 ha in total) mainly dedicated to sugar cane crops, and an important sugar factory with a production of about 210 000 tons a year.

The coastal aquifer can be divided into five independent units (Gol, Cocos, Pierrefonds, Saint-Pierre, Remparts) which currently supply an annual water volume of 28 $\times 10^6$ m^3 (in 2005). Taking into account the La Fournaise aquifer system, the groundwater exploitable resources of the study area are estimated to be around 130 $\times 10^6$ m^3. The two branches of the Saint-Etienne River (namely the Bras de la Plaine and the Bras de Cilaos) which provided respectively 82.5 $\times 10^6$ m^3 and 22 $\times 10^6$ m^3 in 2005, are the main surface water resources of the area. Many little springs located on the volcanic plateau are exploited for drinking water supply for a total of 11.5 $\times 10^6$ m^3 in 2005. The estimated annual available resources of the study area are up to 250 $\times 10^6$ m^3.

The main stakeholders and actors of water management in the study area are two private companies in charge of providing drinking water (CISE and VEOLIA), a public company which operates a few wells and the two important uptakes in the rivers to supply all the area with non-treated water (SAPHIR), and a town association which exploits an important spring (Syndicat des Hirondelles).

Relying on the analysis of all the available information and data related to water in the area, a conceptual scheme was established (Fig. 4) in order to identify the main resource units (groundwater, springs and the two rivers), the main demand units (towns for drinking water, industries and irrigated areas) and the links that existed between them (including water producers and suppliers).

Main management issues and suggested options

The water issues and management problems were identified by means of stakeholders and expert interviews. They are mainly related to the current important use of surface water, both for irrigation and for drinking water. It appeared that the drinking water demand cannot always be satisfied due to the poor quality of river water during the rainy season. Moreover, water flows downstream from the river's uptakes are usually lower than the required environmental protected rates. And finally, as the population of the area increases by 1.4% each year, it is clear that other resources will have to be exploited in the coming years in order to face the growing drinking water demand.

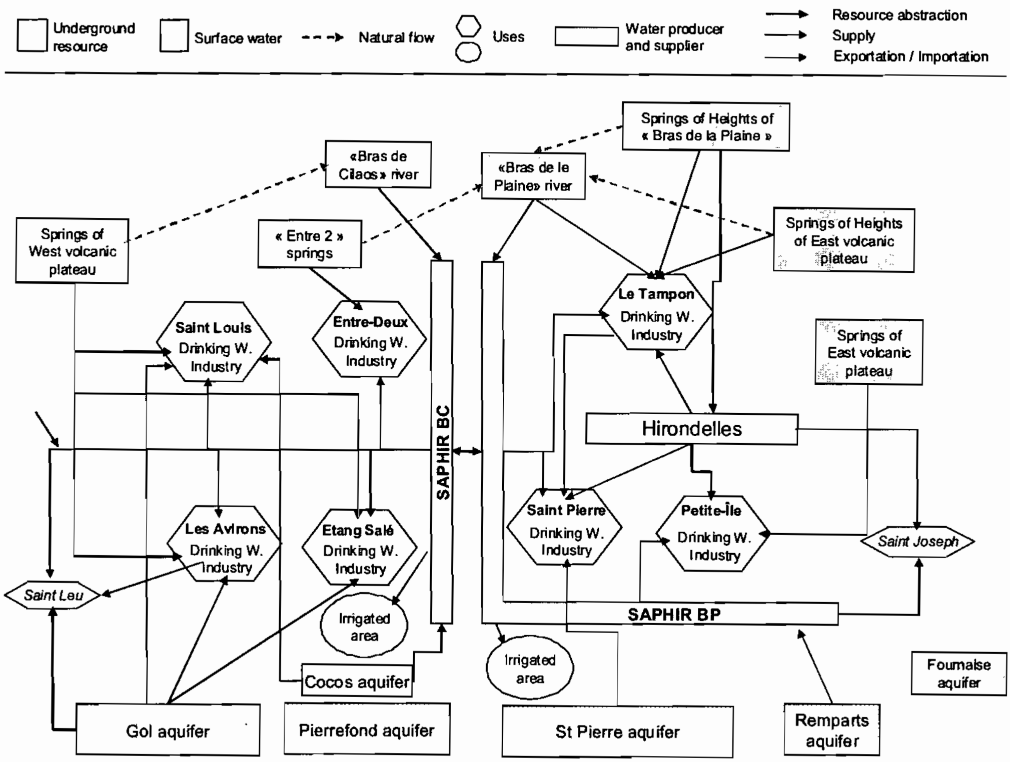

Fig. 4 Conceptual schema of resources and demand in the La Réunion case study.

In order to address these problems, several management options were proposed:

- systematic treatment of surface water when harnessed for drinking water;
- improvement of supply networks efficiency (from 55% up to 75% in 2020);
- modification of the water distribution schemes with more water coming from aquifer exploitation and less from river and springs exploitation; and
- application of measures to encourage people to save water in order to reduce the drinking water demand.

These options were combined to form scenarios which were economically assessed through a cost-benefits analysis.

SimAGE: a numerical tool dedicated to IWRM of the study area

In order to assess the environmental and economic impacts of different management scenarios, all the previous results were integrated in a numerical simulator named SimAGE. Developed using VBA for Excel, SimAGE is a tool that allows the exploration of management scenarios through a user-friendly interface. The user defines his scenario through global parameters such as the simulation horizon, the application or not of measures to encourage people to save water, or the parameters that determine the agricultural demand. He also sets specific parameters for each town, such as the drinking water demand parameters (number of inhabitants, network efficiency, etc.), the industrial demand level and the supply scheme (list of local and outside exploited resources or providers).

Then, SimAGE computes different water balances at a monthly time step: water demand and available water resources for each town, quantity of water to supply to its clients for each producer, and subsequent water withdrawal in each resource unit. Finally, a cost–benefit analysis is performed for this user-defined scenario. Its impact on the water price and on the consumer bill is also evaluated.

An Excel file is generated that contains all the results for each simulated scenario. The first worksheet presents a synthesis of all the results. Green or red colours are used to display the main indicators of scenario sustainability level. The subsequent worksheets present the balance between demand and resource, or available and exploited resources at different scales. The last worksheet is dedicated to economic results.

Discussion of the SimAGE approach

From a technical point of view, the conceptualizion and development of SimAGE benefited from lessons learned within previous projects, especially the Hérault Middle Valley model (Lanini *et al.*, 2004). The SimAGE interface (a) was designed to look like a conceptual scheme which allows the users to share a common representation of their socio-hydrosystem; (b) includes a real economic module and thus proposes a real coupling between environmental and economic scenario assessment; (c) although not standalone software, runs on MS Excel which is widely available.

SimAGE is a decision support tool which aims to explore water management scenarios. It has not been designed to find the best management option for the case study (it does not include an optimisation module). Actually, several different users are likely to use SimAGE, and they all have different individual constraints and criteria to define the best option. For example, a town mayor could aim to ensure drinking water supply in his town under any circumstances, whilst another could have the objective to lower water prices.

As SimAGE provides results at different scales, each user can understand the environmental and economic impacts of the scenario he has defined, not only for that individual case, but also for the other stakeholders and for the whole area. Thus, SimAGE can be used to inform debates between stakeholders who are responsible for defining a water management policy at the case-study scale.

CONCLUSION

In most cases, looking at only one aspect of the water resource (surface water only or groundwater only) does not lead to efficient IWRM, and even engenders mistakes which often trigger ill-defined, poorly-targeted, and erroneous investments, at high cost for the society. IWRM need to take at least the full water cycle into consideration. But often today we need to go one step beyond and IWRM needs to address the neighbouring economic or socio-economic context as well. This is the basis for sustainable development.

As shown in this paper through two examples, there is a variety of approaches, methodologies and tools that could be used to support IWRM. The choice must be done according to the context, the objectives, the scale of the problem to tackle and the degree of precision that is needed. Actually, rationalizing day-to-day exploitation requires good knowledge of the water resource and the best possible mathematical modelling approach in relation to the quality of the data available and the degree of precision needed. Minimizing economic exploitation and maintenance costs is often highly desirable. In contrast, larger-scale water resource management can often be dealt with using global or lumped hydrological models consistent with the scale addressed and the precision and density of the available data; however, it requires a real economic component able to compute a cost–benefit or cost-effectiveness analysis of water management options.

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Fuzzy group decision making and its application in water resource planning and management

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Abstract Multi-criteria decision making (MCDM) and group decision making (GDM) are two well practiced approaches for solving decision-making problems. Group decision making combines opinions from decision makers (DMs) into a coherent group decision. The aim of this paper is to develop a new fuzzy group decision making algorithm and apply it to a real-world groundwater development problem to select the most preferred alternative. Five DMs were selected to express their judgments about the relative importance of the criteria (water pumped, cost, and risk) and have given their preferences as ordinal preferences, utility values, fuzzy preference relations and multiplicative preference relations. The decision makers' opinions were aggregated based on a fuzzy group decision making. An ordered weighted-averaging (OWA) operator has been used to aggregate decision makers' opinions. A consensus measure is also defined to evaluate convergence between the group and individual opinions. The outcomes of the proposed algorithm show that the proposed fuzzy group decision making approach is a relevant approach to aggregate the individual expert's opinion and is a desirable tool to reach consensus among DMs.

Keywords fuzzy group decision making; groundwater resource management; aggregation

1 INTRODUCTION

Multi-criteria decision making (MCDM) and group decision making (GDM) are two widely practiced approaches to solve decision making problems for ranking of alternatives. Numerous applications of multi-criteria decision making techniques to various water resources planning problems have shown that they are well suited to these problems and may be employed efficiently (Abrishamchi *et al.*, 2005; Mianabadi & Afshar, 2008).

More information (knowledge), creativeness and better understanding by working in a group are considered as advantages of group decision making over multi criteria decision making (Chen, 2005; Choudhurya *et al.*, 2005). Therefore, various researchers have focused attention on increasing the ability of the group to make the qualitative decisions (Ng & Abramson, 1992; Robertson, 2002; Chen, 2005; Mianabadi & Afshar, 2007).

The aim of this paper is to develop a new fuzzy group decision making procedure. It investigates the feasibility of applying a fuzzy group decision making consensus-based methodology in selecting the best possible alternative for groundwater resources development. Hence, in Section 2, a background of the consensus-based group decision making model has been developed. Approaches towards consensus in-group decision making and selection of the best possible alternative are introduced in the Section 3. Here, we describe the methodology and represent it in the form of an algorithm. In Section 4, the environment, alternatives and attributes are addressed and a real-world case example is solved.

2 FUZZY GROUP DECISION MAKING, GENERAL CONCEPTS

Kacprzyk *et al.* (1992) defined the ultimate goal of a procedure in group decision making as to obtain an agreement between the experts on the choice of a proper decision (i.e. to reach consensus). Generally, the approaches towards consensus in the literature can be divided into two groups (Chen, 2005). The first treats consensus as a "mathematical aggregated consensus" (Ng & Abramson, 1992). In the second type, the experts are encouraged to modify their opinions to reach a closer agreement in opinions (Hsu & Chen, 1996).

In a fuzzy environment, the group decision making problem can be solved in four steps as depicted in Fig. 1. In the first step, alternatives, criteria, and DMs are specified. The weight of each criterion and DM are specified and decision makers assess performances of the alternatives. In the second step, preference evaluations from each expert must be unified. The third step is to aggregate the opinions of all group members to a final score for each alternative, followed by score rating to select the preferred alternative. This score is often a fuzzy set or a linguistic label, which enables the decision makers to order the alternatives. Finally, the facilitator assesses the consensus level and the individual contribution to the group decision. If consensus measure is bigger than at least the amount of consensus measure identified by the facilitator, decision making is finished. Otherwise, to reach a consensus, the decision makers (DMs) should modify their opinions.

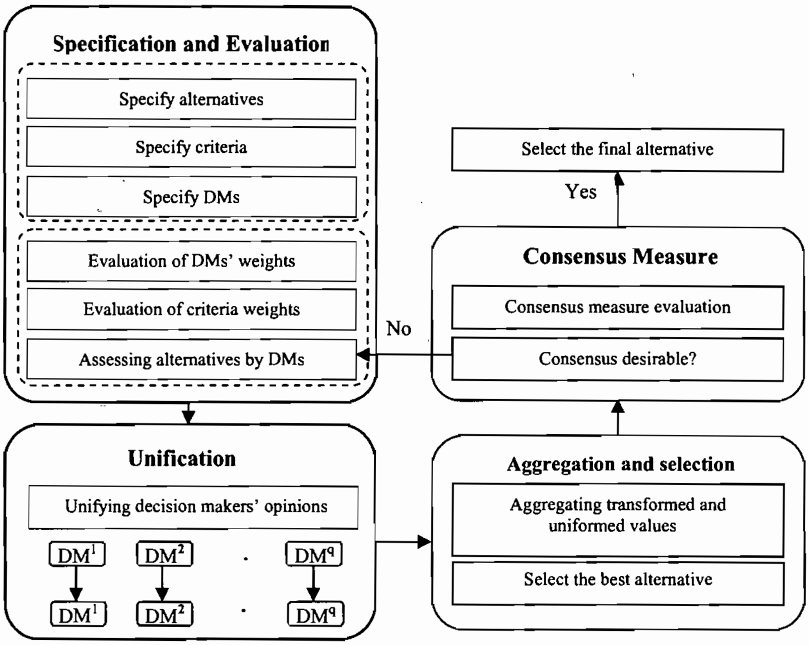

Fig. 1 Fuzzy group decision making process (adapted from Mianabadi & Afshar, 2007).

Each expert has his/her own ideas, attitudes, motivations, and personalities. Different experts will give their preferences in different ways; hence, DMs' opinions must be unified employing a unification process. Herrera-Viedma *et al.* (2002) state that group members may express their opinions as: (1) ordinal preferences, (2) utility values, (3) fuzzy preference relations, and (4) multiplicative preference relations. These opinions can be converted into various representations using appropriate transformations. In this paper, fuzzy preference relations have been used to unify opinions. Fuzzy relationships in the evaluation are used to incorporate the uncertainties in the decision opined by a particular decision maker. The decision making becomes difficult when the available information is incomplete or imprecise (Zadrozny, 1997; Choudhurya *et al.*, 2005).

In the aggregation stage, all experts' opinions are usually combined to form a final rating for each alternative. The selection of aggregation function plays an important role in the accuracy of the final solution (Chen, 2005). An *aggregation operator* is a function $F: I^n \rightarrow J$ where I and J are real intervals and denote the set of values to be aggregated and the corresponding result of the aggregation, respectively (Smolíková & Wachowiak, 2002). Input vector I and output value J can be linguistic variables or numerical values. In the proposed algorithm, the ordered weighted averaging (OWA) operator is used to aggregate individual preferences. An OWA operator is an aggregation operator with an associated vector of weights $\sum_{i=1}^{n} w_i = 1, w \in [0,1]^n$ such that (Yager, 1988, 1993, 1994):

$$F_w(x) = \sum_{i=1}^{n} w_i . b_i \quad , \quad x \in I^n \tag{1}$$

with b_i denoting the ith largest element in $x_1, \ldots, x_n$.

The weighting vector is calculated as follow (Yager, 1993):

$$w_i = Q(\frac{i}{n}) - Q(\frac{i-1}{n}) \quad , \quad i = 1, \ldots, n \tag{2}$$

where Q is a linguistic quantifier that represents the concept of fuzzy majority and is used to calculate the weighting vector. A fuzzy linguistic quantifier may be defined as follows:

$$Q(r) = \begin{cases} 0 & \text{if}: \quad r < a \\ \dfrac{r\text{-}a}{b\text{-}a} & \text{if}: \quad b \le r \le a \\ 1 & \text{if}: \quad r > b \end{cases} \tag{3}$$

where (a, b) are the ranges of linguistic quantifier Q symbolically. The most common linguistic fuzzy quantifiers used are "*most*", "*at least half*", and "*as many as possible*". Their ranges are given as (0.3, 0.8), (0, 0.5) and (0.5, 1), respectively (Choudhurya *et al.*, 2005).

In addition, to evaluate the consensus measure on selected alternative(s), a concept of the maximum number of rounds, called MC (Maximum Cycle), is also incorporated in the consensus model to avoid the delayed convergence of collective solution after several rounds of discussion. In other words, the maximum cycle concept prevents endless rounds of discussion to reach the consensus (Bryson, 1996).

3 PROPOSED ALGORITHM

The generic structure of the proposed algorithm is presented in 15 steps, as follows (Fig. 2):

Step 1 Every expert in the group, expresses his/her evaluation on each alternative in different preference formats (ordinal preferences of the alternatives, fuzzy preference relation, multiplicative preference relation, utility function). In these assessments, ordinal preferences of alternatives are represented by O_s^i, which defines preference-ordering evaluation given by DM^i to alternative x_s. The fuzzy preference relation is expressed by k_{sm}^i where $k_{sm}{}^i \subset X * X$ with membership function $\mu_{ki} : X \times X \to [0,1]$, and $\mu_{k^i}(x_s, x_m) = k_{sm}^i$, where $X = \{x_1, \ldots, x_n\}$ is a finite set of alternatives. The value of k_{sm}^i defines a ratio of the fuzzy preference intensity of alternative x_s to x_m. Multiplicative preference relations are represented as A^i where $A^i \subset X * X$, $A^i = a_{sm}^i$ and a_{sm}^i is a ratio of the fuzzy preference intensity of alternative x_s to x_m given by DM^i which is scaled in a 1 to 9 scale (Saaty, 1980). The utility function is shown as U^i where DM^i explains his/her preferences on alternatives as n-tuple utility values. The utility value of alternative x_s given by DM^i is presented by $u_s^i \in [0,1]$.

Step 2 Information from Step 1 is transformed into fuzzy preference relationship by an appropriate transformation function. A common transformation between the various preferences is as follows (Chiclana *et al.*, 1998):

$$K_{sm}^i = \frac{(u_s^i)^2}{(u_s^i)^2 + (u_m^i)^2} \tag{4}$$

$$K_{sm}^i = \frac{1}{2}(1 + \frac{o_m^i - o_s^i}{n-1}) \tag{5}$$

$$K_{sm}^i = \frac{1}{2}(1 + \log_9 a_{sm}^i) \tag{6}$$

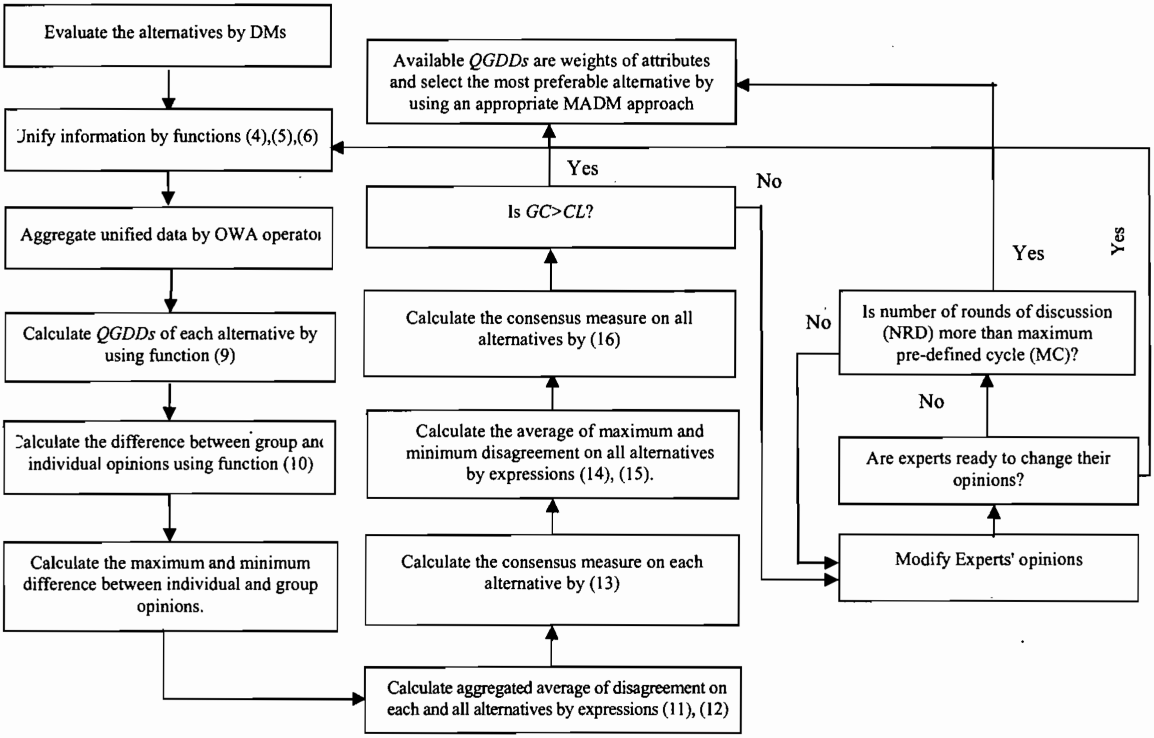

Fig. 2 Proposed algorithm for fuzzy group decision making.

Step 3 Proposed OWA operator to form collective preference relation aggregates for individual fuzzy relations:

$$k_{s,m} = \Phi_Q(k^1_{sm},\dots,k^m_{sm}) = \sum_{i=1}^{m} w_i . k^i_{sm} \tag{7}$$

where k_{sm} is a ratio of the fuzzy group preference of alternative x_s to x_m and Q is the fuzzy linguistic quantifier that represents the concept of fuzzy majority and is used to calculate the weighting vector. As mentioned before, a fuzzy linguistic quantifier is defined as follows:

$$Q(r) = \begin{cases} 0 & \text{if: } r < a \\ \dfrac{r-a}{b-a} & \text{if: } b \le r \le a \\ 1 & \text{if: } r > b \end{cases} \tag{8}$$

Step 4 Quantifier guided dominance degree $(QGDD)_i$ of the alternative x_i is calculated. This quantity calculates the dominance of the alternative x_i over other alternatives and collective alternative:

$$QGDD_i = P^q(x_i) = \Phi_Q(k^q{}_{i1},\dots,k^q{}_{in}) = \sum_{j=1}^{n} w_j . k^q{}_{ij} \tag{9}$$

where $P^q(x_i)$ defines the preference degree or intensity of alternative x_i over others alternatives given by DM^q.

Step 5 Difference between individual opinion and group opinion is calculated by using equation (10) as follows:

$$S_q(C_i) = \left|P^q(x_i) - P^g(x_i)\right|^b \tag{10}$$

where $b \in [0,1]$. Parameter b controls the rigorousness of the consensus process; $S_q(C_i)$ represents the degree of proximity of individual opinion to collective opinion. $P^q(x_i)$ and $P^g(x_i)$ represent the evaluation given by DM^q and collective evaluation to alternative x_i, respectively.

Step 6 Calculate $S(C_i)^{pis}$, $S(C_i)^{Nis}$ the minimum and maximum difference between individual opinions with collective opinion, respectively.

Step 7 Aggregated average of disagreement of all decision makers on each alternative x_i ($CM(C_i)$) and on all alternatives ($CM(C)$) is calculated as:

$$CM(C_i) = \Phi_Q(S_1(C_i),\ldots,S_m(C_i)) = \sum_{j=1}^{m} w_j.S_j(C_i) \tag{11}$$

$$CM(C) = \Phi_Q(CM(C_1),\ldots,CM(C_n)) = \sum_{i=1}^{n} w_i.CM(C_i) \tag{12}$$

Step 8 The consensus measure on each alternative is calculated by using the following expression:

$$GC(C_i) = 1 - \left|\frac{CM(C_i) - S(C_i)^{pis}}{S(C_i)^{pis} - S(C_i)^{Nis}}\right| \tag{13}$$

Step 9 The average of minimum and maximum disagreement of all decision makers on all alternatives ($GSCL(C)$), ($GWCL(C)$) are calculated by following functions, respectively.

$$GSCL(C) = \Phi_Q(S(C_1)^{pis},\ldots,S(C_n)^{pis}) = \sum_{i=1}^{n} w_i.S(C_i)^{pis} \tag{14}$$

$$GWCL(C) = \Phi_Q(S(C_1)^{Nis},\ldots,S(C_n)^{Nis}) = \sum_{i=1}^{n} w_i.S(C_i)^{Nis} \tag{15}$$

Step 10 The consensus measure on all alternatives (GC) is obtained as follow:

$$GC = 1 - \frac{GSCL(C) - CM(C)}{GSCL(C) - GWCL(C)} \tag{16}$$

Step 11 If GC > CL, the process is finished and the calculated $QGDD_s$ in Step 4 is assigned as relative importance of criteria, because $QGDD_i$ is the dominance of the criterion c_i over other criteria. These values are used as collective weights of attributes for selecting the best possible alternative. Otherwise, go to Step 12.

Step 12 In this step, all decision makers are asked to modify their opinions to reach consensus.

Step 13 If the number of rounds of discussion (NRD) is more than maximum pre-defined cycle (MC), stop the process of discussion and assign QGDD$_s$ to criteria. Otherwise, go to Step 14.

Step 14 After modifying the experts' opinions, all steps (from Step 2 to the end) are recalculated again and this process is continued until GC > CL or NRD > MC.

Step 15 According to the group weights obtained, and by using an appropriate multi attribute decision making (MADM) approach, the final and most preferable alternative is selected and is informed to all experts by the group manager.

4 MODEL APPLICATION

A multi objective problem in groundwater resource management was considered to investigate the application of the proposed algorithm. This problem was initially considered by Magnouni & Treichel (1992) and formulated by Duckstein *et al.* (1994) for three objectives of: (1) maximization of total pumping rates, (2) minimization of operating costs and (3) minimization of water-shortage risk by using multi criteria decision making (MCDM). Duckstein *et al.* (1994) solved this model and obtained the 13 Pareto solutions given in Table 1. The aim of this case study is to evaluate the weights of the criteria with respect to the DMs' opinions and then rank the 13 alternatives with respect to the calculated weights of criteria by the simple additive weighting (SAW) method.

Table 1 System *versus* Criteria Array (adopted from Duckstein *et al.*, 1994).

Alternatives	1	2	3	4	5	6	7	8	9	10	11	12	13
Pumped water	32.11	36.87	41.34	44.89	44.89	27.21	22.3	17.4	12.5	27.21	22.3	17.4	12.5
Cost	23.89	32.13	40.37	48.61	56.85	15.56	8.18	4.51	2.39	27.31	19.45	16.97	18.75
Risk	2.52	3.44	4.36	5.27	6.19	3.44	3.23	2.8	2.1	0.498	0.234	0.174	0.125

Five DMs were considered to evaluate the weights of three criteria. The DMs present their views on the various criteria in four different ways. The first DM presents his view in the form of utility functions, the second DM expresses his view as fuzzy preference relations, the third DM proposed his view in a multiplicative preference relation on a scale of 1 to 9 and the fourth DM presents his views as utility functions and the fifth DM remarks his view as ordinal preferences of the alternatives, as follows:

$$\begin{array}{l} DM^1\ [.53,\ .68,\ .3] \\ DM^4\ [.73,\ .62, .19] \\ DM^5\ [2\ ,\ 1\ ,\ 3] \end{array},\quad DM^2 \begin{array}{c|ccc} & 1 & 2 & 3 \\ \hline 1 & .5 & .57 & .77 \\ 2 & .43 & .5 & .86 \\ 3 & .23 & .14 & .5 \end{array},\quad DM^3 \begin{array}{c|ccc} & 1 & 2 & 3 \\ \hline 1 & 1 & 4/5 & 5 \\ 2 & 5/4 & 1 & 6 \\ 3 & 1/5 & 1/6 & 1 \end{array}$$

The various forms of the presented opinions are transformed into fuzzy preference relations by using the previously defined transformation functions:

$$DM^1 \begin{array}{c|ccc} & 1 & 2 & 3 \\ \hline 1 & .5 & .38 & .76 \\ 2 & .62 & .5 & .84 \\ 3 & .24 & .16 & .5 \end{array},\ DM^2 \begin{array}{c|ccc} & 1 & 2 & 3 \\ \hline 1 & .5 & .57 & .77 \\ 2 & .43 & .5 & .86 \\ 3 & .23 & .14 & .5 \end{array},\ DM^3 \begin{array}{c|ccc} & 1 & 2 & 3 \\ \hline 1 & .5 & .45 & .87 \\ 2 & .55 & .5 & .91 \\ 3 & .13 & .09 & .5 \end{array},\ DM^4 \begin{array}{c|ccc} & 1 & 2 & 3 \\ \hline 1 & .5 & .58 & .94 \\ 2 & .42 & .5 & .91 \\ 3 & .06 & .09 & .5 \end{array},\ DM^5 \begin{array}{c|ccc} & 1 & 2 & 3 \\ \hline 1 & .5 & .25 & .75 \\ 2 & .75 & .5 & 1. \\ 3 & .25 & 0 & .5 \end{array}$$

Transformed and unified values in the previous step are aggregated by using the OWA operator and aggregation weights in the aggregation step that resulted from quantifier "*most*" with the domain (.3, .8) are (0, 0.2, 0.4, 0.4, 0). The resulting collective fuzzy preference opinion is:

$$\text{Collective solution}= \begin{array}{c|ccc} & 1 & 2 & 3 \\ \hline 1 & .5 & .422 & .77 \\ 2 & .502 & .5 & .89 \\ 3 & .19 & .09 & .5 \end{array}$$

As mentioned, $QGDD_s$ of the individual homogenized and collective solutions are calculated by equation (9). The fuzzy linguistic quantifier "*as many as possible*" with domain (0.5, 1) is utilized. Hence, the corresponding weight vector with this operator is W = (0, 0.33, 0.67) and $QGDD_s$ for all solutions are:

$$DM^1 = [.42\quad .54\quad .19]\ ,\quad DM^2 = [.52\quad .45\quad .17]\ ,\quad DM^3 = [.47\quad .52\quad .1]$$

$$DM^4 = [.53\quad .45\quad .07]\ ,\quad DM^5 = [.3\quad .58\quad .08]\ ,\quad DM^G = [.45\quad .5\quad .123]$$

Calculate the aggregated average of disagreement on each alternative ($CM(C_i)$) by using the expressions (11) and fuzzy quantifier "*as many as possible*" and the corresponding weight vector w = (0, 0, 0.2, 0.4, 0.4) are presented in Table 2. The consensus measure on each alternative are also calculated by (13) and presented in Table 2. According to the values of Table 2 and functions (12), (14) and (15), the disagreement measure on all alternatives (CM(C)) and the average of minimum and maximum disagreement on all alternatives (GSCL(C)), (GWCL(C)) are calculated using the fuzzy quantifier "*as many as possible*" and are presented in Table 3. The consensus

Table 2 Consensus and disagreement measure on each criteria.

Variables	Criteria		
	1	2	3
$S(C_i)^{pis}$	0.02	0.02	0.023
$S(C_i)^{Nis}$	0.15	0.08	0.067
$CM(C_i)$	0.034	0.034	0.0358
$GC(C_i)$	0.89	0.77	0.71

Table 3 Consensus and disagreement measure on all alternatives.

CM(C)	GWCL(C)	GSCL(C)	GC
0.034	0.071	0.02	0.73

measure on all alternatives, using equation (16), was determined as 0.73. The acceptable consensus measure defined by the group manager is 0.8; therefore, the decision makers modify their opinions to reach a suitable consensus. The group manager ask decision makers to change their assessments. DM^4 and DM^5 agree to modify their opinions. First, DM^5 is picked and a collective solution (.45, .5, .123) offered to him. DM^5 accepts (0.4, 0.5, 0.12) for the relative importance of the alternatives. Then, DM^4 is picked and a collective solution is offered to him which he refuses to accept. He accepts DM^2's solution which is the closest solution to his own solution.

The collective solution and consensus measures are recalculated again. *GC* is now equal to 0.8, i.e. it reaches to the fixed level and secures the required condition to stop the aggregation process. The resulting collective solution in this round is considered as the final consensual solution. The collective relative importance of the attributes or group weighting vector is (.41, .44, .15). According to the weighting vector obtained and the values of the Table 1 and by using the SAW approach, alternative number 9 is obtained as the most preferred alternative for the region's water supply and all decision makers are informed. The final score and rank of each alternative is shown in Table 4.

Table 4 Final score and rank of each alternative.

Alternative	1	2	3	4	5	6	7	8	9	10	11	12	13
Final score of alternative	.0551	.0558	.0581	.0603	.0590	.0573	.0738	.1061	.1757	.0582	.0731	.0801	.0873
Rank of alternative	13	12	10	7	8	11	5	2	1	9	6	4	3

5 CONCLUSIONS

In this paper, a fuzzy group decision-making procedure was formulated to help the decision makers in selecting the most preferred alternative but considering the preferences of different decision makers. More efficient utilization of the available knowledge, creativeness and better understanding are considered as advantages of group decision-making (GDM) over multi criteria decision-making (MCDM) in alternative ranking. This paper adopted a distance measure method to evaluate the consensus measure. The 15-step fuzzy group decision-making algorithm was applied to a groundwater development problem with 13 different alternatives with respect to three criteria. Alternatives were evaluated based on three proposed criteria with five decision makers, Decision makers' opinions about the weights of criteria were transformed into fuzzy preference relations and aggregated using the OWA operator. Application of the proposed algorithm to rank 13 groundwater development alternatives showed that the decision makers might arrive at a conclusion with a reasonable and determinate consensus level if the proposed procedure is followed.

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Application of GA-based TOPSIS optimization schemes for dynamic control of reservoir water levels

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Abstract The issue of dynamically controlling reservoir water levels between safe limits, and according to preferred benefit and loss risks, is of great significance. This paper presents a method based on the TOPSIS reservoir water level dynamic optimization method for different water level optimization schemes, to establish an index system. Two basic TOPSIS functions are used and two types of the set of four models were established for multi-objective optimization on the basis of the "difference-driven" principles. Then we used the Genetic Algorithm (GA) approach to solve the optimization in a solution of the objective weights and obtained the sequence of the schemes at the same time. The method has been applied to the optimization scheme for scheduling of Biliuhe Reservoir in Dalian city in China. The results show that the method is reasonable and feasible.

Key words limited water level; dynamic control; TOPSIS; GA

INTRODUCTION

Dynamic water level control of a reservoir is necessary to determine the specific number of limited water levels in the foreseeable period, which must be between the upper and lower limits of the designed water level; according to the weather forecast information (including short- and medium-term trend forecasting), the rainfall–runoff forecasting information, the real-time regime and the situation and disaster information (Wang & Zhou, 2006). It belongs to real-time scheduling and is another more scientific and rational non-engineering measure of using flood resources than the method of raising the flood control level by steps and forecasting information. The optimization scheme for dynamic control of reservoir water levels is a process for choosing the optimal scheme among the flood control level dynamic control schemes. At present, multi-objective optimization methods use fuzzy comprehensive optimization (Chen, 1998; Yin *et al.*, 2001), grey comprehensive optimization, Artificial Neural Network optimization (Wu *et al.*, 2001), Analytic Hierarchy Process optimization (Yin *et al.*, 2001; Zhang *et al.*, 2002), and various improved or coupling methods (Zhang *et al.*, 2006). Because the optimization scheme involves a large number of evaluation indicators, which have different dimensionalities, and the weight of each indicator is difficult to identify, for practical applications these methods involve certain difficulties. This paper presents a method based on TOPSIS (Technique for Order Preference by Similarity to Ideal Solution) to determine the water level limit dynamically, using the Genetic Algorithm to optimize the TOPSIS function model, and applies this method for Dalian Biliuhe Reservoir Scheduling Optimization.

TOPSIS METHOD

The TOPSIS is a multiple attribute decision making (MADM) technique in which the alternatives are ranked by their distances between ideal and negative ideal solutions. The TOPSIS method was first developed by Hwang & Yoon (1981) and ranks the alternatives according to their distances from the ideal and the negative ideal solution, i.e. the best alternative has simultaneously the shortest distance from the ideal solution and the farthest distance from the negative ideal solution. In practice, TOPSIS has been successfully applied to solve selection/evaluation problems with a finite number of alternatives (Jee & Kang, 2000; Yong, 2006; Zhang *et al.*, 2007) because it is intuitive and easy to understand and implement.

Suppose a MADM problem has m alternatives, and n decision criteria. Each alternative is evaluated with respect to the n criteria. All the alternative performances, related to each criterion from a decision matrix, are denoted by $\boldsymbol{Y} = (y_{ij})_{m\times n}$. Then the procedure of the TOPSIS method consists of the following steps (Xu & Wu, 2006):

Step 1 Construct the normalized decision matrix $\boldsymbol{Z} = \{z_{ij}\}$. The various attribute dimensions are converted in this step into non-dimensional attributes, in order to allow comparisons across the attributes. An element of the normalized decision matrix is calculated as follows:

$$z_{ij} = y_{ij} / \sqrt{\sum_{i=1}^{m} y_{ij}^2}, \qquad (i = 1, 2, \cdots, m; j = 1, 2, \cdots, n) \tag{1}$$

Step 2 Construct the weighted normalized decision matrix. The Decision Maker assigns weights to each attribute. The weighted normalized decision matrix is constructed by multiplying each element z_{ij} with its associated weight w_j:

$$x_{ij} = w_j \cdot z_{ij}, \qquad (i = 1, 2, \cdots, m; j = 1, 2, \cdots, n) \tag{2}$$

Step 3 Determine the ideal and the negative ideal solutions. The ideal solution $\boldsymbol{x}^*$ and the negative ideal solution $\boldsymbol{x}^0$ are defined as $\boldsymbol{x}^*=\{x_1^*, x_2^*, \ldots, x_n^*\}$, $\boldsymbol{x}^0=\{x_1^0, x_2^0, \ldots, x_n^0\}$ where:

$$x_j^* = \left\{\left(\max_j x_{ij} \,\middle|\, j \in J\right), \left(\min_j x_{ij} \,\middle|\, j \in J'\right)\right\} \qquad j = 1, 2, \cdots, n \tag{3}$$

$$x_j^* = \left\{\left(\max_j x_{ij} \,\middle|\, j \in J'\right), \left(\min_j x_{ij} \,\middle|\, j \in J\right)\right\} \qquad j = 1, 2, \cdots, n \tag{4}$$

where J is the set of "benefit" attributes and J' is the set of "cost" attributes.

Step 4 Measure the separation of alternatives from the "ideal" solutions. The separation of each alternative from the ideal solution (d_i^*) is given as:

$$d_i^* = \sqrt{\sum_{j=1}^{n} (x_{ij} - x_j^*)^2}, \qquad (i = 1, 2, \cdots, m) \tag{5}$$

Similarly, the separation of each alternative from the negative ideal solution (d_i^0) is as follows:

$$d_i^0 = \sqrt{\sum_{j=1}^{n} (x_{ij} - x_j^0)^2}, \qquad (i = 1, 2, \cdots, m) \tag{6}$$

Step 5 Calculating the relative closeness to the ideal solution. The relative closeness to the ideal solution is defined as follows:

$$C_i = d_i^* / (d_i^* + d_i^0), \qquad (i = 1, 2, \cdots, m) \tag{7}$$

where C_i represents the relative closeness.

Step 6 Ranking the alternatives. Alternatives must be ranked based on C_i in which the lowest score is the best alternative.

TOPSIS value function

It is very important to determine the index weight to evaluate the priority of schemes using the TOPSIS method. At present, general methods to determine the weight have subjective, objective and combined weight methods. These studies play an important role in advancing application of the TOPSIS method in different areas and have become a research hotspot. Generally speaking, the study is relatively less on the essence of TOPSIS functions and the mechanism of the different effects on the TOPSIS method by different TOPSIS functions and different weights. The TOPSIS function is nonlinear, and it is very complicated to determine the objective weights by routine method. But if the TOPSIS function is simplified (Yu & Liang, 2003), it can not reflect the thought of TOPSIS fully and will bring large errors to evaluation results.

The Utility Function Theory points out that the value function should be able to reflect the preferences of policy makers more comprehensively in the multi-criteria decision-making and it is easy to receive the satisfying result for decision-making if the value function is determined. Six commonly used functions of the TOPSIS models are summarized and defined in the paper of Qin (2003). By analysing and comparing the value functions of these commonly used functions of the TOPSIS models, they are compared to two categories of traditional value function model. One kind of non-difference surface is the hyperplane family which is orthogonal with vector v^+:

$$f_1 = \langle v, v^+ \rangle / \|v^+\|^2 \tag{8}$$

And the other is family of hypersphere:

$$f_2 = \|v\| / (\|v\| + \|v^+ - v\|) \tag{9}$$

where <·, ·> represents vector inner product; $\| \ \|$ represents Euclidean vector norm; v^+ is the ideal solution; and v is the sample points.

APPLICATION OF TOPSIS TO OPTIMIZATION SCHEMES OF DYNAMIC CONTROL OF RESERVOIR WATER LEVELS

For dynamically reservoir water level control, because the objective function is complex and nonlinear, it is not easy to determine the comprehensive index weight using routine methods. The Genetic Algorithm, a mechanism is to simulate the principles of survival of the fittest and chromosome information exchange, is a general-purpose global optimization method, which can solve such problems simply and effectively (Jin & Ding, 2002). So, two types of the four-sets models were established for multi-objective optimization on the basis of the "difference-driven" principles (Guo, 2007), then we used genetic algorithms to solve the optimization in a solution of the objective weights, and to obtain the sequence of the schemes at the same time.

Optimization problem 1 With the target of minimum sum all of C_i, determine the objective weights coefficient vector $w = (w_1, w_2, \ldots, w_n)$.

If TOPSIS function is f_1, optimization Model I is established as follows:

$$\min \sum_{i=1}^{m} C_i = \sum_{i=1}^{m} \frac{\langle x^* - x^0, x^* - x_i \rangle}{\|x^* - x^0\|^2} = \sum_{i=1}^{m} \frac{\sum_{j=1}^{n} [(x_j^* - x_j^0)\ (x_j^* - x_{ij})]}{\sum_{j=1}^{n} (x_j^* - x_j^0)^2} = \sum_{i=1}^{m} \frac{\sum_{j=1}^{n} w_j^2 [(z_j^* - z_j^0)\ (z_j^* - z_{ij})]}{\sum_{j=1}^{n} w_j^2 (z_j^* - z_j^0)^2} \tag{10}$$

s.t. $w_1 + w_2 + \ldots + w_n = 1$; $w_i > 0$, $i = 1,2,\ldots,m$

If TOPSIS function is f_2, optimization Model II is established as follows:

$$\min \sum_{i=1}^{m} C_i = \sum_{i=1}^{m} \frac{d_i^*}{d_i^0 + d_i^*} = \sum_{i=1}^{m} \frac{\sqrt{\sum_{j=1}^{n} (x_{ij} - x_j^*)^2}}{\sqrt{\sum_{j=1}^{n} (x_{ij} - x_j^0)^2} + \sqrt{\sum_{j=1}^{n} (x_{ij} - x_j^*)^2}} = \sum_{i=1}^{m} \frac{\sqrt{\sum_{j=1}^{n} w_j^2 (z_{ij} - z_j^*)^2}}{\sqrt{\sum_{j=1}^{n} w_j^2 (z_{ij} - z_j^0)^2} + \sqrt{\sum_{j=1}^{n} w_j^2 (z_{ij} - z_j^*)^2}} \tag{11}$$

s.t. $w_1 + w_2 + \ldots + w_n = 1$; $w_i > 0$, $i = 1,2,\ldots,m$

Optimization problem 2 According to the idea of projection pursuit (Friendman & Turkey, 1974; Jin & Ding, 2002), an evaluation model is established, and the suitable objective weight coefficient vector $w = (w_1, w_2, \ldots, w_n)$ can be obtained. Gather the schemes into some categories, in which the schemes are intensive in the same category and scattered in different categories as far as possible. Then the objective function is:

$$\max Q(w) = S_c D_c, \tag{12}$$

s.t. $w_1 + w_2 + \ldots + w_n = 1$; $w_i > 0$, $i = 1,2,\ldots,m$

where S_c is the standard deviation of each scheme evaluation value C_i; D_c is local density of each scheme evaluation value C_i; S_c and D_c are defined as:

$$S_c = \{\sum_{i=1}^{n}[C_i - \overline{C}]^2/(n-1)\}^{0.5} \qquad D_c = \sum_{i=1}^{n}\sum_{j=1}^{n}(R - r_{ij})u(R - r_{ij})$$

where $\overline{C}$ is mean of sequence $\{C_i|i = 1 - n\}$; R is radius of the window for local density, usually $R = 0.1S_z$; distance $r_{ij} = |C_i - C_j|$; the function value of $u(t)$ is 1, if $t \geq 0$, and else it is 0, if $t < 0$.

If TOPSIS function is f_1, optimization Model III is established. The expression of C_i as follows:

$$C_i = \frac{\sum_{j=1}^{n} w_j^2[(z_j^* - z_j^0)\cdot(z_j^* - z_{ij})]}{\sqrt{\sum_{j=1}^{n} w_j^2(z_j^* - z_j^0)^2}} \tag{13}$$

If TOPSIS function is f_2, optimization Model IV is established. The expression of C_i is:

$$C_i = \frac{\sqrt{\sum_{j=1}^{n} w_j^2(z_{ij} - z_j^*)^2}}{\sqrt{\sum_{j=1}^{n} w_j^2(z_{ij} - z_j^0)^2} + \sqrt{\sum_{j=1}^{n} w_j^2(z_{ij} - z_j^*)^2}} \tag{14}$$

Obviously, models I, II, III and IV are complex and not nonlinear optimization problems; they are not easy to determine by conventional methods, so we use GA to determine these problems.

Eight situations for dynamic water level control of Biliuhe Reservoir in the Dalian city were used for study (Table 1) (Wang & Zhou, 2006). The higher the economic benefits and smaller the risk, the better. So there are two types of indices: benefit type and risk type: C_1: flood resource utilization, C_2: annual generated energy, and C_3, guarantee rate reliability, are benefit type indexes; C_4, risk rate, and C_5, comprehensive risk loss rate, are risk type indexes.

First, the indices of each scheme have to deal with the standardization, criteria matrix as follows:

$$Z = \begin{bmatrix} 0.000 & 0.000 & 0.000 & 1.000 & 1.000 \\ 0.500 & 0.156 & 0.483 & 1.000 & 1.000 \\ 0.500 & 0.304 & 0.534 & 1.000 & 1.000 \\ 0.750 & 0.449 & 0.621 & 0.750 & 0.824 \\ 0.750 & 0.593 & 0.690 & 0.500 & 0.647 \\ 0.875 & 0.628 & 0.741 & 0.000 & 0.206 \\ 0.875 & 0.720 & 0.776 & 0.000 & 0.118 \\ 1.000 & 1.000 & 1.000 & 0.000 & 0.000 \end{bmatrix}$$

Table 1 Index eigenvalues of scheme sets assessment of dynamic limited water level of Biliuhe Reservoir.

S	L(m)	B			R	
		C_1 (%)	C_2 (10^6kW·h)	C_3 (%)	C_4 (%)	C_5 (%)
1	68.1	52.13	11.5103	97.97	0.010	60.73
2	68.2	52.17	11.5503	98.25	0.010	60.73
3	68.3	52.17	11.5883	98.28	0.010	60.73
4	68.4	52.19	11.6254	98.33	0.011	60.85
5	68.5	52.19	11.6622	98.37	0.012	60.97
6	68.6	52.20	11.6712	98.40	0.014	61.27
7	68.7	52.20	11.6948	98.42	0.014	61.33
8	68.8	52.21	11.7665	98.55	0.014	61.41

S: scheme number; L: limited water level; B: benefit type indexes; R: risk type indexes; C_1: flood resource utilization; C_2: annual generated energy; C_3: guarantee rate reliability; C_4: risk rate; C_5: comprehensive risk loss rate.

Secondly, models I, II, III and IV are solved by GA to obtain their optimal weights, as shown in Table 2. Finally, the weights are put into functions f_1 and f_2, to obtain the evaluation results, as shown in Table 3.

Table 2 Objective weights of each model.

Model	Objective weight: C_1	C_2	C_3	C_4	C_5
Model I	0.9066	0.0152	0.0564	0.0122	0.0097
Model II	0.8413	0.0168	0.1231	0.0094	0.0094
Model III	0.0165	0.0006	0.0548	0.8171	0.1110
Model IV	0.0151	0.0007	0.0442	0.7937	0.1463

Table 3 The comparison and ranking of scheme set of four models.

S	Model I		Model II		Model III		Model IV	
	A	Ra	B	Ra	A	Ra	B	Ra
1	0.9997	8	0.9846	8	0.0048	3	0.0547	3
2	0.5000	7	0.5004	7	0.0025	2	0.0290	2
3	0.4998	6	0.4992	6	0.0022	1	0.0265	1
4	0.2506	5	0.2533	5	0.2492	4	0.2483	4
5	0.2503	4	0.2515	4	0.4964	5	0.4945	5
6	0.1258	3	0.1299	3	0.9927	6	0.9453	6
7	0.1257	2	0.1286	2	0.9941	7	0.9519	7
8	0.0003	1	0.0154	1	0.9952	8	0.9453	8

S: scheme number; A: f_1 evaluation value; B: f_2 evaluation value; Ra: ranking.

Table 3 shows the evaluation results of different fixed weight methods of two types of TOPSIS functions. The difference of weights obtained by Model I and Model II are not great, so the order is the same, and scheme 8 is the optimum. Scheme 3 is the optimum one with models III and IV. The establishing principle of models I and II is different from that of models III and IV, so the weights obtained by models I and II and models III and IV are different, and so the differences of the evaluation results are great. This shows that, with the same weight, different TOPSIS functions have certain influences on the evaluation, though it is not very clear. The same TOPSIS function, using different principle modelling obtained from the weight, will have greater affect on the evaluation. Therefore, compared with the TOPSIS function selection, how to select appropriate weights is more critical, when using the TOPSIS method to evaluate, because it obviously affects the evaluation results. Overall, scheme 8 or scheme 3 is the optimal scheme in this paper.

CONCLUSIONS

The reservoir flood control level reasonably affects not only the risk of the reservoir itself and its downstream, but also the reservoir water storage at the end of flood season and the benefit of flood protection. According to a number of alternative scheduling schemes, this study presents: (1) a method for dynamic optimization of water levels of based on TOPSIS, from the perspective of the flood safety, flood resources and enhance economic benefit, (2) use a Genetic Algorithm to optimize the solution, resolve the problem that the TOPSIS function need to be simplified by usual method, embody the basic principles of TOPSIS method completely, (3) set up four different models by two types of typical TOPSIS functions and two principles, and applied these to optimize scheduling of Biliu River Basin in the Dalian city in China. The results showed that this method enriches the theory and practice of reservoir dynamic water level control and can be widely applied.

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7 Integrating Water Resources Management

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Managing the competing water demands from off-stream and in-stream users—a conceptual framework

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Abstract Satisfying the demands of freshwater for growing population, industrialization and urbanization appears to be an enormous challenge to continuing to run the economic wheel without undermining the ecosystem needs. Environmental water allocation is widely acknowledged; however, it is implemented in few countries mainly in developed regions, even though dependency on flow is more pronounced in developing countries. Environmental water requirements are often ignored in traditional water allocation decision-making in these countries. Available environmental flow assessment methodologies focus more on conservation and restoration of the aquatic ecosystems, rather than taking the economic value of water into consideration. However, it is observed that economic aspects play a critical role in the implementation of environmental flow. A conceptual model for water allocation based on marginal benefits from sectoral water-use is proposed in this study to ensure environmental water allocation. An economic–hydrological model is conceptualized in which the economic return from all water uses is to be correlated with river discharge. The derived marginal benefit functions are targeted for both off-stream and direct in-stream uses through analysis of the economic return from off-stream extractions and the variation of in-stream use benefit in terms of flow level fluctuation. Two marginal benefit functions are proposed for the optimization analysis by maximizing consumer surplus and minimizing demand deficit. Such analyses will explore the extent of trade-offs associated with establishing environmental water allocation and subsequently will help in achieving environmental and economic sustainability by having equity between all users.

Key words environmental flow; in-stream use; off-stream use; marginal benefit; optimal allocation

INTRODUCTION

Water, the "driver" of life, determines the pattern of the ecosystem and supports the livelihood of large parts of the population, especially poor people in developing countries. The ever-growing demand for freshwater, that is increasing at a tremendous pace, results in scarcity of water in both quantity and quality and becomes a major development challenge in several parts of the world. Agriculture is responsible for the largest share of freshwater withdrawal worldwide, and this share is observed as being excessively high in low- to middle-income countries. Food production alone requires 3000 litres of water a day for an individual (Molden, 2007). Even though such a figure varies across regions and food habits, it indicates the extent of agricultural water demand. Agriculture and food production not only supports individuals but also plays a central role in the whole economy of a country, especially in developing regions. Agricultural water demand is expected to be doubled by 2050 to feed the growing population (Falkenmark & Galaz, 2007), and, in most cases, environmental water scarcity is due to higher agricultural demand, even though supply to irrigation itself is under real threat in several places, without considering the environment. Hence, it appears to be an enormous challenge to supply freshwater for agriculture to secure food production without undermining ecosystem needs.

The concept of environmental flow (EF), as first appeared in the scientific literature around the late 1940s in the USA, was a single-objective based principle, such as ensuring salmon/trout numbers for recreational fishermen. However, preserving the habitat for the target species needs due consideration of the whole ecosystem process. River water is a resource for multiple uses and the source of several goods and services on which human life depends. In particular, this dependency is high in poor and developing countries. Therefore EF assessment methodologies require a shift from the objective-based, prescriptive approach to an interactive approach for establishing a relationship between river and riverine system (Tharme, 2003). This relationship may then be used to describe environmental and socio-economic implications for various flow scenarios. However, the water requirement of the natural environment is rarely considered and almost ignored in traditional water management and planning, in particular in developing countries. In addition, ecosystem services are given less weight in policy decision making, due to the inability to fully capture the service value for the commercial market (Costanza *et al.*, 1997). In contrast, out-of-stream demands, mainly the irrigation sector, receive institutional and political efforts and support to improve their overall social benefit, even though the irrigation sector has poor efficiency with minimal application of modern technologies (Gleick, 2003; IWMI, 2005). Hence, it is necessary to see whether the social marginal value of off-stream water uses, especially irrigation, differs from that of in-stream uses, and to establish the extent of the differences, with the aim to facilitate the exploration of trade-offs in water allocation options.

Development of a policy to implement environmental flow is still in its infancy in most of the world, particularly in developing countries (Tharme, 2003; Moore, 2004), even though the requirement has long been acknowledged. The main reason behind this is a lack of understanding of the socio-economic benefits and costs involved in environmental water allocation, together with management, legal and institutional constraints, explored through a primary survey of water professionals from both developed and developing countries (Moore, 2004; Scatena, 2004). Insufficient hydrological data and poor public awareness are also cited as impediments to EF implementation. However, another survey (Syme *et al.*, 1999) shows that there is high acceptance across the community of allocating water for the environment. Many water professionals (88% of those surveyed) believed that implementation of EF is a necessary part of solving water scarcity related problems, though the situation is tempered when it comes to the question of trade-offs (Syme *et al.*, 1999). Moreover, in places where EF comes to be implemented, particular tensions are observed (Schofield & Burt, 2003). Successful implementation of EF faces challenges around the world due to uncertainty in both the scientific and economic approaches involved (Gleick *et al.*, 2006). The scientific community has failed to demonstrate clearly the benefits of allocating water for the environment (Schofield & Burt, 2003). It is therefore important to realize the societal cost and benefit of EF to all stakeholders, and this concept is reinforced by analysing the reasons behind successful implementation of EF in some countries where the importance of flows to local livelihoods is given due regard (Moore 2004).

Accurate and acceptable valuation, not only for intrinsic needs, but also for the goods and services that freshwater provides, plays important role in informed decision making for management and for protecting the livelihood of the poor (De Groot *et al.*, 2006). Valuation also helps in comprehending necessary trade-offs in the decision-making process (Farber *et al.*, 2002). Several cases and studies are found in the literature, although not many, in which environmentalists have tried to evaluate the in-stream water and associated ecosystem at times and locations. Examples include: use of the Contingent Valuation Method (CVM) in Montana, USA by Duffield *et al.* (1994); application of CVM and the Travel Cost Method (TCM) in California, USA by Douglas & Taylor (1998); CVM conducted in China by Xu *et al.* (2003); use of TCM in California, USA by Webber & Berrens (2006); and application of the CVM in Mexico by Ojeda *et al.* (2008). These studies, mainly focusing on recreational uses, are mostly performed in developed regions where availability of data and public awareness are comparatively higher than in developing countries. In contrast, dependency of the local people on flow is higher in developing countries, where realization for environmental health with sustainability and recreational uses do not get significant weight. Nevertheless, all these past studies of environmental valuation are aiming at finding the total value, rather than marginal value (MV) – implicitly indicating the change in total value due to change in resource input – which is the best parameter in water allocation decision making that concerns analysis of trade-offs for the management perspective (Gleick *et al.*, 2006; Smakhtin *et al.*, 2006; Moran & Dann, 2008).

Water allocation nowadays noticeably asks for a compromise between social, environmental and economic preferences, where the ecosystem protection objective needs to be satisfied. Insight into economic return and marginal value of the resource from all sectors can help decision makers to decide on better trade-offs (Smakhtin *et al.*, 2006). A substantial amount of work is being performed on water resources-related optimization problems, especially reservoir operation and maximization of benefits or minimization of water shortage in a water resources system, in which a number of cases treated a fixed amount of flow for environment as one of the constraints. However, these do not reflect the natural variability of flow. Recent work by Suen & Eheart (2006) and Shiau & Wu (2007) included EF as an objective function in a multi-objective optimization problem and presented the trade-offs scenario in supply; however, they have not calculated the economic trade-offs scenario.

A CONCEPTUAL FRAMEWORK FOR WATER ALLOCATION

Part of this new perspective on water resources management, the optimum allocation between off-stream and in-stream uses, is considered to be highly necessary with regard to scarcity and significance of the resource in the overall development and economy. A conceptual framework is proposed taking the marginal value produced from sectoral water use as a tool for allocation. A developing country perspective is considered to test the model, where the socio-economic conditions are very poor, the institutional structure is not well-developed and water rights are not properly defined. The proposed conceptual framework is represented in Fig. 1. In the proposed model, all water demands are grouped into two categories, namely off-stream and in-stream. Increased population, urbanization and industrialization generate pressure on off-stream demands, whereas increased concern about environmental sustainability along with poverty reduction give stress to in-stream demands. Water sources, as well as demand and supply, get pressurized from external factors such as climate change, floods, droughts, along with government policy and management. Satisfying both the off-stream and in-stream sides is often a contentious issue, and the question of sustainable allocation appears where lack of data and awareness along with people's reliance on in-stream water for their livelihoods are considered simultaneously. The allocation criterion is selected as the marginal benefit generated by different water uses. Development of a hydrological–economic model is considered, depicting the total benefit function of different water uses, as a correlation between water-use benefit and river discharge. The marginal benefit function derived as the first-order derivative of this total benefit function is considered for optimization of water allocation.

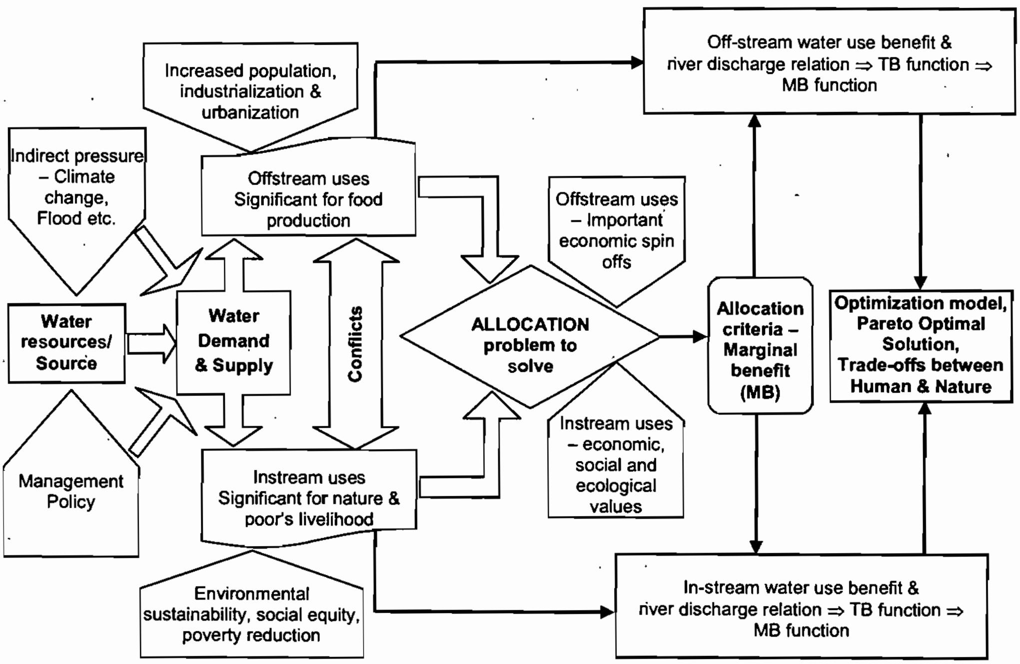

Fig. 1 Conceptual framework for optimum water allocation; TB: total benefit; MB: marginal benefit.

METHODOLOGY

The study targets an in-depth assessment of the functions, goods and services coming from in-stream flow by using an extensive and semi-structured primary survey on the riparian inhabitants. The focus is on subsistence and commercial fishermen, as well as boatmen, whose incomes are fully dependent on river flow. Using primary survey information, the income variation with the seasonal flow variation is depicted. In addition to using primary data, the available time series information on in-stream goods/products from secondary sources is collected and correlated with historic discharge. The study assesses the existing off-stream uses with their economic valuation. From these data and information, a hydrological–economic model is developed for water use, showing the relation between flow and net benefit, by adopting different approaches of economic analysis, in particular: value-added and residual imputation (for agricultural and fishery use); direct pricing and two-point expansion (for industrial & domestic use); cost of alternative source (for agricultural, industrial, domestic and navigational use); and contingent valuation for the non-marketed goods (e.g. bathing, washing, livestock bathing, recreational use). The marginal benefit function is derived using the pre-established flow-benefit function with its first derivatives. Finally, two aggregated marginal benefit functions for off-stream and in-stream uses are derived by adding vertically the non-competitive in-stream demands and horizontally the competitive off-stream demands (Fig. 2). The two aggregated marginal benefit functions are to be analysed further for optimization to maximize the overall benefit and trade-offs analysis, where maximization of consumer surplus (equation (1)) and user satisfaction (minimizing demand deficit) (equation (2)) are taken as the objective function.

The objective function for maximization of benefit/economic return is expressed as:

$$\text{maximize } Z\ \{C_{s\text{-OS}}, C_{s\text{-IS}}\} \tag{1}$$

where $C_{s\text{-OS}}$ and $C_{s\text{-IS}}$ are the consumer surplus for the off-stream and the in-stream sector, respectively.

For the case of user satisfaction, demand deficits are minimized and the objective function is:

$$\text{minimize } Z\ \{D_{d\text{-OS}}, D_{d\text{-IS}}\} \tag{2}$$

where $D_{d\text{-OS}}$ and $D_{d\text{-IS}}$ are the demand deficit for off-stream and in-stream use, respectively. Necessary constraints, e.g. reservoir operation rule, environmental flow requirement and water balance, are to be incorporated in the model. The solution gives the Pareto optimal scenario of water allocation.

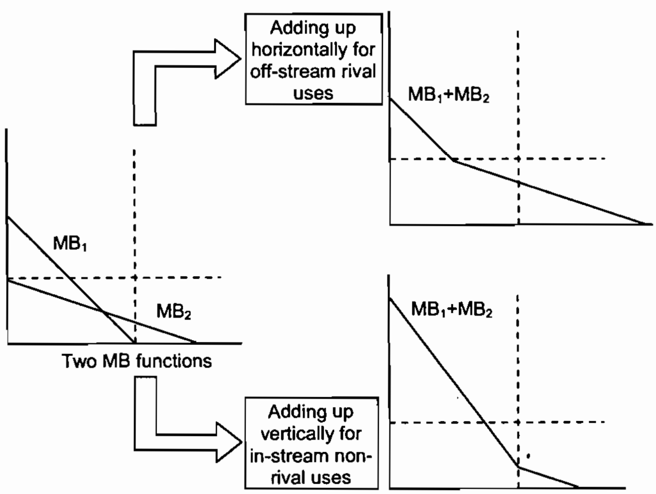

Fig. 2 Developing the marginal benefit function of water uses.

CASE STUDY FROM BANGLADESH—A DEVELOPING COUNTRY PERSPECTIVE

Water resources management in Bangladesh

In addition to contributing to irrigation water supply, rivers in Bangladesh are extremely important for navigation, the most economical mode of transport, and for fisheries. Fisheries have an important role in the economy, maintaining food security and public health through contributing the major share of protein in the daily diet. Fisheries contribute 6% of the country's GDP and provide two-thirds of animal protein (World Bank, 2006). The transport sector accounts for 8% of GDP and water transport shares 15% of that (World Bank, 2005). River flow is also important to prevent saline intrusion, especially in the southern part of the country. Water demand is expected to grow rapidly to feed the increasing population (annual growth rate of 1.43% measured in 2003; BBS, 2005). Rice production for every individual in Bangladesh requires more than 800 m^3 of water in a year (Rashid & Kabir, 1998), and the population is estimated as 220 million by 2050. To cater for this huge population, more than 20 million tonnes of extra food production is required, which in turn will demand a substantial amount of freshwater. Adding to that, fish demand will also be higher to fill the protein gap. Projected fish demand for 2025 is 5.5–8 million tonnes per year (World Bank, 2005). Since the 1990s, the water resources management in Bangladesh has mainly been focused on managing excess water (flood), rather than water shortage. However, with growing interest and explicit recognition of environmental demands in the country's National Water Policy and Water Management Plan, a review of the water management system is imperative. Although on an annual basis the rivers in Bangladesh seem to have plenty of water, they suffer from low flows in the dry season (winter and summer) and this situation is being exacerbated by human interference such as land-use change, deforestation, and more abstraction in the upper catchments. As a whole, water resources plays a key role in the country's economy and both off-stream and in-stream uses are extremely important. Therefore, effective planning and efficient use of water resources of the rivers is demanded, with a comprehensive assessment of the in-stream water requirement and its valuation.

The study site and challenges faced in realizing the conceptual framework

The Teesta, the fourth main river of Bangladesh, has been chosen as a case study for this research. Originating from Sikim, India, the Teesta enters Bangladesh at Chatnai, in Nilphamari District, and finally flows into the Brahmaputra. The total length of the Teesta is 315 km, of which 115 km is inside Bangladesh. It is a sandy, braided river with steep slope and dynamic nature. The Teesta flow has been regulated since 1987: two irrigation barrages have been constructed, by India and Bangladesh, in 1987 and 1990, respectively. The Teesta is the main source of water in northwest Bangladesh, a drought-prone area, but with significant agricultural potential. The barrage on the Bangladesh side has a capacity to supply water to 5400 km^2 of agricultural land and about 70 000 farmers are directly dependent on this water supply. However, the flow has reduced drastically in recent years, resulting in an alarming situation for agriculture, as well as for in-stream users such as fishermen and boatmen downstream of the barrage. Overall, the socio-economic conditions in the region are extremely poor. Agriculture is central to the economy and the main occupations of the inhabitants are farming, labour selling, fishing and rickshaw pulling. The literacy rate is only about 40% (BBS, 2005), concentrated in children and young adults. The banks of the River Teesta are completely rural, with a number of people engaged in fishing and informal commercial navigation as their main livelihoods. The average catch size of fishery in the Teesta is estimated as 250 tonnes/year, with an annual value of around 30 million taka (≈ US$0.5 million).

Developing countries are in general data poor, and this was an obvious issue at the Teesta site. Moreover, several challenges and impediments were faced regarding data collection and survey work at field level. The only off-stream demand at the Teesta site is the abstraction for irrigation; however, the amount of water diverted towards the irrigation canal was not disclosed by the respective authority. Irrigation water requirement, calculated using the CROPWAT model with consideration of an overall efficiency, was used as a proxy for irrigation water supply. On the basis of a conversation with the responsible officer at the barrage operation site, in dry months when flow falls below the irrigation demand, 90% of the discharge is considered as irrigation water diversion.

A semi-structured primary survey was conducted along both banks of the river. To depict the in-stream benefit, the study targeted the in-stream users, namely fishermen, both commercial and subsistence-based, and boatmen. With a contingent valuation approach, at the beginning the questionnaire focused on asking the willingness to pay (WTP) for having a satisfactory level of water flow through out the year. However, this failed: all the respondents were completely scared away by the question of payment, although the hypothetical scenario was clearly delivered. Mistrust and disbelief due to poor literacy and education are the main reasons for this. About 70% of the respondents did not have any educational background and the rest had only a primary level of education. Afterwards, the questionnaire focus was shifted to willing to accept (WTA) compensation, and in that case almost 100% of respondents gave WTA compensation on a daily basis equal to their daily income. Finally, daily income from the fishermen and boatmen is taken as the benefit of in-stream water use. The daily income level varies widely with flow variation across the seasons of a year. Seasonal daily income level is documented and correlated with flow. The marginal benefit function is derived from this function.

CONCLUSION

The conceptual model presented here is the first attempt at solving a water allocation problem based on the marginal value produced from water use with insight into trade-offs between humans and nature. This model will enable construction of numerical models based on the articulated relationships shown. Development of the numerical model derived from this concept is underway for the Teesta River basin of Bangladesh, where numerous challenges and new experiences are encountered and shared through this article. Sharing those experiences will help guide researchers in their work in other basins with similar socio-economic conditions, particularly in developing countries. Application of the outcome from the foreseen numerical model to the field is likely to contribute to environmental sustainability and equity between all users and will promote informed debate concerning the achievement of sustainable development.

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On the relationship between water demand management and integrated water resources management in China

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Abstract There is wide recognition that strategies for sustainable water use are needed due to great water scarcity stresses imposed by both uncertain climatic conditions, and continuous rapid economic growth. In China, water demand management (WDM) is thought to be a valid approach and is expected to play a large role in the management. Firstly this paper introduces the types of WDM methods, and points out that WDM methods should vary with different local situations, then it analyses the relationship between WDM and water supply management, water use management, and water quality management. By discussing the interactions of these parts, it emphasizes that WDM is part of Integrated Water Resources Management (IWRM) and IWRM will be a valid way to ease the conflict of water demand and water supply.

Key words water demand management; integrated water resources management; water-saving measures; sustainable water use

BACKGROUND

Although supply-side alternatives have historically been the preference in China, the challenge of water resources scarcity is greater due to the stresses imposed by uncertain climatic conditions and continuous rapid economic growth. There is wide recognition that the strategies for the sustainable use of water resources are in need. With the successful implementation of Demand Side Management in the electricity industry, which eases the conflict of energy demand and supply, the managers and planners are slowly moving towards Water Demand Management (WDM), and this approach is expected to be a valid measure to solve the problem. However, a common obstacle when dealing with concepts like water use, water efficiency, water demand management, etc. is that these terms tend to be general, misused, and occasionally misleading. For instance, in some authorities it is thought that WDM just means the application of water-saving technologies. With the aim of distinguishing WDM and Integrated Water Resources Management (IWRM), this paper provides the actual meaning of WDM, and analyses the relationship between WDM and IWRM. Most importantly, it points out that WDM is one part of IWRM, and different approaches of IWRM should be adopted according to the actual water problems.

WATER DEMAND MANAGEMENT

Definition of WDM

The idea of WDM comes from the energy industry management and it has been given different definitions. Savenjie & Van der Zaag (2002) thought it a development and implementation of strategies aimed at influencing demand, so as to achieve efficient and sustainable use of a scarce resource. Brooks (2006) gave WDM a new operational definition, and implied the goal of saving water, or saving higher quality water. Generally the concept of WDM refers to initiatives, which have the object of satisfying the existing needs for water with a smaller amount of available resources, and it includes the implementation of policies or measures to increase the efficiency of water use and reduce wastewater generation. Water demand management has become an important issue in Europe, and a number of polices and mechanisms are being used or being formulated to ensure the sustainable use of water.

Approaches of WDM

To encourage water conservation and efficient water use within the users, a variety of means must be implanted, in which structural, economic and socio-political programmes are comprised. Examples of types of measures are given in Table 1.

Table 1 Measures of water demand management.

Means of WDM	Measures
Structural means	Metering Installation of water efficient devices Rainwater harvesting Water recycling and reuse Pressure control devices
Economic means	Adequate water pricing and billing Adoption of economic incentives
Socio-political means	Policies National legislation, local and regional code Information, education Public attention

Information and educational approaches are always necessary, which are helpful to encourage more rational water use and change habits. Among the types of measures of WDM, reasonable water price, water efficient technologies and water reuse are highlighted recently when implanting the WDM programme. Taking water use of Beijing as example, the paper analyses the effect of these three measures.

(1) Technological approaches

Different experiences show that water savings can be achieved by using various water saving devices in households, public places, industries and agriculture. In the countryside, water saving can be achieved through the change of irrigation mode, while the water saving potential in urban areas relies on the application of water-saving devices by users.

Water demand patterns are changing with higher living standards and the applications of new technologies. For instance, in Beijing, the total water use was 3.58 billion m^3 in 2003, while it was only 3.48 million m^3 in 2007, 100 million m^3 less than that in 2003, and this is mainly owing to the adoption of water-saving devices. Water use in 2007 is apportioned as follows: 39% for domestic use, 8% for environment conservation, 17% for industry, and 36% for agriculture. With the improvement of living standards, there is an obvious change in water consumption. Figure 1 shows

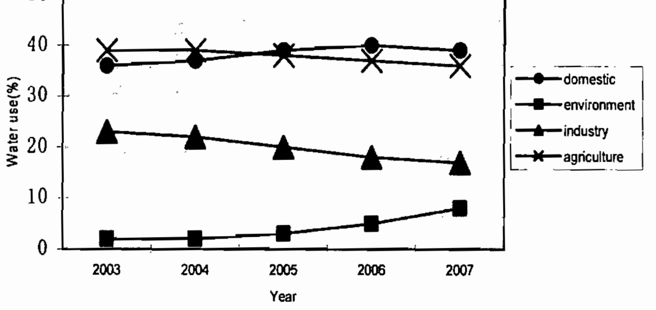

Fig. 1 Change of water use from 2003 to 2007 in Beijing.

Table 2 Typical water use devices of households in Beijing.

Equipment	WC (*L*)	R_{du} (%)	WP (10^4 m^3)	R_{ws} (%)
Tap			3133.3	35
Shower			2693.3	18
With water saving device	7 L/min	45	0	
Without water saving device	>12 L/min	35	2693.3	
Toilets			6368.8	46
One-command toilets	6–9 L/flush	57	2179.4	
Two-command toilets	<6 L/flush	8	3078.7	
Toilets with air devices	1–3 L/flush	0	1110.7	

WC, water consumption per unit with the equipment; R_{du}, popularization ratio of the water use device; WP, potential water saving; R_{ws}, potential water saving rate.

Table 3 Water saving potential for Beijing in 2010.

Water saving patterns	Water demand without application of water-saving (10^4 m^3)	Water saving potential (10^4 m^3)	Water saving ratio (%)
Household	37 587	12 909	34.3
Public uses and services	80 163	26 058	32.5
Industry (electricity)	21 000	11 470	54.6
Water re-use	–	48 000	–

that domestic and environmental water use are increasing slowly, while industry and agriculture use are decreasing. However, there is much water-saving potential existing because the usage of water saving devices is not popular in Beijing. Hence, water-saving technologies may play a big role in WDM in Beijing.

Household consumption in Beijing is apportioned as follows: 40% for face washing, bathing and showering; 26% for toilet flushing; 22% for washing machines; and 9% for dishwashing and cooking. Compared with the rest of the uses, the proportion for drinking is minimal, which is only 3%. Statistics shows that there is a potential to improve the water efficiency of common household appliances such as taps, toilets, and washing machines in 2010. The devices being used and the potential water saving in households are described in Table 2.

As mentioned above, water efficiency can be improved by using water-saving appliances in households, public places and industries. Water reuse will be an effective approach to reduce water consumption as well. Table 3 gives the estimated water demand and expected water saving for Beijing in 2010 by use of different water saving patterns, and the ratio of water saving is more than 30% in 2010.

(2) Water price methods

Much research has demonstrated that the price charged per unit of water affects the user's choice and behaviour towards water use. That is, when the price per unit is relatively high, the utilization rates may be low and the consumers may adopt water-conserving technologies. However, how to make a reasonable and affordable water price is still a programme needing study.

According to the related rules in China, water users are normally charged by water use as resources, the services offered by water suppliers, treatment of wastewater, etc. In Beijing, the water price increased by 740% from 1995 to 2007, while the water consumption maintains the same level. One main reason for this is that water consumption patterns are influenced by a number of factors, e.g. income increase, living standard improvement, climate variations, making it very difficult to analyse the relationship between water price and reduction of water use. Although few case studies gave the result that an increase of water price may reduce water use by a known percentage, price elasticity has been used to describe the sensitivity of demand towards price changes.

$$E_d = \frac{\Delta Q / Q}{\Delta P / P} \tag{1}$$

Equation (1) gives the definition of price elasticity, in which E_d is the price elasticity; Q is the water use; ΔQ is the change of water use; P is the water price; ΔP is the change of water price.

According to the related research, the water price elasticity is 0.164 for domestic use in Beijing, and this means that the water consumption drops 1.64% when there is a 10% increase in price. The income elasticity is 0.388, and this causes an increase in water use. This is why water price has little effect on domestic water use in Beijing. Nevertheless, water price has influenced industry's water use greatly, for its water price elasticity is 0.462. Another important reason is that the water fee charged is relatively low compared to the actual family income, as shown in Table 4, and there is enough space to increase water price.

Table 4 Water charges from 1996 to 2008 in Beijing.

Component	*D*	*I*	*P*	*C*	*S*	WR		WT	
						SW	GW	DU	OU
1996	0.50	0.8	1.20	2.00		0.16		0.00	0.24
1997	0.70	1.0	1.20	2.00		0.20		0.10	0.30
1998	1.00	1.3	1.50	2.40		0.30		0.10	0.30
1999	1.30	1.6	1.80	2.70		0.60		0.30	0.50
2000	1.60	2.4	2.80	2.80	5–30	0.80		0.40	0.80
2001	1.60	2.4	2.80	2.80	5–30	0.80		0.40	0.80
2002	2.00	2.9	3.80	3.80	8–50	1.20	0.30	0.50	1.00
2003	2.30	3.2	4.20	4.20	10–60	1.50	0.60	0.60	1.20
2004–2008	2.80	4.10	3.90	4.60	40	1.10	1.10	0.90	1.50

D, charge of domestic water use; *I*, charge of industry water use; *P*, charge of public service water use; *C*, charge of commercial water use; *S*, charge of special water use; RP, resources preservation fee; *WT*, wastewater treatment fee; SW, surface water; GW, groundwater; DU, for domestic use; OU, for other uses. In Table 4, the unit of water charge is RMB (Chinese Yuan), and 1$ = 6.8 RMB.

(3) Adjustment of industry structures

In the WDM programme, the adjustment of inter-structure within water users is a valid way to achieve water saving and improve water efficiency. Adjustment includes the change of agriculture proportion, and the proportions within industries. Tangshan, a city located in Hebei province of China, is recovering from the big earthquake in 1976 and is growing rapidly based on its industries. However, it has been suffering from water shortage in recent years. The estimated water use was 693 million m^3 in 2005, while after the adjustment of industry structure, the industrial water use dropped down by 70 million m^3, which was 622 million m^3, and the water saving rate is about 10%. The change is shown in Fig. 2. In Fig. 2, the industry structure is measured by a percentage, i.e. the ratio of each industry's production value to the total production value of all the industries.

From the above analyses, the conclusion can be drawn that water-saving technologies may be a valid measure of water saving in Beijing, and the effect of water price on industry water use is obvious. The water saving potential is great by through industry structure. It is necessary to find a way of discouraging excessive water use with affordable prices for all citizens.

WDM does not only mean the application of water saving technologies and water charge, but also the public awareness, attitude, response and participation in such programmes. All these technological, economical and social-politics measures are all included in the IWDM program, and these measures have been carried out in IWRM for years.

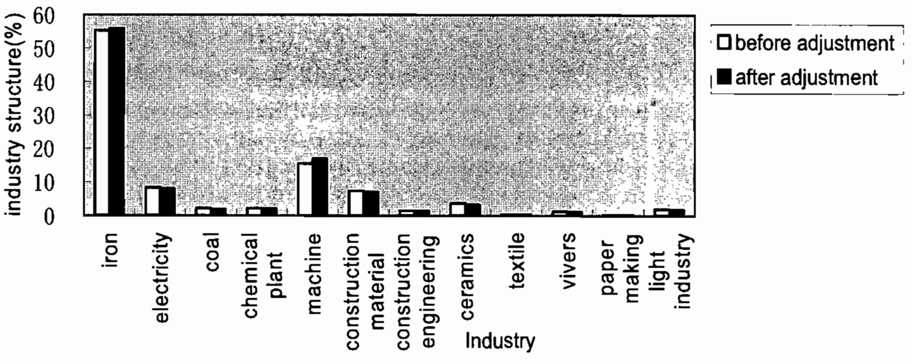

Fig. 2 Adjustment of industry structures.

INTEGRATED WATER RESOURCES MANAGEMENT

Classification of IWRM

IWRM has been put forward in the proposed water framework directive for years. Traditionally, it refers a number of legal: e.g. compulsory use of certain technologies, quota for water use, water conservation; administrative: e.g. information campaigns, user education, programs to increase environmental awareness, concern for public image; economical: e.g. tariff systems, progressive pricing, subsidies for water saving investments); and technological: e.g. survey, planning, regulation; and optimal allocation: approaches or mechanisms being used during the allocation, development, utilization, regulation and conservation of water resources to ensure sustainable water use. According to the different process it emphasizes, IWRM has been categorized to three branches.

(1) Water supply management

Water supply management is a kind of water transport management. It concerns the reduction of water loss during the process of transport. Also, it includes taking measures to increase water supply ability to get more useful water.

(2) Water demand management

In China, this part is normally thought to be both a user-side and source management in the process of water use. It includes using alternative sources to freshwater, e.g. seawater for cooling systems, rainwater for watering gardens, and recycling of used water for other uses. Its target is to avoid excessive water use and reduce water consumption. Besides the essential measures mentioned in WDM, inter-structure adjustment within industries is an important way in this management branch. However, this adjustment is on the basis of water resources optimal allocation and medium- or long-term planning. Especially in drought season or emergency situations, the efforts made to ease the conflicts of different water users is much beyond the measures of WDM, and IWRM measures have to be adopted.

(3) Water quality management

Since different bureaus are responsible for water quantity and water quality monitoring and management in China, water quality is separated from water quantity. Water quality management focuses on the production process improvement and water quality monitoring. It includes measures to reduce water pollution and implement environmental management systems.

RELATIONSHIPS BETWEEN WDM AND IWRM

Although many measures concerning WDM are the same as in IWRM, and nowadays WDM is put forward at a higher level, WDM is only one part of IWRM. Water efficiency and water conservation programmes are not meant to hinder or upstage water supply and water quality management; rather it complements traditional approaches. Furthermore, when WDM is used and operated in a wise and economical manner, demand management strategies can provide decision makers with some extra space for long-term planning. Water supply management may get benefit from the long-term planning, and the water quality will also be improved.

Good water conservation strategies need collaboration from both water users and water supply and water quality managers. Different management should be emphasized, depending on the actual water problems. With the development of the economy of China, water saving measures based on WDM will be a valid approach in most cities. However, there are lots of areas, especially in the countryside, suffering from water shortage owing to a lack of hydraulic structures. Therefore, water supply management will maintain its importance for a certain period. As reported by the Chinese government, the available water will increase by 79.5 billion m^3 in 2020 by constructing water supply facilities and projects. Water pollution is a serious problem during the development of the economy. According to official statistics, the effluent was 75 billion m^3 in 2007, in which only 41.6% met the standard requirement for water quality. So the poor water quality is a major problem for many rivers and lakes, and this influences the available water volume. Strengthening the management of water quality does not only improve the environmental situation, it also promotes the effects of WDM measures and *vice versa*.

CONCLUSIONS

Water supply management, water demand management and water quality management are three important parts of integrated water resources management. With the development of society and the economy, WDM will play a major role in the water-saving programme by the use of water-saving technologies, water price approaches, and adjustment of industry structures, etc. WDM is highlighted, but this does not mean it will replace or hinder the other approaches; rather they complement each other. It is necessary to distinguish different water problems before carrying out the policies and measures.

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Water demand management instead of water supply management: a case study of Yulin City in northwestern China

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Abstract Water Supply Management (WSM) during the 20th century has tremendously benefited many areas around the world. However, with rapid socio-economic development and major engineering interventions, lessons from most countries have demonstrated that WSM is not suitable because it treats freshwater as a virtually limitless resource, and rarely takes full account of environmental and economic impacts. Water Demand Management (WDM) has gradually found its place in Integrated Water Resources Management (IWRM). Yulin City, in the dry northwestern part of China, has experienced water shortages, hindering socio-economic development. Rapid population growth, and industrial and agricultural development, have increased the gap between water supply and demand. With Yulin City as case study, WDM practice is presented in this paper. The WDM measures adopted in Yulin City include legislation, institutional arrangements, water metering, leakage reduction, wastewater re-use, water allocation between multiple sectors, water price, and public education to improve awareness of water saving.

Key words water supply management; water demand management; water price; climate change; water conservation; Yulin City, China

1 INTRODUCTION

With large-scale population growth and socio-economic development in recent decades, the available water resources throughout the world are becoming depleted. This problem is aggravated by climate change, which has a distinct impact on water resources (Ren LiLiang *et al.*, 2002; Li Chunhui *et al.*, 2004; Chen Liqun *et al.*, 2007; Chuang-lin Fang *et al.*, 2007; Zhang Jianyun *et al.*, 2007). These factors enlarge the gap between supply and demand for water and have led to water crises in many countries (Stephenson, 1999; Arnell, 2000; Middelkoop *et al.*, 2001; Shuval & Dweik, 2007; Wang Xiaojun, *et al.*, 2008; Steele-Dunne, *et al.*, 2008).

Economic and environmental reasons have led many to question the technical reliability and institutional capacity of water supply management; managers and planners are slowly moving towards demand management approaches. The benefits of Water Demand Management (WDM) include lower costs as compared to new supply schemes and resource conservation (Stephenson, 1999; Gumbo, 2004; Shuval & Dweik, 2007).

For Yulin City, located in northwest China, water is a strategic resource and a major constraint for agricultural production. Population growth and socio-economic development have resulted in large-scale withdrawals, pollution, ecological deterioration, and river discharge decreases (Wang Xiaojun *et al.*, 2008). In this paper, we define WDM based on the theory of supply and demand, and then contrast WSM and WDM. Yulin City, where a wide range of measures have been taken, will be the case study. Results show that water-use efficiency has progressed during the past years.

2 THEORY OF WATER DEMAND MANAGEMENT (WDM)

2.1 Theory of supply and demand

A fundamental concept in economics is the law of supply and demand. Price is a fundamental economic concept, which determines demand (Petersen, 1994; Stephenson, 1999; Rogers *et al.*, 2002). Figure 1 shows water supply and demand with different price structures.

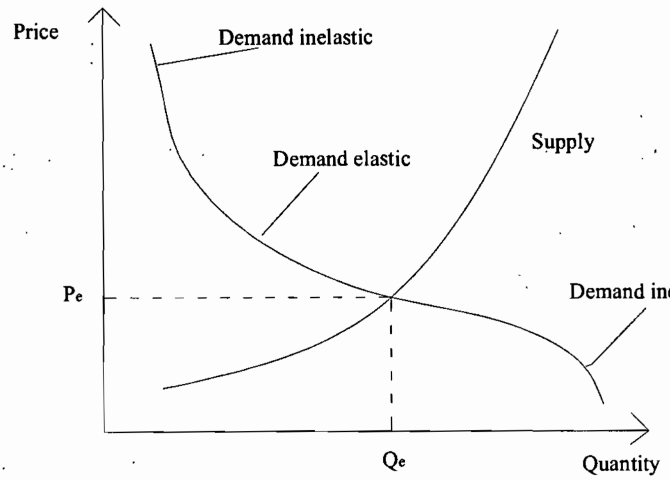

Fig. 1 Water supply and demand with different price structures.

Demand changes as price changes, the ratio of which is called the price elasticity of demand. Similarly, there is a price elasticity of supply. P_e is the equilibrium price when supply and demand are balanced. Given the limited supply of water resources, there is a limit on how much water can be used. Similarly, with increasing price, further reductions in water use may require changes in behaviour that are inconvenient or contrary to personal or social norms. At even higher prices, there will be no demand reduction at all if it means cutting into essential uses such as cooking. When demand does not change much compared to the change in price, demand is said to be inelastic (Petersen, 1994; Stephenson, 1999).

2.2 Water Supply Management

At the outset of socio-economic development, water is relatively abundant compared with demand, and there is almost no need for water resources management. As the socio-economic system develops, water demand grows rapidly and there will be a gap between supply and demand. Water managers increase supply through engineering projects such as dam building, pipelines, diversions and distribution systems. This is known as Water Supply Management (WSM).

Over the decades, WSM has proved to be insufficient to deal with strong competition for water with growing per capita water use, increasing population, urbanization, pollution and shortages of funds. Once there is almost no water available to increase supply, limiting water demand begins to be taken into consideration (Tate, 1990; Frederick, 1993; Stephenson, 1999; Gumbo, 2004; Mohapatra & Mitchell, 2004; Gumbo *et al.*, 2005; Butler & Memon, 2006; Shuval & Dweik, 2007).

2.3 Water Demand Management

Water Demand Management (WDM) is defined as the development and implementation of strategies aimed at influencing demand, so as to achieve efficient and sustainable use of this scarce resource. Besides efficiency, it should promote equity and environmental integrity (Petersen, 1994; Stephenson, 1999). WDM has been widely implemented in many areas around the world, such as South Africa, the Middle East, and Canada (Stephenson, 1999; Gumbo, 2004; Satya *et al.*, 2004; Gumbo, 2005; Shuval & Dweik, 2007; Medellín-Azuara *et al.*, 2008). It has been demonstrated that important benefits of WDM include both lower costs and better environmental protection. In this paper, WDM includes the following:

– The promise of WDM is to have a good understanding of the basic rules of local water resources systems. To achieve the objectives of WDM programmes, a basic insight to the local water system needs to be obtained, such as the carrying capacity, the state of water supply system, and water demand characteristics. WDM needs to be based on local needs and conditions.

- The basis of WDM is government regulations and policies, which means that WDM should be politically acceptable.
- WDM includes structural measures and non-structural measures. Structural measures usually include low-flush cisterns for toilets, leak detection and control systems in distribution networks, drip irrigation in agriculture, etc. Non-structural measures include economic and legal incentives to change the behaviour of water users and the creation of the institutional and policy environment that enables this approach. We highlight a number of possible WDM measures that might be taken: rainwater collection; storm water runoff diversion and collection; re-use especially for irrigation of fodder; progressive pricing policy; water-efficient appliances in households, industries and agriculture; farm gardens using minimum tillage and grey-water irrigation; reduction of leaks from distribution systems; use of lower pressures; and consumer education.
- The aim of WDM is to improve water-use efficiency and achieve a balance between society and nature. In the wake of economic and social development, the objectives of WDM change over time. In the initial stage, WDM aims towards high water-use efficiency through economic measures (water rights, water market, price) to promote efficient use of water. With the socio-economic development and growing environmental awareness, concepts gradually change to safety, health, etc. The aim of WDM then shifts more to quality requirements.

3 THE CASE OF YULIN

3.1 Study area

General features of the area Yulin, one of the regions suffering from serious soil erosion in the middle reaches of the Yellow River, lies in the arid and semi-arid area of the Loess Plateau of northwest China, between 36°57′–39°35′N and 107°28′–111°15′E (Fig. 2). Yulin is a region that encompasses twelve counties with a combined area of 43 578 km^2. It is close to the borders of four provinces, Gansu, Ningxia, Shanxi and Inner Mongolia, and at the centre of abundant resources (Dong *et al.*, 2007).

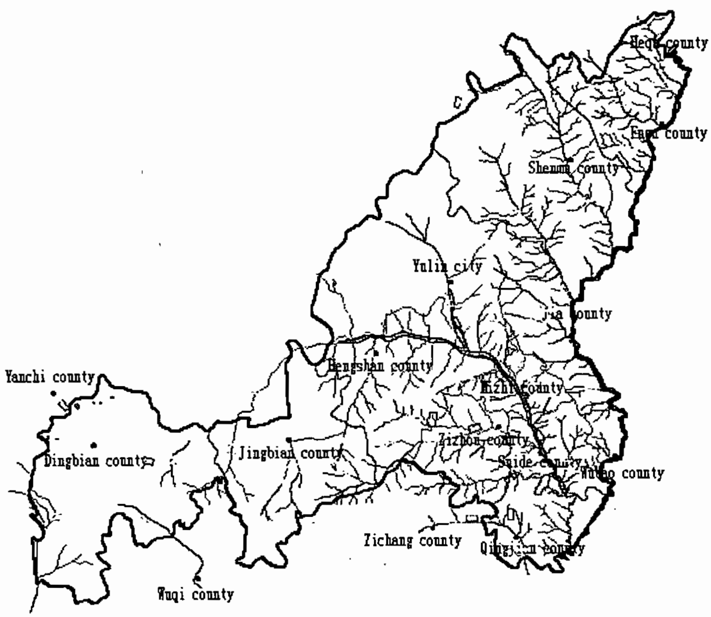

Fig. 2 Sketch map of the location of Yulin City.

3.1.2 Water resources Mean annual runoff over the period 1956–2005 in Yulin City was 22.9×10^8 m^3, equivalent to a mean runoff depth of 52 mm. The annual groundwater recharge is 24.78×10^8 m^3, of which 15.6×10^8 m^3 runs off through the rivers. Water resources are unevenly distributed between the southern and northern parts of the city.

With the development of chemical and energy industries, the economy of Yulin has increased quickly in recent years (Fig. 3). From 1980 to 2005, the GDP rose 22-fold. Given the population growth, per capita GDP has increased 15-fold. Annual per capita water consumption has decreased, from 213 m^3/p in 1980 to 180 m^3/p in 2005. Infrastructure has been constructed to deal with water shortages, increasing water supply (Fig. 4). Increase in demand has outrun increase in supply. For example, four main rivers in Yulin City have run dry several times (Fig. 5).

3.2 WDM in Yulin City

The main objectives of WDM in Yulin City were to promote in the short term (2005–2010) efficient water use through economic measures (water rights, water market, price, etc.). In the long-term (2010–2020), the WDM programme will pay more attention to quality requirements, and quality will become the main target. The approach is to optimize between the economy and ecology. The main tools used to achieve WDM objectives are described below.

3.2.1 Legislation China has adopted four important laws concerning water management: Water Law (adopted in 1988 and revised in 2002), Law on Water Pollution Control (adopted in 1984), Soil and Water Conservation Law (adopted in 1991), Flood Control Law (adopted in 1997).

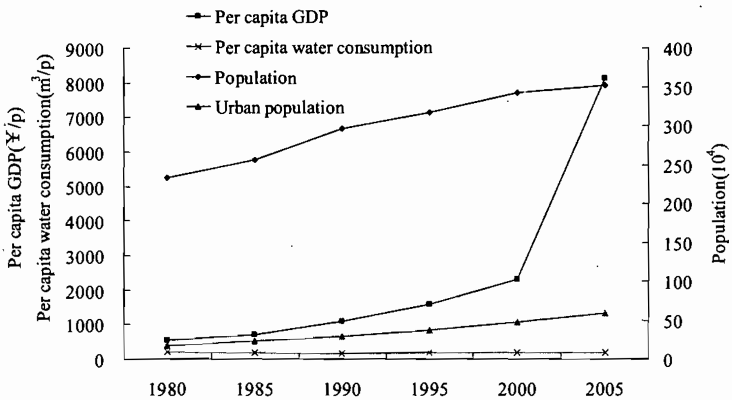

Fig. 3 Population growth and economic development of the Yulin City.

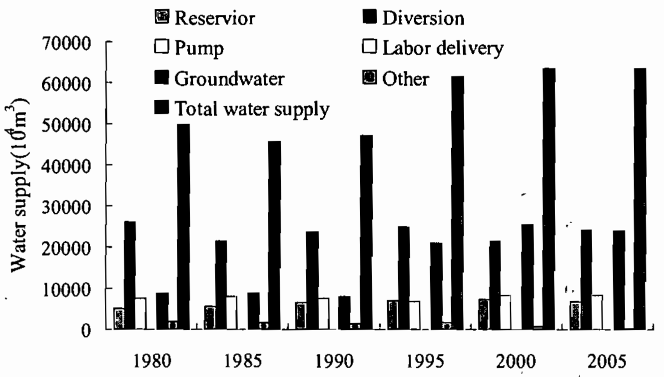

Fig. 4 Water supply of Yulin City.

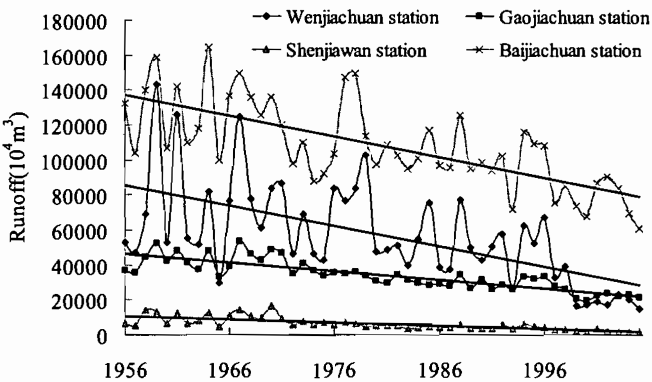

Fig. 5 Runoff changes in the main river of Yulin City.

These four laws are fundamental laws for water resource utilization, protection, management and flood control in China (Zhang Hailun, 2005). Shaanxi Province also adopted additional legislation, such as the General Rule for Water Resources Management of Shaanxi Province, and the Rule for Water Saving Management of Shaanxi Province.

3.2.2 Institutional arrangements Because the initial stages of WDM are politically stressful, given the re-allocation of resources away from previously privileged political constituencies, institutional arrangements are crucial for success. In 2002, the Yulin Municipal Water Affairs Bureau was established; it is responsible water administration of the city and is part of the municipal government. Its main responsibilities are: carry out national and provincial guidelines, policies, laws and regulations; develop policies and regulations and bring them into force after approval; develop protection planning to conform with relevant laws, regulations and standards of national resource and environmental protection; integrate water management to include atmospheric water, surface water and groundwater; put into practice a water withdrawal license system; collect water fees; and, finally publish the water resources bulletin of the city.

3.2.3 Metering and leakages reduction Water metering is a prerequisite for pricing and as such essential for a successful WDM programme. The aim is to improve existing structures in order to have a better control on water demand. Yulin also takes measures to reduce leakage and water losses. Nowadays, the problem of water loss and its control in the water distribution system is becoming more and more important. At present, the loss rate of Yulin is 16% of water supply, implying good opportunities to save water through replacement of older pipes, electronic monitoring, retrofitting campaigns and so on. The water use efficiency of Yulin has improved over the past years (Fig. 6).

3.2.4 Water allocation between sectors Water-scarce regions will generally adjust their socio-economic structures to save water. This includes re-allocation from sectors with lower added value to sectors with a higher added value. Such re-allocation will obviously be advantageous to society as a whole. The classic case is the different values attained in the agricultural and urban sectors. Irrigation has high water consumption but relatively little economic output. An interesting example in the Yellow River Basin is a factory financing the construction of a water-saving drip irrigation system, thereby obtaining the right to use the water thus saved. The opportunity cost for irrigation water may be only half, or less, than the best alternative use.

3.2.5 Economic methods Since the Dublin Conference on Water and the Environment, it has become generally accepted that water should be considered an economic good (Stephenson, 1999; Savenije & van der Zaag, 2002). Therefore, economic methods can be implemented for demand reduction. Such economic methods may include pricing mechanisms, incentives and penalties. Water pricing is a fundamental economic tool to influence water demand. Yulin City has adjusted

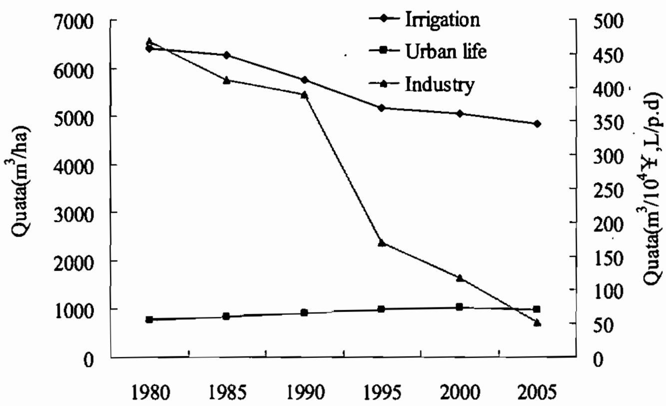

Fig. 6 Quota of Yulin City in the past years.

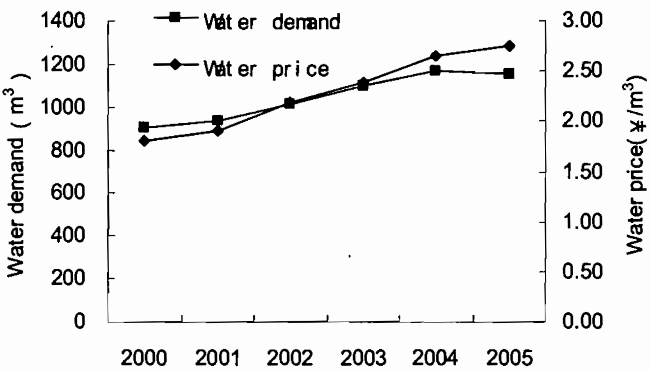

Fig. 7 Water price and water demand in Yulin City.

the water price several times, and water use has changed accordingly (Fig. 7). Adjustment of the water price is one of most effective economic levers for WDM programme.

4 CONCLUSIONS AND DISCUSSION

Based on the implementation of WDM in Yulin City during the past years, we can draw the following conclusions:

1. Results showed that WSM treats freshwater as a virtually limitless resource, rarely takes full account of environmental or economic impacts of municipal water services, and assumes that current levels of water demand are insensitive to policy and behavioural changes.
2. Yulin City has a serious shortage of water following the recent economic development. In the past years, local governments mainly relied on WSM. WDM has been promoted in Yulin City through a wide range of structural measures and non-structural measures including legislation, institutional arrangements, metering, and leakage reduction to improve water-use efficiency, water allocation between sectors, and economic methods.

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Economic assessment of differential-quality water demands from a metropolis and its peri-urban environments

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Abstract Resource sustainability, range and size of project operation and the level of acceptability will influence the success of innovative change in behaviour in using scarce natural resources such as surface water and groundwater. This paper looks at a combination of water demand, public choice and financial sustainability of water supply augmentation in Delhi, India, in both planned urban and unplanned peri-urban areas having differing levels of planning and resource availability. The preference heterogeneity of different households for water supply scenarios differentiated by their "quality" (potable or non-potable) and "source" (surface water or groundwater) is examined through choice experiment (CE), using an iterative bidding game. It is argued that more attention must be paid to diverging perceptions of the quality and availability of water and how it should be accessed and delivered to maintain both the resource sustainability and acceptance by the public.

Key words public choice; urban water; resource quality; pricing; planning; willingness to pay

INTRODUCTION

The current water scarcity scenario has changed people's perception and awareness about "source limits" and the economic realities of water supply made available by the public utilities. In an environment of constrained availability, the water supply options and people's preferences vary according to several factors including: planning levels, local perception of water scarcity, individual water-use requirements, and previous experience with irregularity in supply (Adamowicz *et al.*, 1994; Louviere *et al.*, 2000; Ryan & Wordsworth, 2000; Hanley *et al.*, 2005). In a large metropolis, the current water supply system typically falls well short of offering various customer classes a product that precisely fits their needs. A solution may be reasonably acceptable to people in planned areas, but, at the same time, totally unacceptable to customers in unplanned areas. and *vice versa*. Given the social, economic and environmental externalities associated with unsustainable water use, it is important that the customer's "preferred choice" informs the investment in infrastructure restructuring. The combined use of surface water and groundwater that recognizes site-specificity and the communities' preference structure can greatly influence the social and economic sustainability of communities in a growing metropolis.

This paper looks at a combination of water demand, public choice and financial sustainability of water supply augmentations in Delhi, India, in both planned urban and unplanned peri-urban areas having differing levels of planning and resource availability. The preference heterogeneity of different households for water supply scenarios differentiated by their "quality" (potable or non-potable) and "source" (surface or groundwater) is examined through choice experiment (CE). The values resulting from the analysis are assessed in terms of water supply augmentation options and their practical limits, incorporating the choice and preferences from the heterogeneous planning environments typical of a metropolis. The outcome indicates that a public policy is needed that is tailored to the varied preference behaviour of communities and recognizes hydrological complexities associated with conjunctive management of groundwater and surface water.

THE CONTEXT

The Delhi Jal Board (DJB) acts as an autonomous body under the Government of the National Capital Territory (NCT, 1483 km^2), and is estimated to serve around 14 million people. Although the DJB has a production capacity of around 2900 million litres a day (MLD), the amount of non-revenue water (NRW) assessed by various studies is estimated to be as high as 59% including the real losses (through leaks) and apparent losses (through theft or metering inaccuracies). High levels of NRW are detrimental to the financial viability of DJB, as well to the quality of water itself. Note, the gross annual water demand for the city is projected to increase from 2050 MLD at present to 4700 MLD in 2021—an increase of 75%—in order to cater for both the present deficiencies and future predicted population growth. Against this growing demand, the net water supply (i.e. excluding leakage and losses) from the nine supply sources developed to date (2008 level) is only 696 million gallons per day (2630 MLD), giving rise to a 33% deficit in net demand. Therefore, although the availability according to the production capacity is over 200 litres per capita per day (Lpcd), the DJB is currently unable to provide a level of service to match the demands of the population.

Allocation of water among the users is performed through scheduled intermittent supply, which leads to significant wastage of water due to the practice of emptying storage tanks before every new round of supply, as well as quality problems due to sewage infiltration in distribution pipes. The proper functioning of the distribution system is further jeopardized by the common practice by individual users of installing suctions pumps in order to increase the amount obtained during the limited period of supply. Various supply augmentation schemes that rely upon interstate water transfer would eliminate the demand–supply gap by 2021; however, such approaches are costly and would remain financially non-viable in the long run due not only to inefficient water use, but also to the possibility of inter-sectoral and inter-state water allocation conflicts. Much of the future demand would be satisfied if the water losses in raw water transmission and treated supply can be drastically reduced. The pressure can also be reduced by promotion of multiple water sources augmented locally by both private and public means, including in-house water use efficiency and conservation.

In the study area, despite the existence of a piped water system, at least 36% of the planned urban population meets 90% of its water need from personal tube wells. Reliance on groundwater becomes almost 100% in many cooperative society flats who resort to deep drilling to withdraw groundwater (Dutta, 2007). They have their own water distribution systems that often provide "24×7" water supply. The residential societies and real estate developers have in the past joined in a cooperative effort to devise an alternative management plan to augment the availability and improve the quality of these systems, depending less on the public water supplies. The additional water is sought from the local bore wells due to the fact that groundwater is a more dependable water source than the surface water supplied from the water utility. In the unplanned peri-urban areas, a very high proportion of the population is outside the piped network and in such areas groundwater is the main source of water. According to the study, the demand–supply gap in planned colonies is nearly 20% less than the gap in unplanned peri-urban colonies. The households dig shallow wells fitted with handpumps, or install motors, and draw subsoil water. Owing to this situation of a growing population without a commensurate increase in the availability of raw water, the groundwater in many areas has been over exploited. This has disturbed the hydrological balance leading to a decline in the productivity of wells, increased pumping costs and greater energy requirement. Utilisation of groundwater in peri-urban settlements is very common, which means that people consider this resource as potentially important. However, the quantity of freshwater is very marginal and most of the groundwater reserves consist of brackish or saline water.

METHODOLOGY

Preferences for change *versus* the *status quo*, and customer's willingness to pay (WTP) are estimated through utility function-based discrete choice models using a maximum likelihood estimation technique. The dependent variable—households' choice of water supply scenario—is observed as a discrete variable. Using the statistical package LIMDEP 8, multinomial logit (MNL), multinomial probit (MNP) and multinomial nested logit (NL) models are used to estimate households' WTP in both planned urban and unplanned peri-urban residential units. The estimated parameters of the choice models define the utility functions for each alternative with different source and quality—groundwater or surface water. Multivariate regression analyses are employed to: (a) test the hypotheses based on the statistical significance of coefficients; (b) assess customers' preferences for water supply options; and, (c) estimate WTP (Bateman & Willis, 1999; Wedgwood & Sanson, 2003; Oca de *et al.*, 2005). In the choice experiment, there are basically three broad choice-sets with different quality, quantity and reliability attributes:

- dual-quality river and local groundwater: 20–25% potable water (36 Lpcd) meeting WHO or EU standards (e.g. 80/778/EEC Standard) and 75–80% non-potable water (129 Lpcd) meeting extended limits of IS 10500;
- single-quality river water (165 Lpcd) meeting IS 10500 quality standards with increased hours of supply;
- current scenario:
  (i) those who are willing to pay for the existing supply, if the supply is maintained as per notified timing with additional private investment in water purifiers or bottled water for potable needs; and
  (ii) those who are not willing to pay anything and demand no improvements in supply attributes (opt-out from the choice experiment).

DATA ANALYSIS

In the first instance, choice responses are analysed for planned urban settlements irrespective of water quality based on source (groundwater or surface water). Both MNL and MNP functions are employed to find any large difference between the coefficients and significance of parameters. In the case of planned urban settlements, a groundwater dummy variable, household size, daily consumption, awareness level and price of water are found to be significant variables in explaining the difference in probability of selecting an alternative arrangement of water supply, irrespective of source distinction. A positive coefficient for daily consumption and awareness levels means that an increase in these predictors leads to an increase in the predicted probability of choice occasion and paying behaviour; similarly, negative coefficients on groundwater dummy variable, household size, and price of water means that an increase in these predictors leads to a decrease in the predicted probability of choice occasion and paying behaviour for alternative water supply. The MNP model results in a WTP value of Rs12.54/kilolitre (0.27 US$/kL), whereas the MNL model results in a WTP value of Rs12.98/kL (0.28 US$/kL) for the alternative water supply option irrespective of the source distinction between groundwater and surface water.

Table 1 shows the ratio of parameter estimates and preference behaviour of households in unplanned peri-urban areas without making any source distinction between groundwater and surface water. Education and awareness of the respondents is positively correlated with their preference behaviour, implying that there is a positive relationship between a respondent's WTP and the number of education years as well as his level of awareness (Louviere *et al.*, 2000). Respondents with personal groundwater bore wells are less likely to pay more than the existing amount for an alternative water supply. This is because they have already invested a significant amount in securing a reliable water supply. The range of monthly bills likely to increase with alternative scenario significantly affected ($p < 0.001$) the price sensitivity (MacDonald *et al.*, 2005).

The coefficient of current monthly water bill is negatively signed: respondents getting higher water bills appear to be less likely to increase their payments for improvements in water supply. Conversely, respondents who are getting lower monthly bills are more likely to choose the non-*status quo* option. The negative coefficients for households which have developed a large water storage capacity show that they have already invested a significant amount of money in storing water to avoid unreliable water supply; therefore, their additional WTP for an improved water supply system is actually lower. The results demonstrate that the culture of unreliability in the

Table 1 Comparison of ratio of parameter estimates and other statistics for MNP and MNL models for unplanned peri-urban areas irrespective of source distinction.

Variables	Coefficient (normal)	Coefficient (logistic)	Ratio of parameter
Constant	−1.83900467 (−5.372)	−3.45269559 (−5.442)	1.877481
EDUCATION	.03507168 (2.046)	.03144157 (1.010)	0.896495
HOUSEHOLD SIZE	−.17160736 (−6.592)	−.32090264 (−6.285)	1.869982
QUANTITY_DUMMY	−.30470228 (−2.070)	−.51301725 (−1.946)	1.683667
MONTHLY_BILL	−.00029037 (−1.858)	−.00048517 (−1.729)	1.670868
AWARENESS	.22016594 (2.617)	.33292274 (2.169)	1.512145
GWATER_DUMMY	−.37703663 (−2.570)	−.64226197 (−2.447)	1.703447
BELIEF	0.00073714 (2.108)	0.00079279 (1.255)	1.075494
BID	−.2688606 (−11.294)	−.48991235 (−9.928)	1.822181
Log-likelihood function (LogL)	−261.119	−248.8049	
Restricted log-likelihood ($LogL_0$)	−402.584	−402.584	
Chi-squared	282.9295	307.5594	
Prob[ChiSqd > value]	0.0000000	0.0000000	
Hosmer-Lemeshow chi-squared	22.38609	49.94611	
P value	0.00425	0.00000	
McFadden	0.35139	0.38198	
Mean WTP (Rs/kL)	11.4509	12.5011	
Mean WTP (Rs/month)	144.28	157.51	

study area diminishes the willingness to pay for services. Respondents' household size is a significant variable, with the negative coefficient meaning larger household sizes would have less likelihood of paying higher for improved water supply (Hensher, 2004; Cai *et al.*, 1998). Similarly, the coefficient for quantity of daily supply is a significant variable with negative value. In this context, a large majority of the customers receive water for 3–4 hours daily and it can be inferred that the level of satisfaction amongst customers in both planned and unplanned areas is very low. The probability of a respondent agreeing to pay more decreases with increasing bid price (Raje *et al.*, 2002; MacDonald *et al.*, 2005; Willis *et al.*, 2005). The monthly bills that they would be willing to pay in these areas are as different as their consumption structure is dissimilar: Rs19.8 kL/month (0.43 US$) in planned areas and Rs 12.6 kL/month (0.27$) in unplanned areas.

Modelling preference heterogeneity with source distinction

The assessment of preferred choice behaviour is based upon estimated parameters and is a function of the random component assumed for preferences. The presence of two sources of uncertainty—parameters and preferences—and the additional source of variation between planned urban areas and unplanned peri-urban areas gives differing sets of results. As the partial effect or interpretation of the slope coefficients depends upon the unobserved heterogeneity of the respondents, the interpretation becomes useful when the endogenous explanatory variables interact with heterogeneity—in this case, the alternative specific constant (ASC). However, the attributes of the choice do not vary with the respondents, only the socio-economic characteristics vary. Therefore, we cannot use the ASC to interact with the constant choice specific attributes. This is taken care of in the interaction effect using a nested structure where two variables, "extra duration" and "bid values", vary with respect to the alternatives chosen. In this case, sources of heterogeneity are

Table 2 Estimation of parameter coefficients for two scenarios using a nested structure under the multinomial logit model (MNL Nested).

Variables	Coefficient	Standard error	*t* ratio
ASC_DUAL*	−3.88048620	1.03252163	−3.758
ASC_SINGLE	−2.76757203	0.78471756	−3.527
EXTRA_DURATION	0.16523668	0.01391350	11.876
PAYMENT	−0.00413130	0.00062691	−1.760
Interaction effects			
ASC_DUAL			
DUA×PLANNING	−0.33562158	0.07616614	−4.406
DUA× AWARENESS	0.56248998	0.15107321	3.723
DUA× AGE	−0.03886440	0.01050529	−3.700
DUA× HEAD	0.41415322	0.29802436	1.390
DUA× ENV	1.04699051	0.32747920	3.197
DUA× EDUCATION	0.26606744	0.04890373	5.441
ASC_SINGLE			
SIN× PLANNING	0.22353802	0.06939939	3.221
SIN× AWARENESS	0.74980124	0.13190306	5.684
SIN× AGE	−0.03292063	0.00887819	−3.708
SIN× HEAD	0.60775243	0.25175857	2.414
SIN× ENV	1.10176310	0.28798764	3.826
SIN× EDUCATION	−0.00066405	0.02558244	−.0260
Pseudo R–square			0.449
Log likelihood function			−642.3527
E (WTP): SINGLE QUALITY			Rs189.32
E (WTP): DUAL QUALITY			Rs295.05

* ASC_DUAL takes the value of one for Dual quality option and zero for Single and BAU options; similarly, ASC_SINGLE takes the value of one for Single quality options and zero for Dual and BAU options. Here, BAU option is designated as the omitted level so that the parameter estimates on included levels represent the change from the *status quo* option.

better modelled and controlled for using a flexible form of specification such as nested models, as the determinants of heteroskedasticity can be better attributed.

The estimated results of customers' WTP show that changes in service attributes between dual quality and single quality are large enough to give differing estimates of the benefits. As very few households opted for dual-quality supply options in unplanned areas, the WTP estimated from the MNL (Nested) from a combined sample for dual quality shows a preference for planned areas only (Table 2). In planned areas, households are willing to pay Rs295/month for decentralized dual-quality water supply. People in planned areas seek collective action to ensure cost minimization by a decentralized treatment option for high-quality potable water. The WTP for centralized single-quality improved water supply for both planned and unplanned areas put together is Rs189/month, while the current average monthly bill paid to the utility company is Rs89. This shows that households are willing to pay more then double their monthly bill for single-quality improved water supply with assured reliability.

CONCLUSIONS AND POLICY IMPLICATIONS OF THE STUDY

When a public water system is unavailable, insufficient, unaffordable, or inoperative, households pump groundwater privately to meet their needs. The alternative strategies might include: direct use of groundwater from a private tube well, development of a private small supply network fed with untreated groundwater, or supply by tanker, groundwater remaining the primary source of raw water in most of those private supply chains (Saleth & Dinar, 2001; Llorente & Zérah, 2003; Maria, 2006; Briscoe & Malik, 2006; Ramachandran, 2008). Many households either partially or totally substitute groundwater for water purchased from private vendors and the public supply, depending on the cost of extraction and the quality of the water. The main reason behind this situation is the lack of a reliable water supply, which comes largely from surface water sources. In spite of the public neglect of groundwater as a resource for urban water supply, groundwater plays a central role in meeting urban needs through a variety of private and uncontrolled systems (Maria, 2004). Note that, although there are no comprehensive statistics on how many urban and peri-urban dwellers in Delhi rely on groundwater for their water supply, it is estimated that more than 80% of the total population depends partially or fully on groundwater. While 100 000 tube wells were officially registered in 2001, the actual number ranges from 360 000 to 400 000 (Dutta, 2007). While the planned urban settlements incorporate water design principles to some extent, the peri-urban settlements present unique resource attributes as a result of differing socio-economic values, flow asymmetries, and joint production potential (Tovey, 1998).

A large number of households in planned urban areas favour decentralized dual-quality water using both local groundwater and municipal surface water. However, single-quality improved supply regardless of source distinction becomes the most preferred choice if planned urban and unplanned peri-urban households are modelled together. This preference variation across sampled households testifies to a "scope effect", i.e. differential water quality and reliability provisions have differing value estimates between planned urban and unplanned peri-urban areas. By preserving the surface water–groundwater connection, conjunctive management is the preferred strategy for maintaining sustainable supply of a limited resource. However, the public utility must make adequate provisions to maintain safe-yield and control pumpage, to prevent local water tables from experiencing long-term decline. There are three important implications of the study:

Quality differentiated water with source distinction Due to diverse socio-economic status and heterogeneous customer preference structures, public choice and economic values in decision making transform into a site-specific problem. The findings indicate that it is promising to develop differential distribution systems for potable and non-potable water end uses, having heterogeneous customer preference structures. Accordingly, it is desirable to consider a water sensitive approach in the planning phase when developing a master plan for the cities and towns. The system may be desirable from both the customer's preference as well as the resource sustainability points of view.

Decentralized approaches Even though overall gain in welfare is positive, there exists a significant divergence among the respondents about the extent of these benefits between planned urban and unplanned peri-urban areas. The quality differentiated water supply with source distinction would be economically viable for newer colonies and urban extension, owing to the obvious cost of retrofitting in older settlements. People are able to support such decentralized options because they increase their wellbeing and hence increase their ability to pay the required contribution from their discretionary income, making them financially sustainable. The preference for dual quality water with source distinction between groundwater and surface water is almost negligible in the unplanned peri-urban areas, implying that what they primarily need is better water supply. Rural and peri-urban areas adjoining the metropolis are often not well situated to receive imported surface water from the public water supply due to economies of scale. For communities residing in such areas, groundwater is the most accessible and secure water source. Such communities are concerned with having sufficient water supplies for future growth. If groundwater pumping is restricted because of its effect on adjudicated surface water, growth and development in such areas could be curtailed. The involvement of communities in the conception and management of socially desirable alternative systems is very critical, especially when the natural resource is in limited supply. Because groundwater is easier to access and costs less than water from piped systems, groundwater abstraction cannot be easily regulated. Groundwater management strategies by the local governing body and the residential society need to ensure sustainability of groundwater development in the future. Groundwater assessment, monitoring and regulation are key strategies for groundwater management in both planned and unplanned areas of a growing metropolis.

Emergence of private capital and market solution It has been observed from the study that people have a deep understanding of water in their immediate environments and tend to cooperate in times of adversity to avoid high transaction costs that would result from limited resource availability. There are several instances of reciprocal externality wherein households themselves absorb the cost of over-extraction, in terms of higher pumping cost against declining water tables, and cost of salinity in terms of decentralized treatment cost. In several housing societies, the economies of scale act as externalities leading to an optimum investment in the decentralized treatment technology (such as reverse osmosis or ion exchange plant) and the emergence of a voluntary co-operation by the residents for good quality water. The ensuing economic approach of a "free market" for planned urban settlements generated by private capital might lead to significant cost reduction as well as social welfare, but it is questionable to what extent a scarce resource would be conserved for future generations. Therefore, the placement of a regulator is strongly recommended.

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Assessing water allocation strategies in the Krishna River basin, South India

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Abstract Water allocation rules are put into place to ensure that various parties receive a portion of developed water supplies. In the Krishna basin, India, all the water available is fully allocated to some purpose for a large part of the year. Over 90% of the allocated water is for irrigation. However, due to increasing demands from the domestic and industrial sectors, as well as expansion in irrigation areas, there is growing competition between the different water use sectors, as well as the three riparian states that share the Krishna basin. In this study, the WEAP (Water Evaluation and Planning) model is used to assess two case studies in which the implications of two separate water transfer schemes are analysed. The first case study presents the feasibility of a proposed water transfer scheme from the Godawari River at the downstream part of the Krishna River. The second case study presents analysis from the Upper Bhima catchment in the upstream part of the Krishna basin. In the second case study, the impact of water transfer out of the basin for electricity generation, on downstream agricultural water demands, is analysed. Results from both studies stress the fact that water resources management in the region has to be done on a seasonal basis by taking monthly variability into consideration. In both cases, water scarcity occurs during the critical dry months.

Key words water transfer; water allocation; Krishna River basin

INTRODUCTION

The Krishna River basin in central India is one of the most water stressed river basins in the country. The basin stretches over three states: Andhra Pradesh, Maharashtra and Karnataka, and is home to 70 million people. Large-scale irrigation development in the last four decades, accompanied by more recent large increases in urban and industrial demand, have all but fully utilized the water resources available in the basin. In hydrological terms this means that the Krishna basin is a closed basin, with little or no water flowing to the sea. Traditional water uses, like agriculture, face increasing competition from other potentially higher-value uses (e.g. for urban supplies and power generation) as urbanization and regional economic development continue. Increasing demand for water by rapidly growing urban and industrial sectors in major cities like Hyderabad and Pune, has not been factored into water allocation plans for the basin as a whole. The demand from urban centres is expected to double in the next 20 years, creating additional pressure on an already water stressed system. Despite the lack of additional supplies, development of both surface water and groundwater continues, primarily to meet this increasing demand. The problems in the basin are compounded by severe environmental degradation arising from insufficient river flows, overexploitation of groundwater causing salt water intrusion in the coastal areas, and untreated wastewater discharge into the basin's rivers. Several inter- and intra-basin transfer schemes from water surplus rivers to water deficit rivers have been planned as a solution to try to meet growing water demands.

As a part of the Krishna basin study conducted at the International Water Management Institute (IWMI) the WEAP (Water Evaluation and Planning) model (SEI 2001) was used to assess the implications of different water allocation scenarios fur future water allocation options, food production and long-term resource sustainability within the three states in the basin. In this paper, however, we only present two case studies in which the implications of two separate water transfer schemes are analysed. The first case study presents the feasibility of a proposed water transfer scheme from the Godawari River basin in the downstream part of the Krishna River. The characteristic feature of the study is the simulation of the impact of various feasible cropping patterns on water demands as well as the explicit inclusion of environmental water requirements in the simulations. The second case study presents analysis from the Upper Bhima catchment in the upstream part of the Krishna basin. In the second case study, the impact of water transfer out of the basin for electricity generation is analysed.

CASE STUDY 1: STUDY OF THE GODAVARI–KRISHNA LINK AT POLAVARAM

This study evaluated the plausible future scenarios of water availability and use under conditions of various cropping patterns and with the explicit inclusion of environmental water requirements for one of the links of the National River Linking Plan (NRLP) – from the Godavari River at Polavaram to the Krishna River at Vijayawada – "Polavaram Project". Figure 1 shows the proposed project, including the site of the Polavaram Reservoir and the command area of the link canal. The project includes two canals, i.e. one on the right and one on the left bank of the Godawari River. The Polavaram–Vijayawada link command area is located on the right bank, with the link canal starting from the proposed Polavaram Reservoir. The left main canal will transfer 3663×10^6 m^3 for irrigation and industrial needs. The link canal on the right bank will divert 5325×10^6 m^3 for irrigation, domestic supply and industrial use. The planned Polavaram Dam is to have a live storage of 2130×10^6 m^3. The annual total water use is, however, estimated to be 8000×10^6 m^3. Since the planned storage is small in comparison to the water use, run-of-river flows will be utilized to ensure the expected benefits of the project. Thus the project will function more as a barrage combined with limited storage use. The project also includes a hydropower component (GOI, 1999). It has been estimated that the proposed reservoir will submerge around 63 000 ha of land, which at present hosts 250 villages with a total population of 145 000 (Census, 1991; GOI, 1999).

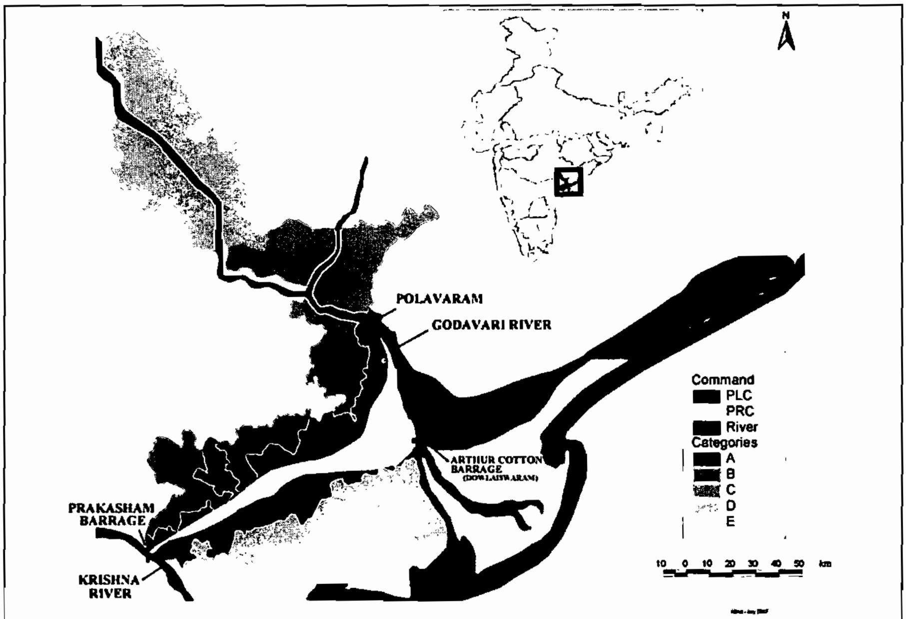

Fig. 1 A schematic map of the proposed Polavaram Project. PLC and PRC are the Polavaram left and right bank command areas, respectively. Category A, the command area for the link canal; Category B, mandals upstream of the link command area; Category C, area submerged by the proposed reservoir; Category D, mandals upstream of the proposed Polavaram Reservoir; and Category E, mandals downstream of the link canal command area. Locations A and C will be directly affected by the project. Locations B, D, and E will be indirectly affected by the project.

The current existing cropping pattern in the command area is dominated by paddy, sugarcane and tobacco (Bhaduri *et al.*, 2007). Increased upstream development, especially through the construction of reservoirs and irrigation systems, has resulted in a decline in downstream flows, which has affected the cropping patterns in the Krishna Delta. When enough water is available, usually two rice crops are grown per year, but it has been observed that during dry years, only one rice crop is grown with another less water intense crop being grown during the rabi (dry) season (Dr Chandrashekhar Biradar, IWMI, personal communication). Once the link is built, it is proposed that paddy, sugarcane, chillies and pulses should be planted (GOI, 1999). Furthermore, irrigated crop intensity is expected to reach 150%. In the Godawari Delta, however, two paddy (rainy season) crops are grown per year, but only with supplemental groundwater use. In order to assess the benefits of the proposed Polavaram Project, two main scenarios were developed and simulated.

- Scenario 1: Reference Scenario: water use under the current supply and demand network. The water sources are groundwater and the river channel.
- Scenario 2: With the Polavaram Reservoir and link canal: water supply *versus* demand after the construction of the Polavaram Project. The water sources are the Polavaram Reservoir and link canal, groundwater and the river channel.

As 95% of the cultivated area is already under irrigation (Bhaduri *et al.*, 2007), it was assumed that substantial increases in new irrigated area will not be possible. Therefore, in the two scenarios, the agricultural land in the link command area was kept constant. Agriculture is still the major water user (<90%) compared to domestic and industrial demands, and increased agricultural production is the main goal of the Polavaram Project. Therefore, the anticipated benefits of building the Polavaram Reservoir and the link canal system are mainly based on the improved water supply and the subsequent increases in cropping intensity and yields. The effect of the Polavaram Project was tested by running the above two main scenarios under different crop rotation systems: (1) paddy-paddy, (2) paddy-pulses (representing a low water intensity crop), and (3) sugarcane only. Each crop rotation condition was run with and without environmental flow (EF) requirements/demands. The domestic, industrial and livestock water demands were kept constant in all runs. The scenario results were compared with each other and discussed in terms of unmet demands.

The starting point of the analysis was the development of catchment water demands. The demands in the study area are from agriculture, the domestic sector, industry, and livestock. Domestic water demand was given the first priority, followed by agriculture, industry and livestock, in that order. The supply sources built into the model were precipitation (for the catchments), surface water and groundwater. Simulations were conducted over the period from June 1991 to May 2005. The Polavaram Reservoir was simulated using the salient features published in the government feasibility report (GOI, 1999). According to this report, the link canal is designed to transfer 5325×10^6 m^3 of water per year. The environmental flows (EF) requirements have been estimated using the desktop method described in Smakhtin & Anputhas (2006).

Results of Scenario 1: Reference scenario with current water use

Under the current water use system, the average annual unmet demand for the period from June 1991 to May 2005 in the command area of the link canal is 1655×10^6 m^3 for a paddy-paddy system. Figure 2 shows the monthly average unmet demands aggregated for agriculture, domestic use, industry and livestock for the link command area. The unmet demands occur in all months except July and August (peak of the monsoon), and are for surface water as no further withdrawal from groundwater is possible. The maximum withdrawal rates from groundwater were based on the storage capacity and groundwater recharge rates for the area. Changing cropping patterns may decrease the unmet demands. For example, planting only one paddy crop during the rainy season and pulses (a low water intensity crop) during the dry season will decrease water deficits up to 51% (Fig. 2). As expected, giving EF a high priority in the water allocation scheme increased the

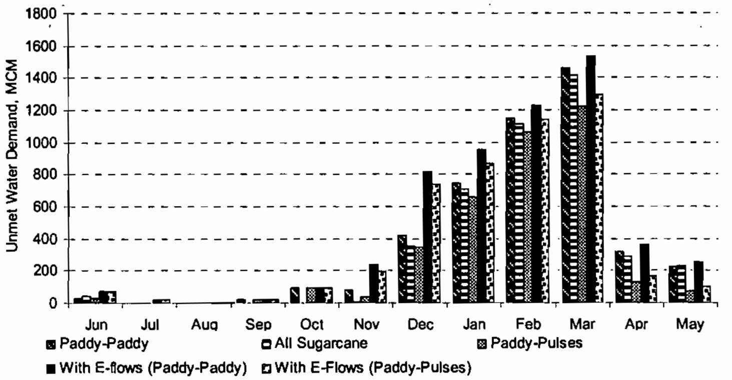

Fig. 2 Scenario 1: Monthly average (1991–2004) unmet demands from agriculture, domestic use, industry and livestock for the sub-watershed falling under the link command area.

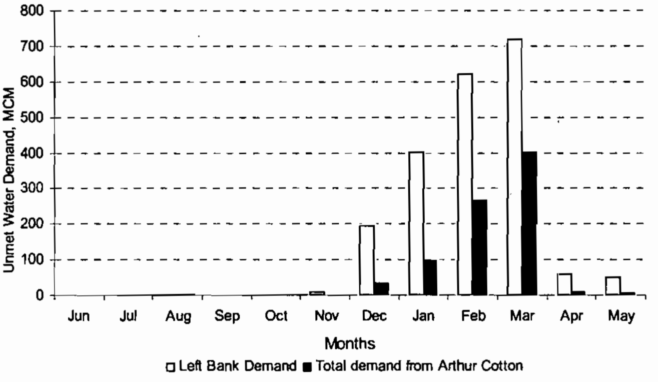

Fig. 3 Scenario 1: monthly average (1991–2004) unmet demands based on water requirements from Arthur Cotton Barrage and the Polavaram left bank.

unmet demands for other users (agriculture, industry, domestic). The unmet demands are highest for the simulation, which combines paddy-paddy rotation and EF requirements (Fig. 2).

Annual demands from the Arthur Cotton Barrage in the Godawari Delta are 8199×10^6 m^3 for irrigation and 378×10^6 m^3 for domestic and industrial use (GOI, 1999). Assuming these demands are coupled with a paddy-paddy cropping system, the mean annual simulated unmet demand for the command area of the Arthur Cotton Barrage would be 818×10^6 m^3. This constitutes 10% of the mean total annual demand. The water deficit in the Godawari Delta is in the rabi and dry seasons (December–May, Fig. 3). There is no deficit in the months from June to November. Similarly, 2057×10^6 m^3 mean annual unmet demand was calculated for the left bank command area in the Godawari. Similar to the Arthur Cotton Barrage command area, water deficit in the left bank command area is only in the rabi and dry seasons (December–May, Fig. 3).

Annual analysis for the Godawari showed that within the 14-year modelling period, the EF requirements are not met during the dryer years (based on rainfall data). Figure 4 illustrates that the unmet EF requirements are highest in June, when water demand for agriculture is high. The unmet EF plot seen in Fig. 4 is simulated with a paddy-paddy cropping pattern. Delays in the onset of the rainy season will affect water availability for EF. Paddy sowing was assumed to start in June, therefore, if the monsoon does not start in June, irrigation water demand will increase. The EF requirements are met from August to November.

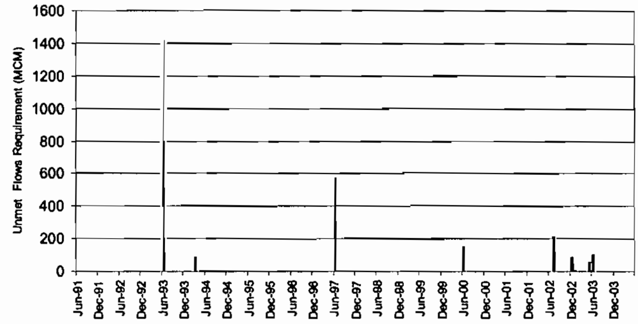

Fig. 4 Scenario 1: Unmet environmental water demand under current conditions with paddy-paddy cropping pattern (environmental flow requirement is given the highest priority).

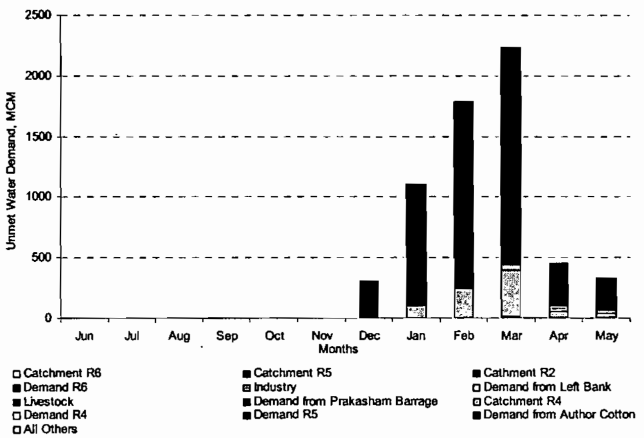

Fig. 5 Scenario 2: Monthly average (1991–2005) unmet water demands under paddy-paddy crop rotation.

Scenario 2: with the Polavaram Reservoir and link canal

The simulations with the link canal and reservoir show that within the link command area, there are minimal unmet demands for agriculture, domestic, and livestock requirements (Fig. 5). Figure 6 shows monthly average unmet demand (for 1991–2004) for agriculture, domestic use, industry and livestock for the link command area under different cropping patterns as well as with EF requirements. The unmet demands occur during the period from December to June and changing the cropping pattern to paddy-pulses almost nullifies the unmet demands, which exist under other crop rotations (Fig. 6). This is definitely an improvement for the link command area compared with Scenario 1. Introducing EF for the downstream of the Krishna and the Godavari, especially coupled with a paddy-paddy cropping pattern, increases the unmet demands during the months of January–June (Fig. 6). When comparing these values to Scenario 1 above, one can conclude that, although the water deficit situation improves within the link command area, if and when EF requirements are set, there will be a deficit in the link command area under a paddy-paddy cropping system.

The mean annual unmet demand for the left bank command area was 799×10^6 m^3 and the Arthur Cotton Barrage command area was 5270×10^6 m^3. Compared to Scenario 1, water deficit is smaller for the left bank command area, but higher for the Arthur Cotton Barrage command area, which is expected since water in the Godawari is being stored and diverted to the Polavaram command area. As with the current situation (Scenario 1), the water deficit in the Arthur Cotton Barrage command area is only in the rabi and summer seasons (December to May). The unmet demands situation for the Prakasham Barrage irrigation area shows improvement as there was no water deficit, with the exception of the year 2003, which was a particularly dry year. This water deficit occurs again only in March and can be alleviated by growing pulses or another lower water-intensive crop during the rabi season. Therefore, the analysis with the link canal (Scenario 2) showed that, although the pressure on water resources within the left and right bank command area reduces, there will be increased deficit in the Arthur Cotton Barrage command area. This deficit occurs, however, only during the rabi and summer seasons.

Analysis for the Godawari showed that within the 14-year modelling period, the EF requirements were not met during June in 1993, 1997, 2000 and 2003. In the simulation, EF requirements were set under a paddy-paddy cropping pattern where paddy sowing was set to start in June. Therefore, as the agriculture demands during this month are high, and if the monsoon rains that start usually in June are delayed, there will be unmet demands for agriculture as well as for environmental requirements. In both scenarios, June has the highest unmet EF for the Godawari. The storage in the Polavaram, is utilized within each year; therefore, in this case, the reservoir also does not provide water to compensate for delays in the onset of the monsoon rains. The EF requirement situation, which is more critical in the Krishna, does not improve after the link and water transfer, as most of the water that is transferred will be utilized for *en route* irrigation demands. In the Krishna, the highest unmet EF demands are also in June and July – at the start of the monsoon season.

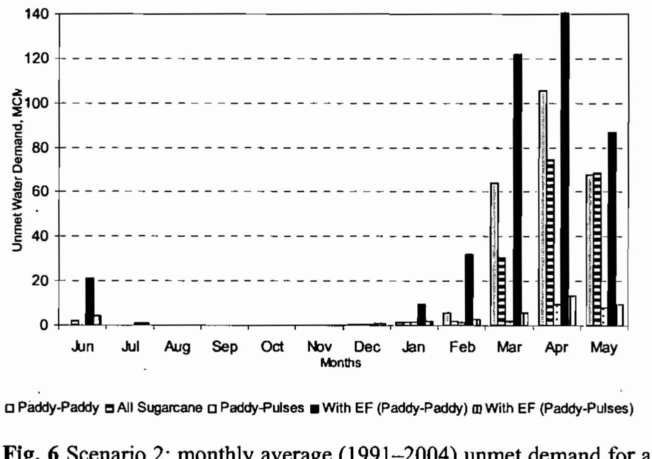

Fig. 6 Scenario 2: monthly average (1991–2004) unmet demand for agriculture, domestic use, industry and livestock for the link command area.

CASE STUDY 2: ASSESSMENT OF THE WATER TRANSFER SCHEMES IN THE UPPER BHIMA CATCHMENT

A major part of the water available in the Krishna basin originates from the humid regions of the Western Ghat Mountains where precipitation exceeds 5000 mm. The Upper Bhima and Upper Krishna catchments served by Western Ghats are therefore very important as they contribute significantly to Krishna River flows for downstream use (Immerzeel *et al.*, 2008). The Upper Bhima catchment is additionally important for the state of Maharashtra in the context of serving intersectorial water demands, including hydropower, agriculture, industry and drinking water supplies. Due to catchment development and increase in utilization, the water released to the main stem of the Krishna from the Upper Bhima catchment has declined by 59%. The Mulshi, Andhra and Tata lakes projects divert water westward out of the Krishna basin for hydropower production (Table 1). Therefore, these water transfers have been a source of tension as the other downstream states, which share the Krishna Basin, as well as irrigation projects downstream within the Maharastra state, claim that it aggregates the water scarcity problem in the basin.

In this case study, the impact of decreasing water transfer from the basin to the westward projects on the unmet demands of the downstream command areas within Maharastra as well as the outflow into downstream states were assessed with two scenarios:

- Scenario 1: Reference Condition – the water transfers westward are according to the current practice.
- Scenario 2: Decreased Transfers – the water transfers westward are decreased by 20%.

Although the cropping pattern in the Upper Bhima catchment consists of many food and commercial crops, three of the most dominant crops are sorghum, millet and sugarcane.

Results from scenarios 1 and 2

As can be seen from Fig. 7, which presents average monthly unmet demands for the years 2000–2004, the water demands are met during the monsoon months (June–September). The end of the dry season, i.e. April and May, are when the unmet demands are highest. Figure 8 presents unmet demand calculations with 20% reduction in the westward diversions. As can be seen from Fig. 8, even with 20% reduction, the unmet demands remain similarly high for the dry months of April and May. Therefore, a better management alternative would be to reduce the transfers during April and May. However, a cost–benefit analysis on the productivity of the trade-off between transferring water for hydropower and reserving it for agricultural use should be done. Social and equity issues should also be considered. Changing the cropping patterns and planting less water-

Table 1 The salient features of projects diverting water westward out of the Krishna basin to the Upper Bhima.

Project	Purpose	Live storage ($\times 10^6$ m^3)	Gross storage ($\times 10^6$ m^3)	Power potential (MWH)*
Mulshi	Hydropower	523	554	150
Andhra	Hydropower	353	354	72
Tatalakes	Hydropower	265	273	72

* Megawatt hours.

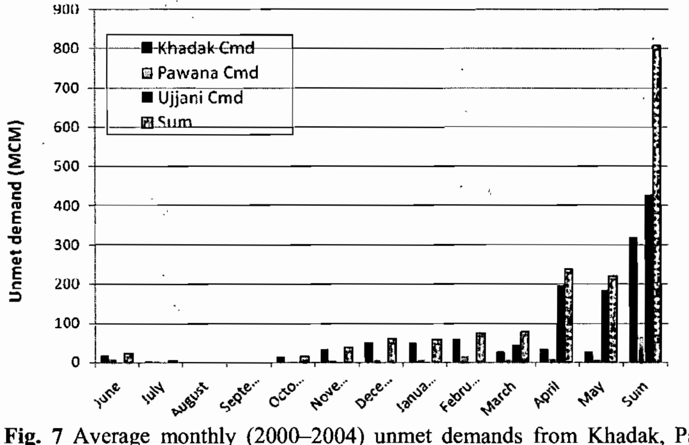

Fig. 7 Average monthly (2000–2004) unmet demands from Khadak, Pawana and Ujjani command areas with westward diversion and actual cropping patterns.

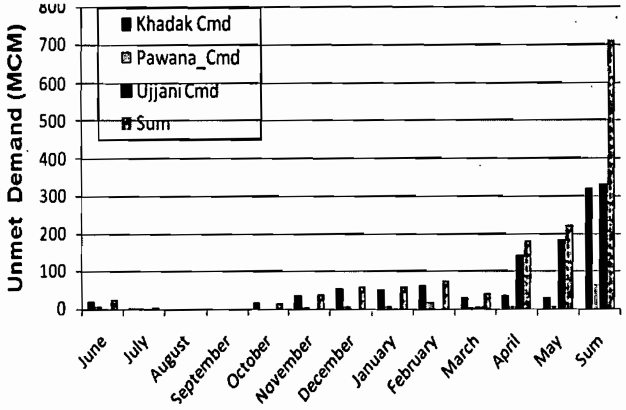

Fig. 8 Average monthly (2000–2004) Unmet demands from Khadak, Pawana and Ujjani command areas with 20% reduced westward diversion and actual cropping patterns.

intensive crops in the summer months would also be another way to manage the water scarcity in these months.

Furthermore, decreasing the westward transfer by 20% will not necessarily increase the flows downstream from the outlet of the basin, as the extra water will probably be used for agriculture and other purposes within the basin. Therefore, if outflow from the catchment is to be increased, then reservoirs have to release water to the river and not use it for irrigation.

CONCLUSIONS

This study suggests that water resources management in the region has to be done on a seasonal basis by taking monthly variability into consideration. Inter-basin water transfers have been an integral part of water resources management all over the world. However, without careful integrated planning and analysis, the proposed high-investment schemes might not be able to operate as planned and eventually might not deliver the expected long-term benefits.

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Integrated surface water and groundwater resources management in Makueni District, Kenya

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Abstract Makueni District is one of the arid and semi-arid areas in Kenya falling under agro-ecological zone IV-V, and occupying a total land area of 7965.8 km^2 representing 0.014% of Kenya's land cover. The district receives rainfall in two seasons with a rainfall pattern similar to the national rainfall pattern. Rainfall contributes significantly to the availability of water resources in the district. The water resources mainly comprise the few perennial rivers traversing the district and the numerous seasonal streams cutting across deep valleys in the district. The major rivers include the Athi River with its various tributaries of Muooni, Kiboko and Kibwezi, all of which become seasonal in places. Groundwater resources are scanty due to the nature of the basement rocks. The only significant sources are found within the volcanic systems but the yields are relatively low and the water is somewhat saline. Certain areas have springs that contribute enormous amounts of water and include: Simba, Kiboko, Umani and Kibwezi springs. Water from these sources is utilized for horticultural production, domestic water use and livestock watering. These activities constitute the main economic undertakings by the populace. However, due to increasing population and re-distribution in the district, the available water resources are unable to meet the development goals of the district, as well as afford a sustainable use of the available water resources in the district. This paper examines the Integrated Water Resources Management (IWRM) strategies in addressing exploitation and development challenges in the district, and offers options in the management strategies of the available water resources in relation to population dynamics, development strategies and sustainable environmental conditions.

Key words integrated strategies; water resources and development options; Kenya

INTRODUCTION

Makueni District, and by extension the larger Machakos District, lies within the semi-arid areas of Kenya. The district occupies a land area equivalent to 7965.8 km^2, representing 0.014% of Kenya's total land cover. It borders the districts of Machakos to the north, Kitui to the east, Kajiado to the west and both Tsavo National Park and Taita Taveta district to the south, as indicated in Fig. 1. The district is generally semi-arid, with a few humid land-mass pockets. The semi-arid areas, located in the southern parts, receive a mean annual rainfall of between 550 and 700 mm while the humid zones, characterized by the hill masses of Kilungu and Mbooni, receive 700–850 mm per annum. In terms of aridity, the district is in category C with 50–85% aridity conditions according to the Republic of Kenya Environmental Action Plan for Asal (1992). This paper examines the cross-cutting issues relating to the exploitation of water resources, environmental conservation and management, and sustainable development applicable in the district. The results of this paper can be replicated in other similar arid and semi-arid districts in the country, as well as laying sound environmental conservation and management policies which are sustainable.

PHYSICAL CHARACTERISTICS

Machakos District, and to a large extent Makueni District, occupies an area approx. 14 250 km^2, with the land rising from slightly below 600 m m.s.l in Tsavo at the southern end of the district to above 1000 m m.s.l in Yatta Division and 1600 m m.s.l in the northern outskirts of Nairobi Province (Machakos, DPP 1994–1996). According to Dodson (1953), the geological foundation on which the present-day relief of the entire district rests, falls into three major rock types, namely: the Precambrian Basement system, Tertiary volcanics and Pleistocene volcanics, as shown in Fig. 2. The Precambrian Basement system covers the greater part of the district, which includes the hill masses of Mbooni, Kilungu, Mbitini and Nzaui, which are formed of granite rocks. The

Tertiary volcanic system includes the Miocene phonolites covering the Kapiti plains and the Yatta Plateau. The last geological system is the Quaternary volcanic activity (Pleistocene basalts), which is associated with the formation of the Chyulu ranges and extension of basalts rocks covering parts of Kibwezi, Makindu and Mtito Andei divisions.

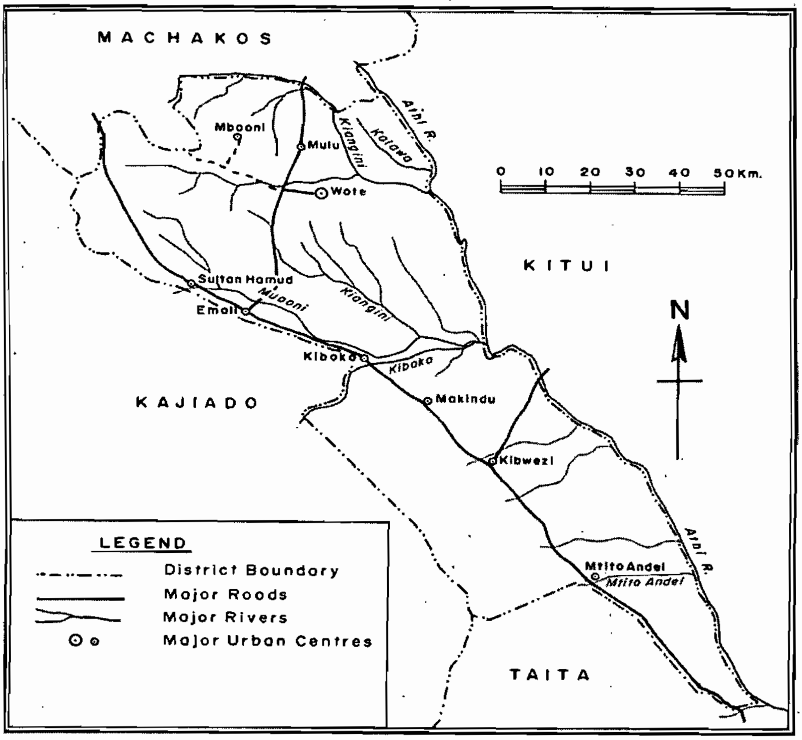

Fig. 1 Makueni District and bordering districts.

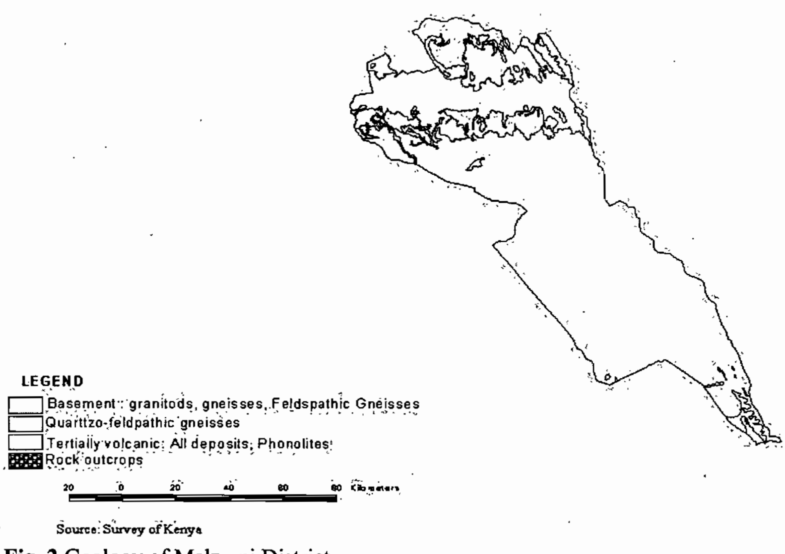

Fig. 2 Geology of Makueni District.

The district falls within the zone of arid and semi-arid lands, and is characterized as an area of extreme variability of rainfall. Tiffen *et al.* (1994) state that rainfall in the district is characterized by small total amounts, strong seasonal distribution and high temporal and spatial variation from year to year and from season to season, while Stewart and his collaborators (Stewart, 1991) developed a method of "response farming" which under experimental conditions allowed the optimal management of variable rainfall. Soil erosion by water is the most conspicuous form of land degradation in the district, with the main forms as inter-rill, rill or gully erosion. In physiographic terms, the district can be referred to as flat plain land with scattered hills and rock outgrowths as a result of prolonged soil erosion. The most noticeable hill masses are the Kilungu and Mbooni rising approx. 2500 m above sea level in the northern tip and the Chyulu hills to the southern part. These areas have been affected by volcanic activities of the distant past and fertile volcanic soils support agricultural activities, as well as contributing significant amounts of groundwater flow to the main perennial river, the Athi River. The plains support large scale ranching activities and are a source of meat to the Kenyan market, in addition to being a major source of income to the people.

POPULATION DYNAMICS

According to the latest population census of 1999, Makueni District had a total population of 771 545, as compared to the previous population census of 1989 when the population was 636 996, with a projected population of 912 689 by the year 2009. The number of females according to the last census was 398 906 and males were 372 639, as shown in Table 1. Of 771 545 people in the district, 353 744 (slightly over 50%) are categorized as economically inactive implying high dependency ratio (Republic of Kenya, 2001).

Table 1 The population density per division in Makueni District.

Division	Area (km^2)	Males	Females	Total	HH	Density
Tulimani	126	15 354	17 353	32 707	6 301	266
Mbooni	141	26 348	29 635	55 983	10 331	395
Kisau	301.2	23 804	26 706	50 510	9 224	168
Kalawa	330	12 673	13 660	26 333	4 357	80
Kilome	359.4	22 319	23 885	46 204	8 631	129
Kilungu	178.3	31 556	36 185	67 741	12 740	380
Kaiti	239.8	22 052	24 055	46 107	8 529	192
Kasikeu	2709	17 370	18 349	35 719	6 852	132
Mbitini	229.7	23 411	25 318	48 729	8 947	212
Wote	362.7	20 092	20 261	40 353	7 744	111
Matiliku	240.5	18 333	20 534	38 867	6 893	162
Kathonzweni	886.7	31 397	34 341	65 738	10 798	75
Nguu	350.3	9 529	9 722	19 251	3 345	55
Makindu	880.2	24 917	25 382	50 299	9 907	57
Kibwezi	944.8	39 797	40 441	80 236	16 282	85
Mtito Andei	931.2	33 601	33 062	66 663	13 354	72
TOTAL	7966	372 639	398 906	771 545	144 320	97

Source: Adapted from Wambua (2008) and Population Census (1999).

Mbooni, found in Agro-ecological zone 2 (AEZ 2) is the most densely populated division with 395 people per km^2 by 1999 compared to 333 people per km^2 in 1987 as shown in Table 1. This division has high population pressure, which is leading to land fragmentation. Nguu Division, which was recently curved from Makindu Division is the least densely populated area with 55 people per km^2. Nguu Division lies within AEZ 4-5. The high population found in Mbooni and

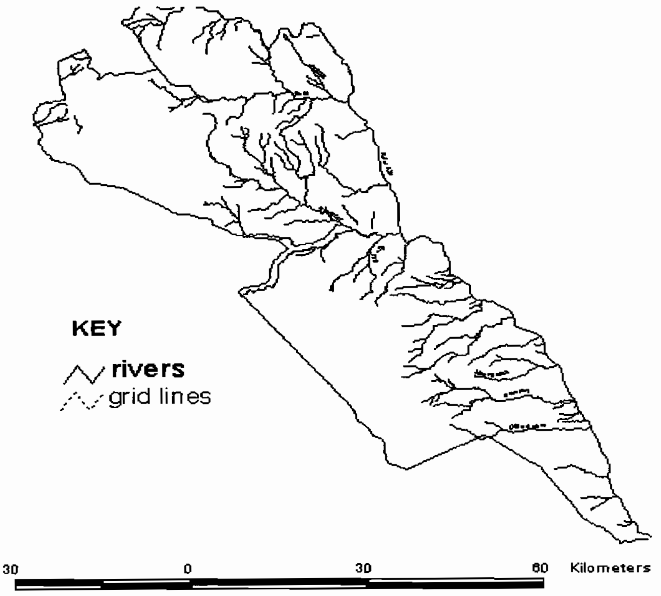

Fig. 3 Drainage systems and water resources.

Kilungu divisions is attributed to good, reliable and effective rainfall, as well as high potential in soils, while low population found in Nguu, Kathonzweni, Makindu and Kibwezi is due to low land potential as a result of unreliable rainfall and poorly developed soils.

The rapid increase of the district's population is partly attributed to immigration of people from neighbouring districts onto the "settlement schemes" in the district. These newly opened settlement schemes are in Kibwezi, Makindu, Mtito Andei and Nguu divisions (Kibwezi, Kiboko, Nguu, Masongaleni and Mikululo settlement schemes). The majority of the people moving to the lowland areas are from Mbooni, Kilungu, Mukaa, Kasikeu and other parts of the district where land is constrained (Republic of Kenya, 2001).

WATER RESOURCES

Water is perhaps the most valuable and yet the most scarce resource of the arid and semi-arid areas (Asals). Sources include groundwater and surface water, which vary greatly both spatially and temporally. Where available, the water source points have been developed and in the process have become the focal points of environment degradation as well as community conflict areas. The main water resources in Makueni District include; the main Athi River with its various perennial tributaries of Kiboko, Muooni and Kibwezi, together with a number of other seasonal streams as indicated in Fig. 3. The Athi River occupies a total land area of 70 000 km^2 with a mean annual rainfall of 585 mm representing a mean annual runoff of 19 mm. The river has a combined annual mean discharge of 8.8 $m^3 s^{-1}$ in the headwater areas increasing to 23.6 $m^3 s^{-1}$ in the middle areas and 33.6 $m^3 s^{-1}$ at the mouth (Malindi) (MOW, 1992). The main groundwater sources include the springs of Simba, Kibwezi, Umani and Mzima, with annual mean flows ranging from 0.15 to 0.4 $m^3 s^{-1}$ together supplying substantive amounts of groundwater resources. The other water source is rainwater which is normally harvested and stored in tanks for domestic water supply or collected as runoff into earth dams for small-scale irrigation, domestic water use and livestock watering. Makueni District reported scarcity of safe water exceeding 80% in the poverty report done by the Ministry of Finance and Planning (2000). Moreover, the increased number of stakeholders demanding water, the decreasing volume of good quality water and the scarcity of land, call for accurate and often small-scale water management strategies. The Athi River, which is the main perennial river, has a relatively high annual runoff rate of 19 mm representing 11 402.57 million m^3 $year^{-1}$ (Kithiia & Ongwenyi, 1996), making up the bulk of the surface water resources in the district. Groundwater potential is highly variable in both quantity and quality and is based on shallow water sources which are affected by variations in rainfall and river flooding. In addition, the groundwater resources have been adversely affected by the process of sand harvesting in most of the seasonal rivers by reducing the groundwater recharge potential. This has to some extent affected both the agricultural and economic activities within the district, as well as contributing to increasing environmental degradation in the district.

WATER RESOURCES AND DEVELOPMENT ISSUES

Water is an important natural resource for human living, production, and societal development. River basins with abundant water of high quality, together with the convenience for utilization, are historically preferred places for human habitation and development (Chen *et al.*, 2008). Today, unwise use of water resources has led to many problems, such as eco-environmental damage, water pollution, and abnormal hydrological cycles among others. Thus, water resources utilization should be well planned for sustainable development and environmental management particularly in a vulnerable environment such as in Makueni District.

The available water resources in Makueni District (both surface and groundwater) have been extensively developed for domestic water supply, irrigation (horticultural production), and livestock watering. However, horticultural production is widespread in most parts of the district. Horticultural crops of significance include; citrus fruits, e.g. oranges, lemons and mangoes. Other crops are brinjals (egg plant), Karera, paw paws, cucumber and hot chillies for both local and export markets. These crops are grown under open irrigation systems – mainly channel and furrow irrigation methods. Due to the rate of evaporation in the district, a lot of water is lost, sometimes leading to crop failure due to water shortage and other associated environmental problems.

SUGGESTED WATER MANAGEMENT PRACTICES (BMPs)

Due to the climatic conditions experienced in the district and limited available water resources coupled with soil characteristics, Makueni District has limited development options that relate to the use of water resources. The most viable and practicable option would be collection of runoff water and damming of the various seasonal rivers to augment the available surface water resources. The groundwater resources from the springs of Simba, Kiboko, Umani and Kibwezi, with an annual mean flows ranging from 0.15 to 0.2 m^3 s^{-1}, provide the most reliable sources of water. These springs draw their water from the Chyulu Hills volcanic system in the northwestern side of the drainage basin. This water is used for domestic water use and livestock watering, as well as for small-scale irrigation. This in turn, can be used to expand on the land hectarage under irrigation with a focus on horticultural crops with a high demand in the urban areas and for export markets. Rainwater harvesting option is constrained by the exorbitant cost of constructing water tanks and lack of proper house roofs, and is therefore limited to domestic water use and not for irrigation purposes. Makueni District inhabitants can be encouraged to diversify their crops with a focus on drought resistant cash crops such as cotton, sunflower and sisal, as well as drought resistant food crops, namely sorghum and millet, which require less water. The fact that, the district is one of the semi-arid districts with a high probability of rainfall (500–900 mm) and rain-fed crop failure, many people have tended to preoccupy themselves with charcoal burning, resulting in severe deforestation and environmental degradation. This, in turn, is a contributing factor to crop failure, shortage of rainfall and heavy reliance on food relief (food aid) from the central government and other organizations. There is therefore a need for change of attitude and perception among the inhabitants and policy makers.

There is a need for concerted efforts towards soil and water conservation practices, focusing on farm terracing, contour ploughing, strip cultivation, furrow systems and cut-off drains. In addition, the government, local authorities and communities should work together in controlling sand harvesting along the river beds to allow for groundwater recharge. This will go along with promoting agriculture, environmental conservation and provision of adequate water for both domestic and livestock use. Increased efforts towards environmental conservation are most likely to increase the rainfall infiltration within the basin, and hence the groundwater recharge rates and further the available water resources. This is likely to augment both the surface and groundwater resources.

People should be integrated in the decision making in regard to the use of the water resources in the area, as well as in the conservation and control of water quality in the rivers close to the small-scale irrigation schemes spread over the district and the basin. This will ensure safer use and quality maintenance of the waters in the entire drainage basin and the district as a whole in trying to achieve the situation advocated by Tiffen *et. al.* (1994), the "more people less erosion" scenario.

CONCLUSIONS

The study found that there is a need to put more emphasis on the conservation and management of the meagre water resources in the district. The strategies should be on the projects that allow augmentation of surface water resources, as well as allowing for increased groundwater recharge rates. These strategies include both water and environmental conservation such as practicing terracing, strip cropping, mulching, and drip irrigation. In addition, sand harvesting should be discontinued in the many river basins in the district to allow infilling of river beds and to increase groundwater recharge rates.

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Managing space for water in regional integrated water resources planning in China

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Abstract More and more attention is being paid to Integrated Water Resources Planning (IWRP) and Management (IWRPM) within a jurisdiction region in China. This paper focuses on the principles and procedures of drawing up projects to manage space for water in a sustainable manner in regional IWRP based on finished practical applications in several Chinese river-net areas in the Yangtze River delta. By analysing the functions of regional space for water, projects proposed in IWRP for management purposes include those of preservation, protection, restoration, water conservancy and waterscape, among which natural wetland is to be preserved, while space for seriously polluted water needs to be restored, spaces for supplying drinking water source and nature reserves are to be protected, and space for water conservancy projects including flood control works, water supply and drainage projects, etc. are to be developed while space for waterscape is reserved to lay out natural and artificial waterscapes. To make projects and IWRP scientific, the functions of space for water are analysed within a region, with factors such as population growth, urban planning, land-use plan, regulatory requirements, temporal and spatial scales are considered systematically and water eco-region zoning is partitioned. A case study is illustrated in Jiangyin county.

Key words projects for management purpose; space for water; regional water eco-region zoning; IWRP; IWRM

INTRODUCTION

Space for water is one kind of space that encompasses the river and its branches, linked lakes, wetlands and other kinds of water bodies, flood plains and partial slopes that are directly affected by flood, as well as the corresponding aquifer systems within these areas. Space for water plays an important role in keeping the integrity of systems of water resources, ensuring water cycle and regeneration, and realizing sustainable development of water resources (Deng Wei *et al.*, 2003).

Human activity has led to dramatic changes to the space for water in the river-net area of China and which is now far from the original one and shows man-made characteristics: (a) High artificial degree: accompanying rapid urbanization, functions of space for water have been exploited excessively, one-sidedly and simply. Not only is some natural space for water reshaped, but also much artificial space for water, such as ancient canals and canalized natural space for water is excavated. (b) Filled space for water: along with the high artificial degree, a lot of water conservancy projects such as flood gates, sluice gates, dams, pump stations, culverts, and weirs are set on space for water. Some small and medium-sized space for water has been filled up in order to get more construction sites. (c) Seriously polluted water: most water area is polluted seriously, and the waterfront is even used to pile up rubbish. Consequently, the functions of space for water such as water plant and animal conservancy habitats, landscapes, and ecological recreation have been completely lost. Human activity harms the sustainable supply of water ecosystem services since loss of space for water changes the processes of the water cycle and greatly aggravates the ecological equilibrium. Inspiringly, more attention has been paid to the situation and research on the functional management of space for water have been launched.

At the international level, the harnessing of space for water has experienced a process from early construction of flood control dams and cutting-off rivers to initiating recovery of damaged river systems. In the early stage, a key problem of water management is pollution control, and then the experts turn their attention to the relationship between water pollution and aquatic organisms which live in the water and at the water margins, and begin to adopt ecological theory to carry out comprehensive control which takes the pollution control as a core and considers the dredging and layout of space for water. In China, research on the management of space for water began in the 1980s. At present, the terminal control method is mainly used to take the tertiary control into account, which contains the idea of "decrease of source, interception and repair of the river" and is gradually permeated (Guo Huaicheng *et al.*, 2007). The practices of space for water utilization and conservation are carried out mainly relying on the experiences of years of water conservancy work and one-sided requirements on space for water. While over the short term these programmes may be rational with respect to public or private policy objectives, over the longer term many result in both economic inefficiency and the erosion of natural services (Balmford *et al.*, 2002). While some goods or services, such as flood control, etc., are supplied, some other functions may be restricted as the tall flood walls built for controlling floods make many large obstacles between humans and water, and hamper the application of recreational functions. So the functions may be incompatible with each other, and it is necessary to draw up projects of space for water in order to realize sustainable utilization of water ecosystem services through scientific planning. Projects of space for water are made in regional IWRP, and it is the managing action of space for water.

The relationship between IWRP and IWRM

Water is an integral part of an ecosystem and a natural resource, as well as a social and economic good; according to Loucks (2000), "Sustainable water resources systems are those designed and managed to fully contribute to the objectives of society, now and in the future, while maintaining their ecological, environmental and hydrological integrity", so it needs to be managed synthetically. IWRM usually goes after IWRP in which water related projects are arranged. IWRP is an integration of socio-human factors, economic issues and the ecological system which means that the society will continue to benefit from the utilization of the water resource while maintaining the environment and the resource base to meet the needs of future generations; therefore, attention should be paid to ecological, social and economic aspects of water in IWRP. "Integrated Water Resources Management is a process which promotes the coordinated development and management of water, land and related resources, in order to maximize the resultant economic and social welfare in an equitable manner without compromising the sustainability of vital ecosystems" (GWP, quoted from Jonker, 2002). Unlike those things that cannot be managed such as rainfall, wind and other natural processes, people's activities can be managed. Therefore a more suitable definition of Integrated Water Resources Management would be: managing people's activities in a manner that promotes sustainable development (improves livelihoods without disrupting the water cycle) (Jonker, 2002). IWRM seeks to manage the water resources in a comprehensive and holistic way that consider the water resources from a number of different perspectives or dimensions; engineering, economic, social, ecological and legal aspects need to be considered, as well as quantitative and qualitative aspects, and supply and demand. Once these various dimensions have been considered, appropriate decisions and arrangements can be made (Savenije & Van der Zaag, 2008). Moreover, the "management cycle" (planning, monitoring, operation and maintenance, etc.) needs to be consistent.

Management projects of space for water in regional IWRPM

The increasing complexity of regional economies and the increasing degree of interrelationships between segments of society virtually require that plans for the water resources sector be consistent with and complementary to plans being developed for other sectors of the human endeavour (US National Water Committee, 1966). There is an urgent need for multi-sectoral planning. Regional IWRP lays particular emphasis on functions of water which meets specific needs of different sectors and implementation. It is the investigations of a specific structural or non-structural measure in sufficient detail to determine whether it will meet established goals, objectives, and criteria and if its implementation is possible within the estimated costs and within the limits of financial feasibility (Matondo, 2002). It is the broad evaluation of alternative measures for meeting hypothesized goals and objectives, with recommendations for action plans and programmes.

Management projects of space for water are the managing actions of space for water, of which the aim is terminal implementation for management. The functions of water, key field and preferential management areas and implemental action are determined through projects of space for water including water, transition and riparian areas. Functional demands for water reflect the distinction between human and nature which is applied to make projects and actions as planned in IWRP, and the projects can be carried out conveniently through the functional management of space for water.

Sometimes, water eco-region zoning needs to be made prior to drawing up projects of space for water due to regional differentiation. The aim of water eco-region zoning is to reflect the spatial distributing laws and properties of the water ecosystem. Since it is made in a natural region on most occasions, the methodology of water eco-region zoning in a jurisdiction region is different, especially in the factors considered during zoning.

Overall, management projects of space for water is based on regional IWRP through which regional IWRM is implemented.

METHODOLOGY

Analysing human demands for regional space for water

A healthy water ecosystem provides a series of ecosystem services and functions which can be classified into economic, ecological and socio-cultural values, according to their different value features. The functions of economic value include water supply, transportation, flood regulation and production; functions of ecological value include purifying, climate regulation, habitat and hereditary resource; and functions of socio-cultural value include recreation, aesthetics and culture, science and education. Among these functional services, some are for human requirements and others are for the water's demands. Hence the essence of regional IWRPM is to coordinate the demands of people on the water ecosystem. During the process of urban development and human activities, some functions become predominant and overused while others are ignored, restricted or damaged. Generally, the evolution of its capital function is expanding from flood control, freshwater supply to purification and recreation, and present capital functioning services are analysed and future requirements are anticipated. The sequence of importance of each demand needs to be ranked to determine a predominant function of space for water. Other functions are ranked on the premise that predominant services are normally functioned. For example, functions of supplying drinking water sources and regulating storage are reckoned as predominant and will be strictly retained, while functions that provide ecological and recreational services should be considered in the best possible way since they become more and more important to citizens. From an economic perspective, values of the waterscape depend upon the capacity of the system to ameliorate negative impacts of human activity or support productive aspects. Thus, the same waterscape would be valued differently depending on the presence of humans and the need for specific services.

In essence, ranking of the important sequence of water functions is to seek the equilibrium between the supply of all the functional services of space for water and the demands of mankind.

Making eco-region zoning for regional IWRPM

Analysis of functions of space for water related to regional IWRP is mainly for the management purpose, so it is made before IWRP and implemented during IWRM. Projected measures and non-projected measures will be proposed in IWRP, so space for water may be classified into different water eco-region zoning and functional space according to projected and non-projected measures. A water eco-region is a land unit which possesses a relatively homogeneous freshwater ecosystem or bio-organism as well as its environment, and it evidently shows the integrity and spatial differentiation of space for water. Therefore, it can help the manager to implement IWRM of the same ecological feature and resource property. Since water eco-region zoning is a complex task and it is always made at a rather large scale, the process at the regional jurisdiction scale may be partitioned according to regional natural water characteristics, human utilization history, socio-economic development degree, spatial layout of industries, urbanization ratio and anticipated requirements.

Classifying the functions of space for water according to IWRP

To implement IWRM properly, realize socio, economic and environmental functions of regional water and maintain a healthy water ecosystem, projects of space for water are divided into five classes according to their functions as projects of preservation, protection, restoration, water conservancy and waterscape, and each one can be further projected.

Natural wetlands need to be preserved while space for seriously polluted water needs to be restored. Drinking water source, river bank, nature reserve and flood defence work areas are to be protected. Space for waterscape is reserved to layout natural and artificial waterscapes. Space for water conservancy projects usually includes developing flood control works, water supply projects, drainage projects and water dispatching projects, etc. The aim of a water conservancy project is to develop comprehensive functions and benefits of water engineering facilities featured by benefit realization and risk aversion through changes of the operation of existing or proposed future water facilities so that present water resources can be adequately used, river quality can be improved, and water ecology, environment and scenery can be restored, improved and protected and in turn to ensure a sustainable water resource development.

Procedures

The main procedures of drawing up management projects of space for water are as follows:

1. make on-the-spot investigations to find out present utilization type and improper use of space for water and analyse human impacts on it;
2. anticipate human demand for water according to factors such as population growth estimates, urban plan, land use plan, climate variability, regulatory requirements, temporal and spatial scales, socio and environmental effects;
3. make regional IWRP to lay out projects on space for water and eco-region zoning; then make management projects of space for water according to IWRP;
4. analyse the implementation possibilities of the proposed projects for IWRM.

CASE STUDY

Area description

A case study is briefly examined in Jiangyin City which is located in the Yangtze River delta with 988 km^2 area. Jiangyin is among the best of the national 100 powerful counties and possesses a huge wealth of space for water occupying 12.8% of the surface area. The urbanization area is mainly distributed in the central eastern part of the city. There has been extensive urbanization since 1980s. This growth continues and at current projections the population is predicted to rise from the present 120 million (2007) to 217 million by 2020. The urbanization rate was about 39% at the end of 2007, and is projected to be 69% by 2020. The conversion rate of rural into suburban land cover will increase to 198 km^2. To meet increasing demands for functions of water, the authority of Jiangyin decided to make IWRP and implement IWRM.

Problems

Jiangyin has many advantages regarding its water resources, such as: water surface ratio is relatively high, plenty of wetland resource is possessed, water quality is good in part of the area, underground water level is well preserved, but there are still several disadvantages about its water. For example, water quality is deteriorating, the water surface ratio is declining, the biodiversity is lessening, the waterfront green land corridor is insufficient, non-point source pollution is increasing year by year, water and soil erosion by rock exploitation is severe, pressure of water eco-environment protection is enlarging and projects harmful to ecosystem have been constructed on some rivers.

RESULTS

Water eco-region zoning

The development degree of the economy and the quality of eco-environment of Jiangyin shows evident spatial differences. For convenient making of projects and implementation of IWRP, Jiangyin City is partitioned into five water eco-regions: central city, western, eastern, southeastern and southern Jiangyin, according to the natural characteristic of its water system, utilization history, social-economic development degree, spatial layout of industries and anticipated requirements (Fig. 1). The main functions, industrial layout, problems, tasks and measures of each sub eco-region are illustrated.

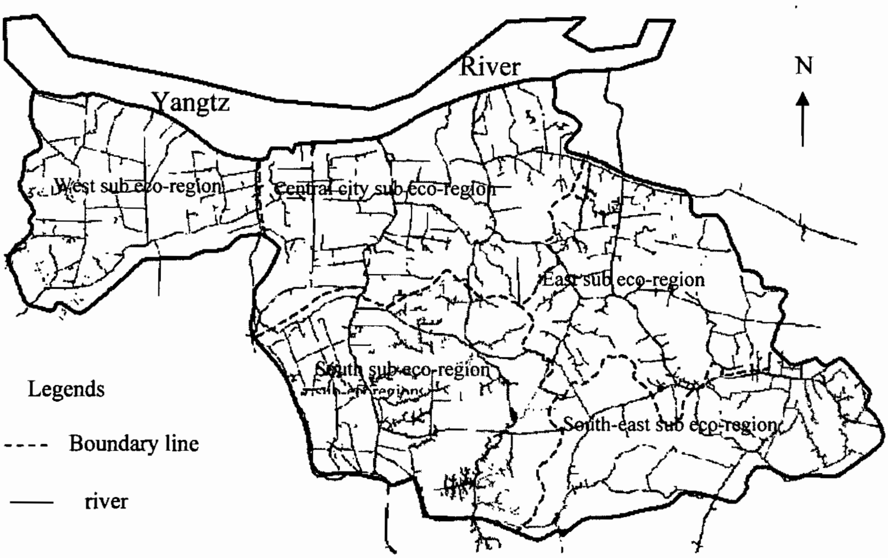

Fig. 1 Diagrammatic sketch of water eco-region zoning result of Jiangyin.

The goal of the central city sub eco-region is to coordinate the need between economic development and water ecosystem health; the western sub eco-region's goal is to minimize the impact of economic development and population growth on space for water; the eastern sub eco-region's mission is to become a coordinated area between economic development and protection of water functions; the assignment of the southeastern sub eco-region is to reduce the harmful impact from upstream rivers and adjust the industrial structure; and the goal of the southern sub eco-region is to preserve the natural water ecosystem, minimize the impact of human activities and develop eco-agriculture as well as an eco-tourism industry to harmonize the relationship between humans and water.

Management projects of space for water based on IWRP

Following IWRP and the results of eco-region zoning, projects of space for water in Jiangyin have been proposed (Fig. 2).

1. Eight key preservation, protection and rehabilitation areas are projected: (a) a drinking water source protection area on the Yangtze River including Shiqiao and Xiaowan, which extend 4500 m and 8000 m along the river bank respectively; (b) a preservation area for dispatching clean water, i.e. a clean water corridor preservation area on Baiqugang River with 10–60 m width on each bank. It is to be constructed as a blue and green corridor which passes through

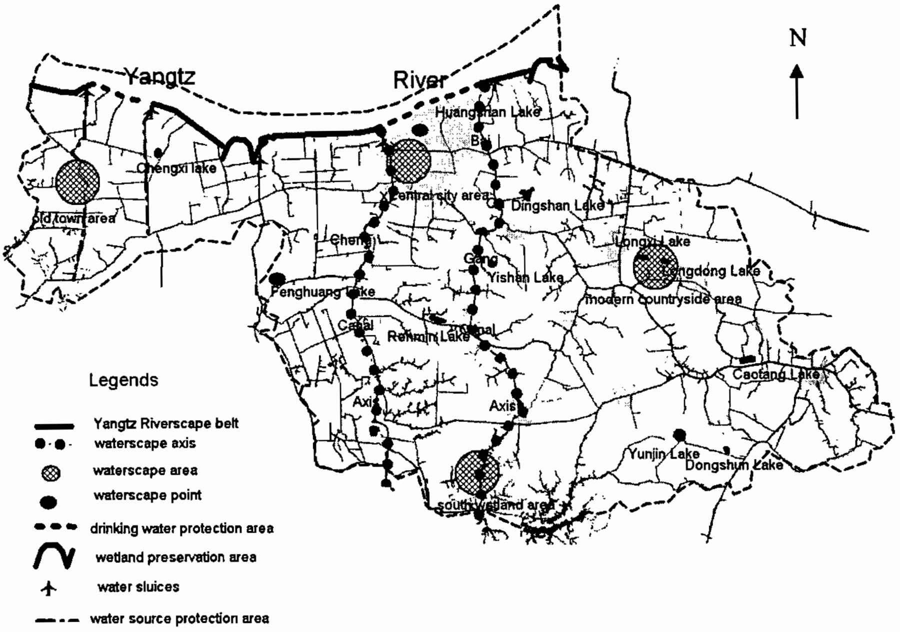

Fig. 2 Diagrammatic sketch of the results of management projects of space for water in Jiangyin.

the whole city; (c) a wetland preservation area for the Yangtze River including two different sections; (d) a wetland protection area for the inner rivers in Mazhen, which are of great importance for their ecology, landscape and eco-tourism; (e) a water source protection area for flood combat readiness including nine existing and planned lakes; (f) a water rehabilitation area in inner cities including the famous city moats; (g) a natural water system protection and preservation area of countryside; (h) water source protection area of inner rivers including the Ligang and Xingou water courses, which are two main rivers flowing into Yangtze River, whose water quality affects that of Yangtze River directly.

2. Water conservancy projects including water sluices, new-blown space for water, water plants and sewage treatment works. Five water sluices flowing into Yangtze River and 30 inner water sluices are planned, eight new lakes are to be developed, two water plants are to be extended and a new water plant and eight sewage treatment works are to be built.
3. The pattern of the waterscape projects are designed as "one landscape belt along the Yangtze River, two principal landscape axes of Xicheng canal and Baiqugang channel; four accumulated waterscape areas, and 11 waterscape points to be built on the existing and new lakes.

CONCLUSIONS AND RECOMMENDATIONS

IWRM usually goes after IWRP in which water related projects are arranged within a jurisdiction region. In this way the mutual dependence between sustainability and integration becomes a guiding principle for developing management projects of space for water. The five classes of projects of space for water are: preservation, protection, restoration, water conservancy and waterscape, and each one can be further specified. Regional water eco-region zoning is made according to water natural characteristics, water utilization history, socio-economic development degree, spatial layout of industries, urbanization ratio and anticipated requirement for the implementation of IWRPM.

The projects of space for the water of Jiangyin, designed by using the methods and procedures presented above, include protection projects for drinking water sources on the Yangtze River, water in reserve, the river system with historic culture in old downtown, preservation projects for diversion of clean water from the outer river system and wetland, waterscape layout projects and water conservancy projects such as water supply and drainage, sewage treatment works, sluices and levees.

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Keyword index

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