WaterStrategyMan

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DELIVERABLE 7

METHODOLOGY REPORT ON THE QUANTITATIVE ANALYSIS OF WATER SYSTEMS

AVAILABLE METHODS FOR ESTIMATING WATER SUPPLY, RESOURCE AND ENVIRONMENTAL COSTS

MULTI CRITERIA APPROACHES FOR ASSESSING WATER RESOURCES SYSTEMS

EXISTING INDICATOR APPROACHES FOR WATER RESOURCES ASSESSMENT

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1 Introduction

The overall objective of this deliverable is to present and assess available methods for the quantitative analysis of water resources systems in terms of

a) Water resources and water infrastructure systems,

b) the estimation of economic and environmental costs and

c) multi-criteria decision making in water resources management.

In addition, it is aimed at describing and evaluating a number of indicators to describe water resources systems with respect to the dimensions mentioned above.

This document is organised in four sections. Section I reviews methods and tools for water management analysis, section II presents a critical overview as well as a proposal for a consistent methodology for an economic assessment of water resources systems. In section III, a recommendation for a multi-criteria-decision-making (MCDM) approach is made that is based on a critical evaluation of commonly used methods.

Finally, frequently used indicators that could be used for an evaluation of water management strategies are presented in section IV.

Section II was prepared by the Office International de l'Eau with contribution from CNRS (Bernard BARRAQUE) the Hebrew University of Jerusalem (Eli FEINERMANN), KIVUN (Gadi ROSENTHAL) and NTUA while the remainder of this deliverable was written by RUB.

2 Section I: Water Management Analysis

2.1 Introduction

Effective management of water resources at catchment level warrants some anticipation of how water resources are going to change in the future under the influence of both natural and man made changes.

Methods for the quantitative analysis include (1) methods for the analysis of water supply including water availability and water management options, (2) methods for analysing the water demand in different sectors, (3) models and tools for forecasting both water demand and availability and (4) methods for both optimising and simulating water resources systems at river basin level.

Consequently, this section is organised in the following way: The first part discusses appropriate models for analysis water availability. The second part concentrates on models to estimate the water demand per sector in a long-term perspective. Models for single components of a water management system are described before some methods for analysing water resources systems at river basin level are presented. Finally, some recommendations are made for selecting models and tools in the framework of the WaterStrategyMan project.

It is important to note that this report is not aimed at describing existing models and available software packages in detail but to give a general overview of methods for the qualitative analysis of water resources systems. Computerised models that are available as software packages and DSS for water resources management will be described in a different report.

Models represent the problems in a (simplified) way that enables information to be processed quickly and efficiently.

A model is generally composed of three components (Major and Lenton, 1979):

- Parameters; numerical values that describe fixed of well-known properties of the system
- Variables defining the behaviour and the performance of the system being modelled
- Constraints describing the relations that describe the system's operation on the parameters and variables

River basin models are indispensable tools for aiding the decision making process in river basin management. They are used to assess the river basin management with regard to environmental, economic and social effects of alternative water management policies and to explain and understand the underlying processes in the system.

However, one should acknowledge the inherent limitation of models due to uncertain input data, simplifications and the fact that quantitative analysis alone represents only a small part of the overall planning and decision making process.

2.2 Water Management Analysis

The first step in water management planning should be a comprehensive analysis of the existing circumstances of the water management balance in the region of interest (typically a river basin). A simple hydrological water balance consists of comparing input (long-term precipitation), output (long-term) runoff, evapotranspiration and the change of storage in the catchment.

The level of detail of such a water balance can vary greatly and ranges from simple surface water balances, groundwater balances, water quality, combined balances of groundwater and surface water to complex investigations of both groundwater and surface water and water quality and quantity.

In a water management balance, water demand and supply in the region are compared to assess the challenges and options of water management which is aimed at compensating differences of between demand and supply today and in future.

A water balance analysis can be done at different spatial scales and may range from horizontal balance of a river reach to (sub-) basins. The scale depends on the objectives of the balance and the available data.

A water balance can be based on long-term yearly average values or monthly values of both stochastically simulated or observed mean values.

In complex river systems with numerous water users and water management objects the water management analysis encompasses detailed balances using simulated time series of water demand. In cases of water scarcity, the analysis focuses on considering priorities to different water users depending on their location, operating rules of reservoirs and many other factors.

Detailed water management models are indispensable tools for assessing and planning water management in particular if a long-term perspective is considered. They allow for an integral assessment of the existing conditions of water management in a river basin as well as for an assessment of water management interventions in the basin.

It is of utmost importance to first analyse the existing water resources system and its problems with regard to water quantity (water shortages) and water quality as well as a number of predefined objectives. The importance to have reliable and comprehensive data on the river basin scale cannot be overemphasised.

If the performance of the water resources system is assessed for the current conditions, the indicators discussed in the report for WP 4.3 may be used. These indicators describe either the performance of a single object of the system using criteria such as risk, reliability and resilience or for the system as a whole if aggregated indices are used.

2.3 Models at Subsystem Level

The purpose of this chapter is to present models for describing single components of river basin management. The first part describes models for water supply that are composed of components for water availability as well as components for the management of water such as reservoirs, waste water treatment plant etc.

In the second part, models for describing and analysing entities where a water demand exists are described.

The models described here can be used for both analysing the existing situation with regard to water management as well as for forecasting the response of such elements under changed conditions.

Given the restrictions imposed by the available data as well as the purpose of the DSS to be developed in the WSM project emphasis is placed on those models that are (1) easy to use, (2) do not require very detailed data and (3) are widely accepted in river basin management modelling. Rainfall-runoff models are not described in this report.

2.3.1 Water supply components

Water availability

Knowledge about the hydrological regime of a region or a catchment is a crucial prerequisite for any hydrological work. The available water has be assessed with regard to quantity and quality of groundwater resources, surface water and marine or coastal waters.

Groundwater resources

Groundwater resources are of high importance, especially in arid and semi-arid regions where surface water is limited. They include deep and shallow aquifers that are connected to rivers, streams or seas and non-rechargeable (fossil) resources that have been created by precipitation during the last Ice Age. Increasing needs for groundwater systems have basically two implications; the "mining" of groundwater (in which the abstraction exceeds the rate of replenishment) and the degradation of water quality due to point and non-point pollutants. In coastal areas, overexploitation of aquifers can reverse the natural flow into the sea, so that seawater intrusion occurs.

For a quantitative analysis it is important to have sound estimates of the recharge of the aquifer in a given time as well as its interactions with surface waters (recharge and discharge).

For an assessment of groundwater resources it is essential to have repeated observations of groundwater levels at a relatively large number of observation wells since groundwater systems respond to short-term and long-term changes in climate, groundwater withdrawal and (artificial) recharge and land uses. Estimates on groundwater storage requires the knowledge of aquifer storage properties and accurate interpolation of groundwater level measurements.

The concept of a *sustainable yield* is commonly used to limit the extraction from aquifers. Sustainable yield is defined as the long-term average annual recharge which can be extracted each year without causing unacceptable impacts on the environment or other groundwater users. The sustainable yield of a given aquifer is usually given as a fraction of the long-term annual recharge but it is clear that it can only be applied individually.

Water quality parameters in groundwater that are usually considered include phosphate, nitrates, ammonia, coliform bacteria, heavy metals, salinity, temperature, iron and manganese.

Surface water resources

Surface waters encompass both rivers and lakes and can quantitatively be assessed by long term averages of the available water resulting from endogenous precipitation. The temporal variations have to taken into account.

The following water quality parameters for surface water are usually considered:

Total organic carbon (TOC), total dissolved solids (TSS), biochemical oxygen demand (BOD), chemical oxygen demand (COD), pH, dissolved oxygen (DO), faecal coliforms, ammonia-nitrogen, nitrate and nitrite.

Since biological as well as chemical processes strongly depend on the temperature, the temperature should additionally be considered.

Marine and coastal waters

Marine and coastal waters used for desalination form practically an unlimited resources of water. For the assessment of its quality additional parameters to the parameters for surface water quality should be considered.

Forecasting water resources

This section briefly reviews available methods for estimating water availability or hydrology in a long-term perspective.

Long-term forecasting models can be classified into three major groups: (1) index methods, (2) storage accounting techniques and (3) conceptual simulation (Maidment, 1993).

Index models relate one or more variables affecting runoff such as precipitation prior to the forecast period or the soil moisture conditions at the time of forecast. Storage accounting models estimate the water stored in the entire catchment and compute the runoff as a function of the storage. Conceptual simulation approaches use a simulation of observed meteorological data for the time prior to the forecast period and estimates of the relevant data for the time of forecast.

Time series models for hydrological processes by estimating parameters that determine the dependency of a given value on his predecessor. Commonly applied models are autoregressive (AR) or moving average (MA) or combined (ARMA) models.

Time series forecast approach the mean value of the time series as the lead time of the forecast increases.

A number of attempts have been made to forecast the water availability on various scales. While earlier models made forecasts for the global and national scale and thereby lacked information of the distribution of water demand and supply on a basin-wide level, recent studies concentrated on forecasts on river basin level.

On the global scale, an attempt was made to model water resources for over 4000 river basins for a long-term perspective. The Centre for Environmental Research (University of Kassel, Germany) developed the WaterGAP (Water-Global Assessment and Prognosis) tool that takes into account physical and climate factors that lead to river runoff and groundwater recharge.

The water availability module computes total runoff, subsurface runoff and slow groundwater runoff (base flow) for any grid cell of 0.5° x 0.5°. The calculation is based on potential evapotranspiration, water content in the root zone and total available soil water capacity, effective rainfall and a calibrated runoff factor. In addition, water in every grid cell is routed to the neighbouring cells taking into account slope characteristics, soil texture and hydrogeological conditions.

An analysis done with data from the global runoff data centre (GRDC) and the University of New Hamphsire indicates that the number of river basins with just adequate water supply (more than 1.700m^3/person*year) will decrease by 6 and another 29 basins will face water shortages by 2025.

The so-called *Water year method* is a method that is implemented in the WEAP package. Based on the long-term average resources for a given supply, four types of non-normal water years (very dry, dry, wet, very wet) are defined that specify the availability of water in relation to the average conditions.

The factors are usually derived from hydrological time series whereby the time series is grouped into quintiles and the variation from the norm is computed for any group (e.g. wet year is equivalent to 1.25 time normal year etc.). The water years can be defined on a yearly basis as well as for monthly variations.

In addition to the method described above historical streamflows can be used for the entire simulation period.

Climate change can have a significant impact on water availability. The intergovernmental panel on climate change (IPCC) warns that "projected climate change could further decrease streamflow and groundwater recharge in many water-stressed countries" (IPCC, 2001). At the river basin scale, however, the direction of the changes is uncertain. Figure 1 shows the impact of climate change scenarios and the related meteorological parameters on runoff on the upper Danube catchment (Schumann and Antl ,2001). Seven General Circulation Models (GCM) have been used as input for a water balance model. The resulting changes in summer and winter runoff indicate that nearly half of the scenarios showed a decrease in summer runoff or more than 20 percent while only 7 percent of the scenarios show a decrease in winter runoff of the same magnitude.

However, the study also comes to the conclusion that future changes in water resources to human-induced changes are highly uncertain and that linking global climate change to regional water resources availability remains a very difficult task.

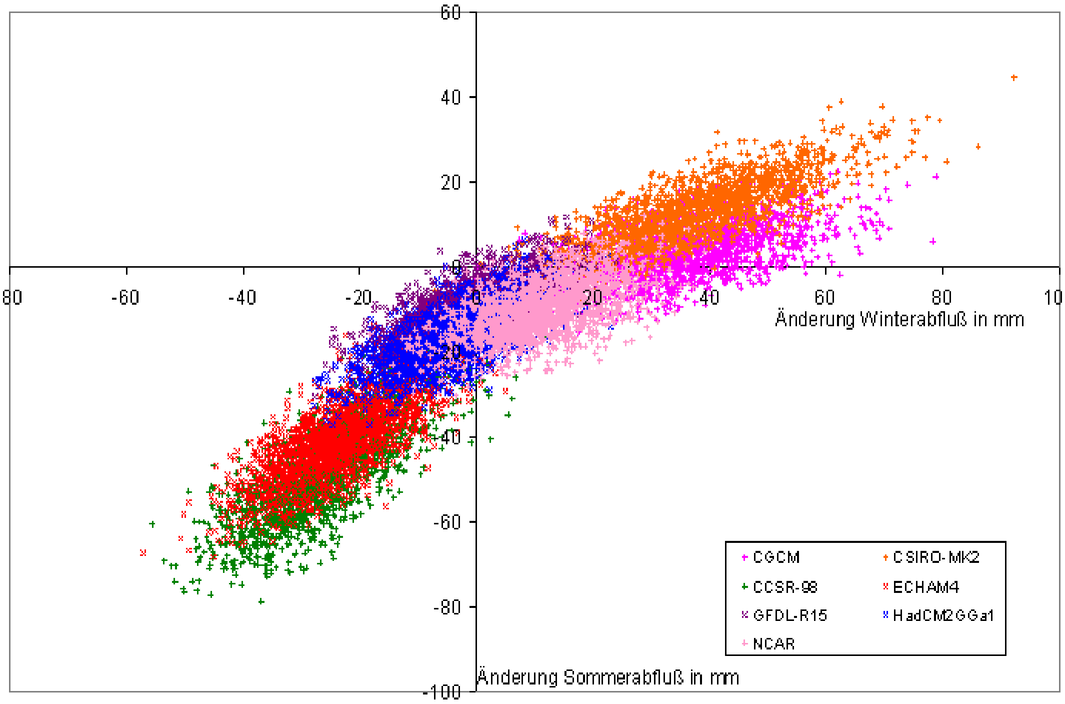


Figure 1: Impacts of climate change on the upper Danube river, deviations of summer and winter runoff in mm (Schumann and Antl, 2002)

Models for water management

Water management structures considered as components with water management models:

- GW management facilities
- Reservoirs including hydropower facilities
- Water treatment plants
- Waste water treatment plants
- Pipelines for inter-basin water transfer

Each management component consists of two different parts: a set of parameters that describe the physical structure of the component such as capacity, capital cost, O+M cost etc. Secondly, the operation can be characterised by operating rules that describe the operation of the component depending on a given state of the component.

In some models, the operation is considered in detail while other approaches use a generalised description of the behaviour of these structures. .

GW management

The physical structures related to groundwater management comprise single wells and well fields for discharge as well as infiltration basins and recharge wells for recharge of groundwater.

The simulation of groundwater flow is based on the general equation for transient flow through a saturated porous media, which is given by

$$\frac{\partial}{\partial x}\left(T_x \frac{\partial h}{\partial x}\right)+\frac{\partial}{\partial y}\left(T_y \frac{\partial h}{\partial y}\right)+\frac{\partial}{\partial z}\left(T_z \frac{\partial h}{\partial x}\right)=S\frac{\partial h}{\partial t}+W\left(x,y,z,t\right)$$

where h is the hydraulic head, T are the transmissivity tensors along the x, y and z axis, S is the storage coefficient and W(x,y,z,t) represents a source/sink term.

A water quality model for groundwater that considers advection and dispersion requires that the velocity field in the modelling domain is known. The advection-dispersion equation is used to simulate the transport of solutes influenced by advection, dispersion and chemical reactions.

Mathematical models to approximate these equations typically use a finite difference (FD) or finite element (FE) numerical scheme.

Both methods solve for the dependent variable at each node in a grid that is superimposed over the modelling domain. It is clear that such models require a tremendous amount of data, computation time and expertise.

In computer models for integrated water resources management at river basin scale, however, a very simple conceptualisation of the aquifer is used. These tools include MIKE Basin 2000, WEAP and many other river basin management tools. In all these models, the aquifer is represented by a single linear reservoir. A linear reservoir is a fictitious reservoir where the outflow Q_o is linear dependent on the storage volume S:

$$S = k \cdot Q_o$$

where S is the storage volume, Q_o is the outflow from the reservoir and k is a storage or retention parameter with the dimension of time. The mathematical solution for the simplest linear reservoir is an exponential decay of storage with time. The reservoir may be emptied if the outflow permanently exceeds inflow and may overflow if the inflow permanently exceed outflow.

If a cascade or a series connection of n equal reservoirs each having the same storage coefficient k is used to represent the groundwater storage the approach is well known in hydrology as Nash-cascade.

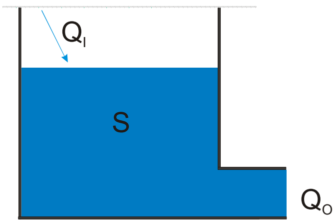


Figure 2: Conceptualisation of an aquifer as a Linear Reservoir Model

With regard to the sustainable use of the aquifers systems it is of crucial importance to define the optimal rate at which groundwater should be extracted. In any case the rate of extraction should not exceed the rate of recharge but defining a rule that prevent excess withdrawal of water as a function of recharge remains a difficult task and can only be done individually.

Groundwater quality plays an important role as it directly affects the water quantity in the sense that polluted groundwater may not be suitable for a given demand. Based on the concept of the linear reservoir for water quantity, simple models for GW quality exist but such models can only give a vague picture of the quality of groundwater.

Conjunctive management of GW and surface water

The conjunctive use of groundwater and surface water can significantly increase the efficiency and the cost-effectiveness and reliability of the aquifer-river system. Stephenson (1991) defines conjunctive use of surface and groundwater "as the management of surface and groundwater resources in a co-ordinated operation to the end that the total yield of such a system over a period of years exceeds the sum of the yields of the separate components of the systems resulting from an uncoordinated operation".

The advantages of utilising groundwater compared to surface water can be summarised as follows:

- creating less of an environmental impact
- smaller losses due to evaporation and seepage
- fewer topographical limitations
- increased reliability
- no sedimentation problems compared to surface reservoirs.

Since the groundwater is in general more expensive and should only be used in times of an emergency, the lower cost argument is debatable and depends on the individual case.

Stephenson (1991) presents a model for the optimum operation of groundwater and surface water sources that optimises the operation by linear programming similar to operating an isolated reservoir. Alternatively, the model can be solved using Stochastic Dynamic Programming (SDP). The conceptual model is depicted in Figure 3.

Other models (Daene et al. 1999) include other objectives such as water quality control and prevention of undesirable overdraft of groundwater in addition to water allocation.

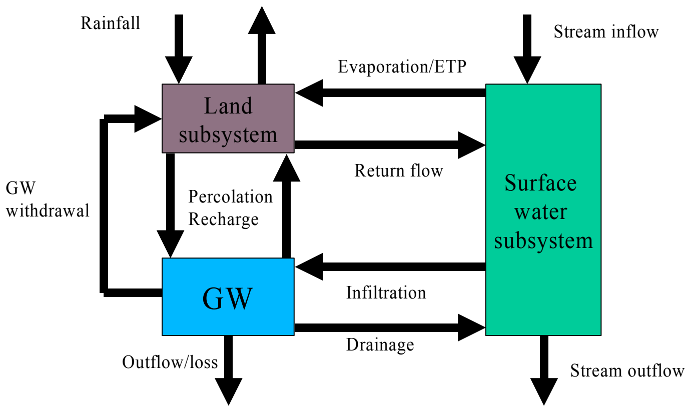


Figure 3: Conceptual model for conjunctive use (Stephenson, 1998)

Reservoirs

A reservoir is characterised by its physical structure described by parameters such as its capacity, height etc. The operation of reservoirs can be simulated using the repetitive application of the hydrological equation

$$S_{n+1} = S_n + I_n - U_n - E_n - F_n$$

where S_n is the storage at the beginning of month n, I_n is the inflow for month n, U_n is the release for month n, E_n is the amount of evaporation (as a function of S_n) and F_n denotes a flood overflow (omitted if not positive). In any case, evaporation from the water surface needs to be taken in to account. Therefore, the stage-water surface relationship of the reservoir has to be specified. In addition, the stage-volume relationship has to be parameterised to calculate the volume as a function of the water table in the reservoir.

Depending on the geological conditions seepage losses have to be taken into account.

A monthly time step is the appropriate resolution for preliminary studies. The draft U is specified as a constant value or as a mathematical function of the storage state. The capacity of the reservoir must be specified in the programme so that spill will occur if the storage capacity reaches the capacity of the reservoir. Furthermore, the simulation procedure requires specifying an initial storage state for the reservoir. The optimal system design is usually obtained by analysing a number of combinations of storage and release so that the optimal operating rule can be found. Upper and lower bounds for the storage can be accommodated.

Operating rules

Reservoir operation models are aimed at optimising operating policies of reservoirs or systems of reservoirs by considering given objectives. Objectives can be low-flow augmentation, flood protection, optimised energy production, recreation, water quality management etc.

Variable draft from reservoirs can increase reliability and total yield and thereby save costs. There are many objectives on which such operating rules should be based, including hydrological, environmental, political and trade-off. Defining effective predefined operating rules for reservoir is a challenging task, in particular if multiple objectives and/or multiple reservoirs are considered. Such rules take into account the losses due to evaporation, the probability of spillage, and the different water users that use both inflow and reservoir storage volumes.

Typical system rules determine the water to be released from the reservoir as a function of the existing storage volume.

Stephenson (1991) describes the following alternative operating procedures for optimising the yield of reservoirs:

- Maximum total yield
- Minimum economic loss
- Continuous hedging
- proportional risk
- sharing
- capacity allocation
- variable draft

Exemplarily, the application of the Hedging rule for drought management in shown in Figure 4 (U_t and Y_t denote reduced and additional draft as a function of storage)

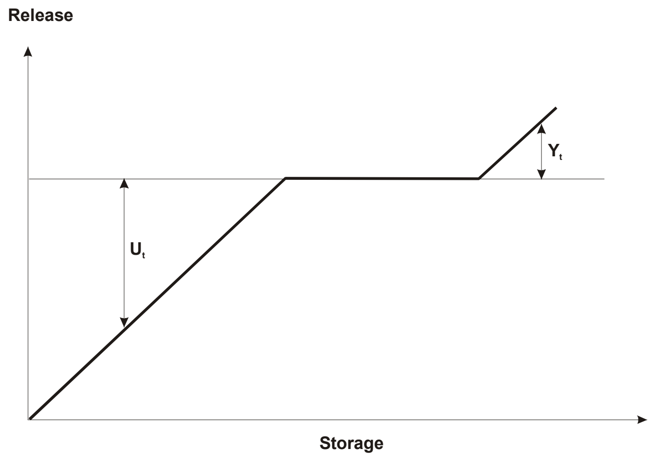


Figure 4: Schematic representation of the Hedging rule for reservoir management

Hydropower

The water demand for hydropower facilities connected to a reservoir structure can be calculated by calculating the amount of water that is needed to produce a given amount of energy.

The amount of energy that is converted by a hydraulic turbine using the energy of water is computed by integrating the power produced by the turbine over time. Power output is computed as

$$P = H \cdot g \cdot Q \cdot e$$

where H is the net available head, Q is the flow and e is the overall efficiency which includes turbine and generator efficiency. The integration over time yields the amount of energy:

$$E = \int_{t=0}^{T} Q \cdot H \cdot g \cdot e dt$$

The integration is usually based on flow duration curves. Long-term power production is subject to the uncertainties in reservoir inflow.

Water treatment plants

Water treatment plants are inevitably connected to a water resource (groundwater, surface water or coastal/brackish waters). Although the processes involved can generally be modelled it will suffice the purpose of the WSM project to consider a balance of effluent and influent of the plant for the quality parameters and assume the quantity to be constant.

A desalination plant can also regarded as a water treatment plant. Water quality parameters for both effluent and influent have to be considered. Typically, these parameters are manganese, iron, hardness and others (see above).

Waste water treatment plants

There exist a number of mathematical models describing the microbiological processes in biological wastewater treatment plant. The activated sludge model ASM (International Association of Water Quality) for instance is frequently used.

Given the purpose of the DSS to be developed in the WSM project it suffices to model a wastewater treatment plant by an entity that reduces the quantity of water and changes the parameters of water quality depending on the type of process involved (secondary treatment, tertiary treatment). Parameters that are usually considered are biochemical oxygen demand (BOD5), total suspended solids (TSS), volatile suspended solids (VSS) , Total Kjeldhal nitrogen (TKN), ammonia nitrogen (NH3-N) and phosphate (P).

Following the recommendations of the WFD, the threshold values for the above constituents given in the Urban Waste Water Directive (91/271) should apply.

Pipelines (inter-basin transfer)

Water transfers involve small-scale transfers in which water is conveyed from one small sub basin to another as well as water transfers from wet areas to those areas having water scarcity problems.

Water transfers from and to neighbouring basins are characterised by flow rate that can vary with time and may be described by operational rules.

For a representation of the transferred water with respect to quality, the same parameters as described above apply.

2.3.2 Water demand components

The following subsection describes the entities that demand water. Four demand types are distinguished: industrial demand, agricultural demand, domestic demand, demand for hydropower and environmental demand.

In addition to these nodes, a number of other water demand nodes can be defined. These include a minimum water demand for navigation, a demand for recreation and others.

Industrial water demand

The amount of water used in industry can be classified as follows:

- processing water which is the water that comes into direct contact with the product,
- cooling water which is used for cooling of various items,
- boiler water used for steam generation
- Water for general purposes (e.g. cleaning and air conditioning).

The industrial water demand may be correlated with the amount of material produced, the value of the product, the number of output units produced etc.

A water demand estimation for industry therefore depends on the variables mentioned above and may be difficult to conduct as long as such data does not exist.

Agricultural water demand

The term agricultural water demand here refers to four different demand sites: irrigation water, water used for livestock, water used in forestry and aquaculture. The two latter ones are in general negligible small and will not be discussed here.

A distinction has be made between water that is used in the sense that it is no longer available to other users (e.g. water taken up by the plants) and water that is used non-consumptively (e.g. return flow from irrigation plots).

To estimate the irrigation water demand, it is in many cases sufficient to compute the total demand as the product of the water duty (i.e. demand per area) and total irrigated area. The most commonly used approach for estimating the crop water demand is the FAO crop coefficient method that is based on a reference evapotranspiration and a crop coefficient Kc that accounts for crop characteristics, crop development, vegetation periods and others. Reference evapotranspiration ET_0 is defined as the evapotranspiration from and extensive surface of green grass cover of a height of 12 cm adequately watered. The crop water requirements for a given crop are given by

$$CWR_i = \sum_{t=0}^{T} \left(Kc_i \cdot ET_0 - P_{eff,i} \right)$$

where P_{eff} is the effective precipitation at time step i.

The net irrigation requirement NIWR for a given scheme or region is the sum of individual crop water requirements divided by the total irrigated area:

$$NIWR = \frac{\sum_{i=1}^{n} CWR_i \cdot S_i}{S}$$

The gross irrigation water requirement (GIWR), being defined as the amount of water that has to be extracted and applied to the irrigation scheme includes losses and is defined as

$$GIWR = \frac{1}{E} NIWR$$

where E is the global efficiency of the irrigation system (i.e. the ratio of the abstracted water that actually reaches the plant). In order to represent different water management options in irrigation planning, it is necessary to disaggregate the global efficiency into three different efficiencies (EEA, 2001), namely:

- Conveyance efficiency, being defined as the efficiency from the abstraction to the network,
- Distribution efficiency, referring to the losses in the distribution network
- Application efficiency, which represents the amount of water that really reaches the plant when, applied in an irrigation plot.

On a global level, irrigation efficiency is estimated to be only around 40 percent (Revenga et. al. 2000)

If the crop yield is calculated, one has to consider the following factors: (1) the physical characteristics of the area (topography, soil etc), (2) the type of crop, (3) the quantity and timing of water and fertiliser application, (4) the available labour and machinery and (5) the land management practices.

Models that take into account the effects of salt accumulation in the soil and the dynamics soil moisture and transport process are in general based on the Richard's equation and can be classified into short-term models and long-term models.

Short-term models are confined to one year or a single irrigation season while long-term models can be regarded as a succession of short-term models. A number of both such types of models have been developed in the last decades, but detailed models that consider the dynamics of the root zone do not seem to be appropriate for the Decision Support System (DSS) to be developed by the WSM project due to the required resolution of detailed data.

Evaporation models are models that simulate the crop yield depending on salinity levels, soil moisture conditions and irrigation strategies by assuming a linear yield-evapotranspiration relationship. These models are usually site specific and very data-intensive. The simplest type of relationship between actual yield and actual evapotranspiration is given by

$$\frac{Y}{Y_{\max}} = 1 - k_c \left(1 - \frac{E}{E_{\max}}\right)$$

where Y and Y_{max} are the actual and maximum dry matter yield in tons per ha, k_c is the crop coefficient and E and E_{max} represent the actual and maximum evapotranspiration in mm.

Estimated production functions compute the yield by relating a number of variables such as salinity, soil moisture and others to the expected yield. Again, such models are data-intensive and may be appropriate for irrigation management at a given site, but are too specific for the purpose of the WSM project.

Domestic water demand

The water demand of human settlements (urban demand) includes demand for domestic uses such as drinking, cooking, kitchen and toilet use, gardening, car washing etc as well as commercial uses such as water demand for offices, stores, laundries, fire fighting, public works and so on.

It is often difficult to clearly distinguish the demand of industrial units in settlements. Water demand for industrial units connected to the urban water distribution network is therefore often considered as part of the urban water demand

The importance of leakage losses that are part of the urban demand cannot be overemphasised here: recent estimates by the European Environmental Agency EEA indicate that the losses of water that due to leakage may amount up to three quarter of the water supplied (Table 1).

Table 1: Estimated losses (% of water supplied) in selected countries (Source: European Environmental Agency, 2001)

Albania	up to 75
Croatia	30-60
Italy	15
Germany	9
Slovenia	40
Spain	24-34

Hydropower demand

If water is used for the production of energy it is not used consumptively, but the flow regime of the river may be changed significantly which, in turn, may affect other users. For forecasting the hydropower water demand depending on the amount of energy to be produced see the previous section.

Environmental demand

In order to conserve the hydrological and ecological function of the drainage network, the physical regime of the river must not be altered or dried up. The amount of water that is needed to sustain an ecological value of an aquatic ecosystem is referred to as environmental water demand. The question of a minimum flow is particularly important in arid and semi-arid regions and must be borne in mind in river basin planning and management. An ecological minimum flow can artificially be maintained by reservoir management.

To model environmental demand, a given river stretch can be assigned a minimum flow requirement that has to be met.

Forecasting Demand

Forecasting water demand is the essential input for decision-making in water resources planning and management.

The demand forecast is heavily influenced by a number of uncertainties. These include general economic uncertainties, climate change implications, trends in population development and technology. During the last 30 years, considerable effort has been made on the improvement of water demand forecasting methodologies, mainly by disaggregation of demand into different components and integration of demand-management effects (Foukh, 2001).

Regression techniques

Regression models for demand forecasting are based on the assumption that variations in water demand W are correlated to a number of variables X_i that influence such demand:

$$W = a_1x_1 + a_2x_2 + \ldots + a_nx_n + e$$

where a_i are coefficients and e is an error term. The coefficients are determined by solving the above equation for water demand observations in the past. In doing so, it is implicitly assumed that the influence of the explanatory variables will keep the same pattern in the future, i.e. they are stationary with time. Various explanatory variables may be selected such as population growth, economic growth, output product for which the water is used etc.

In order to estimate the water demand for a given point in future, it is necessary to project the variations of the variables independently and to compute the water demand using the coefficients determined for past conditions. The limitation of this method lies in the fact that the assumption of stationary coefficients is not true in general and the problem that not all explanatory variables will be included in the regression analysis.

Forecasts based on activity levels techniques

Traditional approaches for forecasting water demand consist of estimating population (or industry output units etc.) and multiplying with an average per capita demand to obtain the mean annual demand. Average per capita demand can be further broken down into demand for different activities such as bathing, showering, toilet flushing etc. but this approach can only make sense if (1) the data is available and (2) the responses on the demand for different water management interventions are known.

The peak demand for a given period (daily, monthly etc.) is obtained by applying peak factors to the annual demand. Such techniques can be regarded as a special type of the above described multiple regression approaches.

Although those methods are very frequently applied due to their simplicity, there a number of limitations; such approaches do not contain any allowance for price elasticity of demand and other factors.

The demand module of WEAP package (Tellus Institute) uses the activity level approach where both activity levels and unit water requirements can vary with time. In addition, there are three methods for projecting activity levels and water use rates to future conditions:

- Interpolation
- Drivers and elasticities
- Growth rates

Drivers are explanatory variables that determine the demand (e.g. population, consumption, industrial output, investment etc.).

Elasticities are econometric relationships such as the water demand as a function of the water price.

WaterGAP, a tool for global assessment of water availability and water demand on long-term perspective estimates water demand for the three sectors domestic, agricultural and industrial. The agricultural sector is further subdivided into irrigation and livestock components. Forecasts are based on the concept of "structural change" (per unit water use changes with the development of economies) and "technological change" (efficiency improvements lead to decrease in water use).

Econometric models

These types of models assume that the water demand is an aggregation of a large number of water use categories that, in turn, depend on a large number of factors. A popular example of such models is the IWR MAIN software package that can be used to forecast both, residential and non-residential demand. The model requires a large number of data and assumptions for the different sectors. The general structure of the model is given in Figure 5.

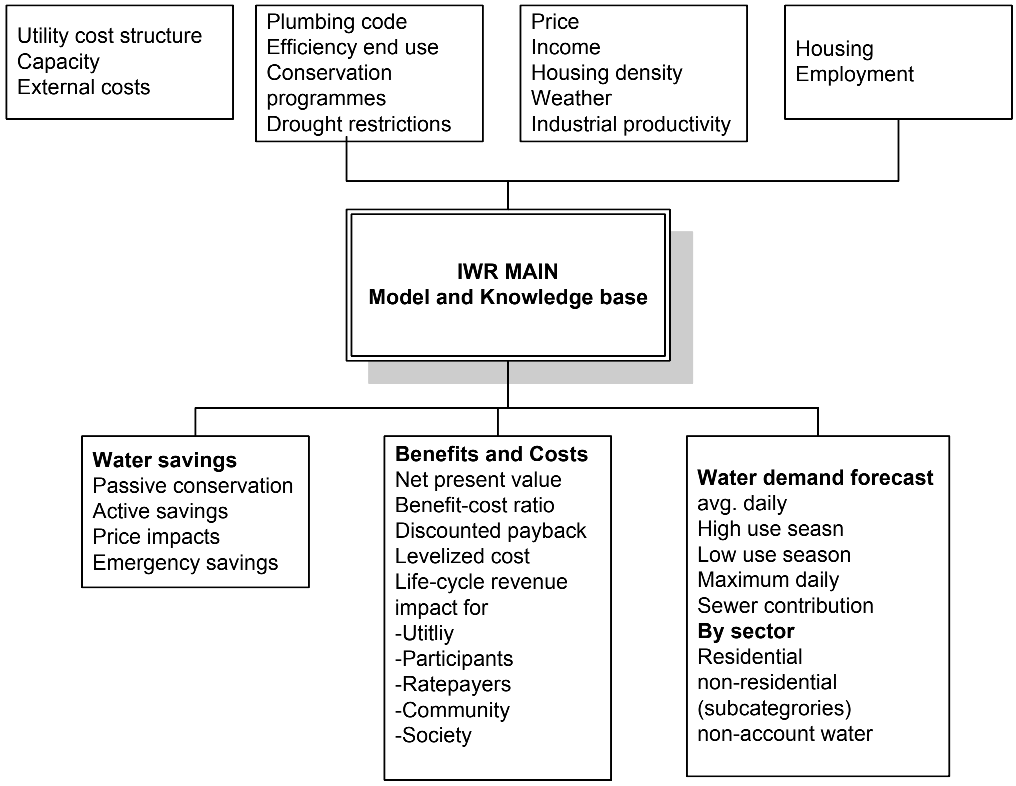


Figure 5: Structure of the IWR MAIN model

Forecasting demographic development

The crucial point in all forecasting models for water demand is to have the best possible knowledge about the future development of population since most of the operational goals of water management are directly or indirectly linked to population.

Demographic models for a given region basically consist of two components; (1) the natural development of the population due to a surplus of births over deaths (or vice versa) and (2) the component that considers migration (immigrants and emigrants).

It is always easier to come to reliable migration figures at the national level than on a sub-national or regional level. The reason lies in the fact that regional migration disappears at the national level and that population parameters are more reliable on national than on regional level. Consequently, reliable demographic data is in most countries only available at the national level.

Population forecasts can be performed generally in two different ways: (1) time series of the past can be extrapolated using mathematical tools and (2) the knowledge of structural and behavioural patterns and changes gained from observations in other cases can be used to model the future population development.

Mathematical tools do not necessarily need a theoretical explanation of the variable that determine population growth. The following approaches are frequently used:

- linear extrapolation
- geometric extrapolation
- linear trends
- non-linear trends
- regression analysis.

As the models are based on past observations it is necessary to analyse the time series very carefully in order to exclude factors that influence the regular type of behaviour (e.g. migration wave).

Scenario approaches

Scenario approaches (sometimes refereed to as "non-formalised models") can be used for both, forecasting and evaluating future development in a more or less comprehensive sense. In systems analysis, a scenario addresses the three following questions: "(1) What can happen? In other words, in the intended operating environment, what events is the system supposed to react to? (2) For each event that can happen, how should the system respond and (3) How can the system be designed so that it handles all the scenarios? That is, how can a single, integrated system be designed to cover all the relevant event sequences" (Sage and Rouse, 1998)

Scenarios can describe one or more conceivable future conditions or paths of development for a given set of variables and relations. They are always the method of choice of formalised approaches are not available or are judged unreliable or unacceptable.

As well as formalised models they compromise stock variables and flow variables, assumptions concerning the behavioural patterns (relations) as well as development constraints.

Based on three different scenarios the water use is computed per sector and per region.

As well as for demand, the scenario approach can be used for water availability forecasts, in particular if a long time perspective is considered.

Table 2 summarises the required data for a water management analysis that considers both water quantity and water quality issues.

Seasonality of demand

Since tourism and irrigation water demands are the dominant water users in all the regions, the overall water demand has a very strong seasonal pattern.

It is suggested that a classical decomposition approach is used which is based on the assumption that the observed value equals a trend factor multiplied by a seasonal and a irregular (random) factor. Formally, the demand D for a given sector in year *y* and month *m* can be computed by ,

$$D(m,y) = TSR$$

where T is the trend component, S is the seasonal and R denotes the irregular (random) component. The trend component T is estimated by a centred twelve-month moving average. The seasonal part of the observed value can be isolated by

$$S(i,m) = \frac{1}{Y}\sum_{y=1}^{Y}\frac{D(i,m,y)}{D_{ma}(i,m,y)}$$

where S(i,m) is the seasonal factor for month m, D(i,m,y) is the observed water use in month m and year y and Y is the number of historical years of record. To ensure that the sum of the seasonality factors add up to one a normalisation step is necessary.

If a trend is to be estimated from year to year data, the seasonal variations need to be removed from the data (by dividing the recorded values of water use by the seasonal indices).

Table 2: Required data for water management analysis on a basin scale

Sector	Water quantity	Water quality
Water supply	Internal renewable water resources	Capacity of waste water treatment plants
	Runoff data at catchment outlet (time series; if n/a: long-term average)	People connected to utility
	Renewable groundwater	Type of treatment (secondary, tertiary,..)
	Groundwater recharge - natural - artificial	BOD and nutrient removal rates in WWTPs
	Non-renewable groundwater	Treated sewage
	Surface water	People connected to waste water treatment
	Desalination plants capacity	No of treatment plants failing the EU waste Water Directive standards
	Capacity of reservoirs	Drinking water quality
	Operating rules for existing reservoirs (objectives)	Quality of marine/coastal Waters
	A(h), V(h) relationships for reservoirs	WQ parameter for surface water bodies
	Transfer from neighbouring regions	BOD per capita
	Water recycling/reuse	N,P and organic matter in rivers
	Losses in distribution system	Nitrate in groundwater
	Unaccounted for water	Area of agricultural land
		Pesticide consumption
		No of livestock
Water demand	Water demand per person	WQ parameters for return flow
	Abstraction from surface water	WQ parameters for return flow
	Abstraction form groundwater	WQ parameters for return flow
	Abstraction from fossil groundwater	
	Final water consumption	
	Number of licences for abstraction	
	Industrial water requirement ▪ consumptive ▪ non-consumptive	WQ parameters for return flow
	Agricultural water demand ▪ consumptive ▪ non-consumptive	WQ parameters for return flow
	Water demand per overnight stay	
	Seasonal demand pattern	

2.4 River basin management models

2.4.1 Introduction

If the components of a water resources system described above and their interactions are modelled in an integrated approach it is important to define the spatial and temporal scales.

The hydrologic system provides a more comprehensive and rational setting for the assessment of water resources systems than any other spatial unit defined by political, administrational or local boundaries and is the appropriate scale for estimating a change in the system performance of water management interventions taking place. This concentration on hydrologically defined boundaries is known as the watershed approach and defined as follows:

"The watershed approach is a co-ordinating framework for environmental management that focuses public and private sector efforts to address the highest priority problems within hydrologically defined geographic areas, taking into consideration both ground and surface water flow" (US EPA, 1996)

A long-term sustainability assessment also requires a longer time perspective. All this warrants flexible tools which are able of reflecting the full complexity of water resources systems and representing all users, policies, resources and their interactions in a way that future developments can be integrated. A simplified schematic overview of water management analysis at catchment level is depicted in Figure 6.

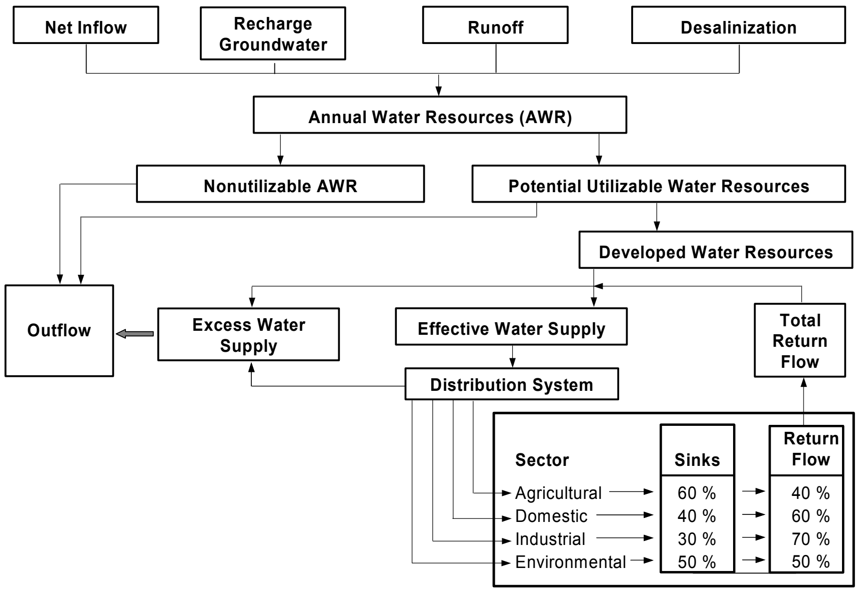


Figure 6: Schematised overview of water management analysis

As can be seen, the annual water resources comprise net inflow, groundwater recharge, runoff and desalination. Part of the annual water resources is potentially utilisable and part of it is non-utilisable. The potential utilisable water resources, in turn are partly used in the sense that the water is no longer available to other users ("sinks" or consumptive use) while part of it may be used again (non-consumptive use).

Models for optimal water management at river basin scale can be broadly classified into simulation and optimisation models; simulation models are models that simulate the behaviour of water resources systems based on a predefined set of rules which can be either actual or hypothetical.

Optimisation models are models that allocate water resources based on objective functions (e.g. economical, environmental or multi-objective functions).

However, models can include both, simulation and optimisation capabilities and both types are covered in this chapter.

Models for river basin management are described here in a more general sense; the implementation of such models in existing Decision Support Systems and software packages will be discussed in detail in work package 5. A general framework for river basin management modelling is given in Figure 7.

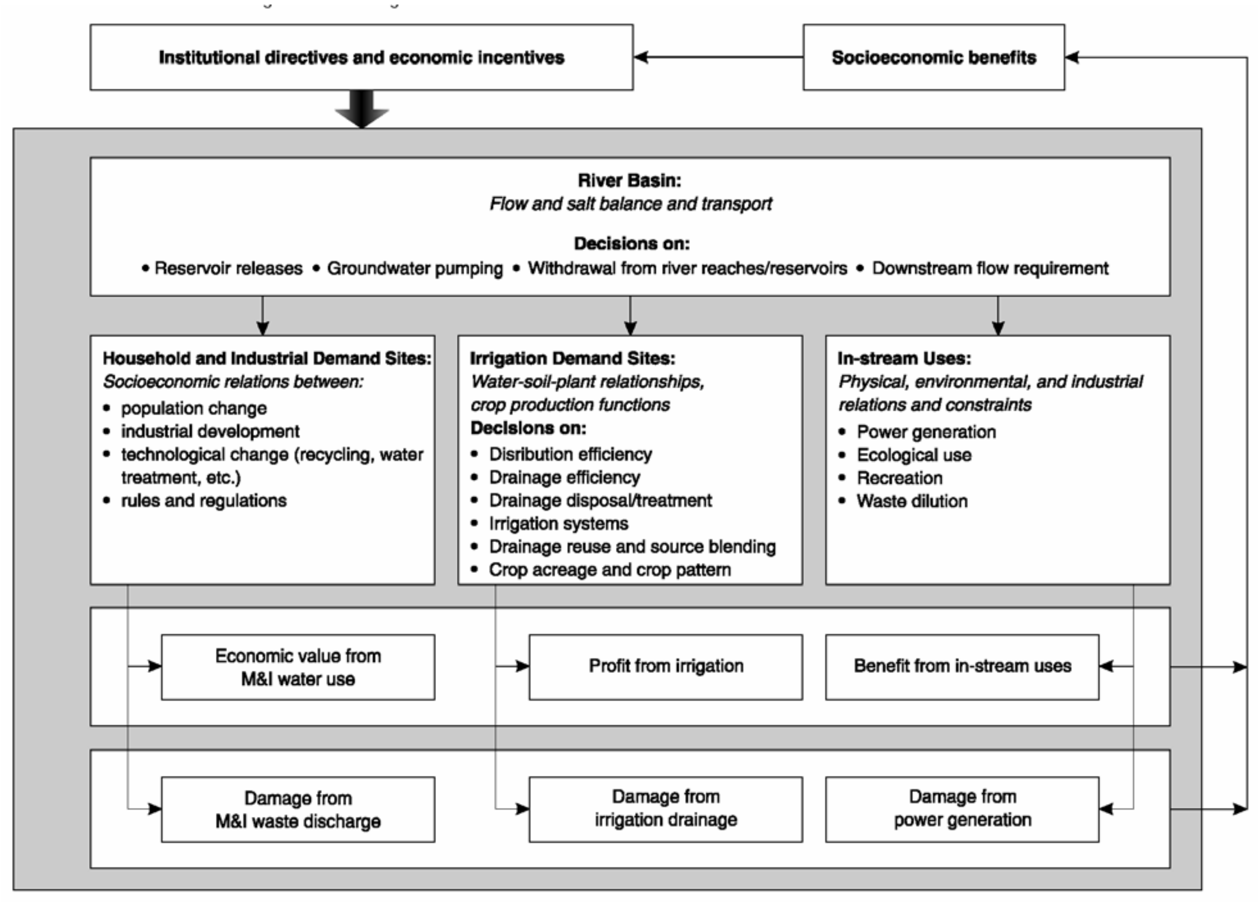


Figure 7: Framework for river basins management modelling (IWMI, 1999)

The first step in designing a water management model at river basin scale is to select the relevant processes and variables of the network using the water management objectives and measures as a starting point.

2.4.2 Simulation

Simulation models are used to assess the performance of water resources systems over a long period of time. The technique is therefore the obvious choice for studying the systems' response to extreme conditions and thereby to identify the components that are prone to failure. River basin simulation models play an important role in identifying the impacts to given scenarios of global climate change as well as population growth scenarios, changing demand patterns etc.

River flow simulation models

In river flow simulation model, the elements of the river basin are usually represented by a number of nodes and branches or junctions represent the interrelations between those elements. The water allocation is typically solved using network-programming techniques whereby priorities can be assigned to both demand and supply nodes. A wide range of these models have been applied in river basin management for (1) optimising the allocation of water and (2) simulating the basin's response under changed conditions. An example of a network representation of a river basin is given in Figure 8. Software packages that are based on the described approach include MIKE BASIN, WEAP, WATERWARE and many others.

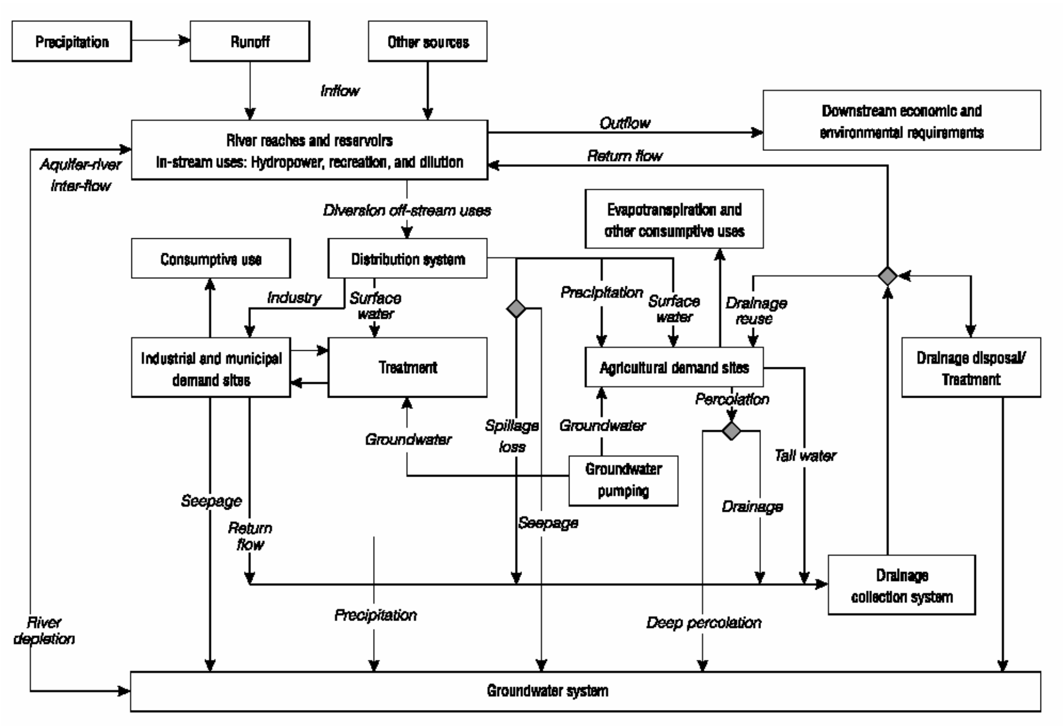


Figure 8: Schematic representation of river basin processes (IWMI, 1999)

River basin quality simulation models

Although this report is primarily concerned with methods and tools for the quantitative analysis of water resources systems, it is necessary to include a section on such tools for the description of water systems qualitatively, since environmental issues are a crucial part in IWRM in particular with regard to the implementation of the WFD.

The need to take multiple objectives including environmental issues into consideration in IWRM has stimulated the development of mathematical water quality models for predicting the impact of alternative pollution control measures.

Water quality models basically consist of a set of equations that describe the physical, chemical and biological processes that take place in a water body.

They are usually distinguished according to the model complexity, type of receiving water body (lake, river etc) and the water quality parameters that the model can predict.

Water quality simulation models vary greatly in their complexity which is a mainly a function of the number and type of water quality indicators, the levels of temporal and spatial detail and the complexity of the water body itself. Small lakes that mix completely are less complex than large rivers and large lakes, estuaries and coastal zones.

Simple water quality models that describe the aerobic status of a water body by modelling biochemical oxygen demand (BOD), dissolved oxygen (DO) and temperature are well established and are applied frequently all over the world.

Prediction of basic nutrients such as phosphate, ammonia, and nitrate works reasonably well for simpler water bodies and lakes. The modelling of heavy metals and toxic organic substances is somewhat more difficult.

Models can only cover a limited number of constituents and care must be taken that the constituents to be modelled are themselves representative for a number of other substances. All models require hydraulic data as well as base concentrations of the water quality parameters under consideration.

Water quality models can be used to analyse the steady-state conditions in which the values for water quantity and quality do not change with time or to simulate the dynamic time-varying conditions of transient phenomena. In many river systems it is sufficient to use 2-dimensional models that assume either vertical or lateral mixing. One-dimensional models assume complete mixing in vertical and lateral directions.

The choice of a particular technique to approximate the governing equations for strongly depends on the type of water body, amount of data, spatial and temporal resolution required and many other factors.

A frequently used water quality model is the Enhanced Stream Water Quality Model QUAL2E that is available from the United States Environmental Protection Agency (EPA). QUAL2E simulates temperature, DO, BOD, chlorophyll A, nitrogen (organic, ammonia and nitrate), phosphorus (organic and inorganic) and coliforms. In addition, any constituent can be simulated provided the user defines its decay properties.

2.4.3 Optimisation Models

Models that optimise the allocation of water in a river basin subject to a given set of rules must have a simulation component that is capable of calculating the hydrologic flows and the respective mass balances.

A large number of methodologies for the optimal allocation of water resources have been developed over the last decades. The common approach is to represent the elements of the hydrological basin by nodes and connections between those nodes.

Nodes can either be supply nodes (representing boreholes, treatment plants, desalination plants etc.) or demand nodes representing demand sites such as urban, environmental or industrial demand. Each demand node is assigned a demand for a given period d_i. If the capacity of the links between demand nodes and supply nodes is denoted f_j for j links, the problem can formally be described by

$$\min imise\left(\sum_i d_i - \sum_j f_j\right)$$

so that the water shortage on all demand nodes is minimised, subject to supply, demand, flow conservation and capacity constraints induced by the physical infrastructure.

It is possible to assign priorities to both demand and supply nodes that indicate the preference of water use and allocation for a given site.

The above problem is a standard problem in Operations Research known as maximum flow problem for which a number of solutions exist (e.g. Nemhauser et al, 1989).

Several other algorithms have been developed that allocate water based on different objective functions.

One such objective function can be maximising the sum of all economic benefits of off-stream and in-stream water use. Mathematically, this objective can be expressed as

$$\max_{p \in P} imise\, NB\left(X_P\right)$$

where NB is the net benefit, p denote the water management plans and the vector X represents the decision variables.

Others include minimising cost of transport and others.

2.4.4 Combined Economic-Hydrologic Models

Early models of this type have been focused on profit maximisation of water use for a given user (irrigation, industrial etc.) rather than on the benefits of water use for all users at the same time.

Typically, economic models are optimisation models whereas hydrologic models are simulation type models which causes difficulties in information exchange between the two. In addition two that, the integration of the two models may be hampered by the different spatial and temporal scales; the area over which economic impacts may have an effect will differ from the catchment area. Temporal scales for economic models are usually longer while the time step is smaller (annual, seasonal) than in hydrologic models. Combined economic-hydrological models have been frequently applied to analyse the economics of irrigated agriculture.

IWRI (1999) distinguishes two approaches to develop integrated economic-hydrologic models; the compartment approach and the holistic approach.

The compartment modelling approach

In the compartment approach there is a loose connection between the different hydrologic and economic components and only the output data is transferred between the components. The analysis is more difficult due to the loose connection whereas the single components of the model can be very complex.

The holistic modelling approach

Models based on the holistic approach use components that are tightly connected to a consistent model. The information exchange between economic and hydrologic components is conducted endogenously and one single technique for optimising the allocation of water resources is used. The crucial point here is to define the relations between economics and the hydrological components on which the economic analysis is based on.

Required data

The following table summarises the data that is required for modelling water management balances at river basin scale (in addition to the data requirements listed in). The table does not include data on economic issues.

Table 3: Minimum data requirements for river basin modelling

Sector	Data	Comments
General	Total basin area	
	Agricultural area	
	Irrigated area	
	Arable land	
	Topography	descriptive
	Vegetation	descriptive
	Geology	descriptive
Climate	Average annual rainfall	
	Time series of rainfall if n/a: variability (high, low)	
	Long-term Seasonal pattern of rainfall	
	Monthly potential ETP Long-term variability	Potential Evapotranspiration
	Mean monthly temperature	
Population	No. of households	
	Population growth rate	
Institutional framework	Development priorities	descriptive
	Capacity building	
	Stakeholder integration	
	…	
Environment	Minimum flow requirements (monthly min. demand)	

2.5 Water Resources Planning Under Uncertainties

2.5.1 Introductory remarks

Uncertainty is always an element of the planning and evaluation process of water resources systems. Uncertainty arises because of numerous factors that affect the performance of the system but cannot be known with certainty at the time the system is planned or evaluated. Basically all components of the water river basin are uncertain; the underlying hydrological processes due to their stochastic nature, the management objectives and evaluation criteria due to uncertainties concerning future conditions.

Although the stochasticity of the hydrological cycle is frequently being refereed to as the major source of uncertainty in river basin planning, some authors have demonstrated that the variations in economic variables, political decisions and other factors are much more important with regard to river basin management (Rogers, 1997).

The objective of this chapter is to discuss some methods for dealing with uncertainties and concentrates on methods that can be used in water resources planning in general and in the framework of the WaterStrategyMan project in particular.

Sensitivity analysis

A commonly used simple technique to deal with the effects of uncertainty is to vary one or more uncertain parameters and than to ascertain the impacts on the systems performance. This approach is commonly known as sensitivity analysis and is aimed at identifying those parameters to which the system is particularly sensitive.

Stochastic simulation (Monte Carlo Simulation)

In stochastic simulations, the first step is to generate random variables representing any input value such as rainfall, streamflow etc. based on the known probability distribution function of these variables. The parameters for the probability function are computed from time series of the respective variables. Basically, the purpose of the stochastic simulation is to obtain a probability function of the output given the probability distribution function of the input. The basic concept is schematically depicted in Figure 9.

Stochastic optimisation)

In the same way simulation models must be extended to incorporate random processes, optimisation models have to incorporate mathematical expressions for variables under uncertainty. Many optimisation techniques such as Stochastic Dynamic Programming (SDP) and stochastic linear programming can be seen as a extension of the deterministic case.

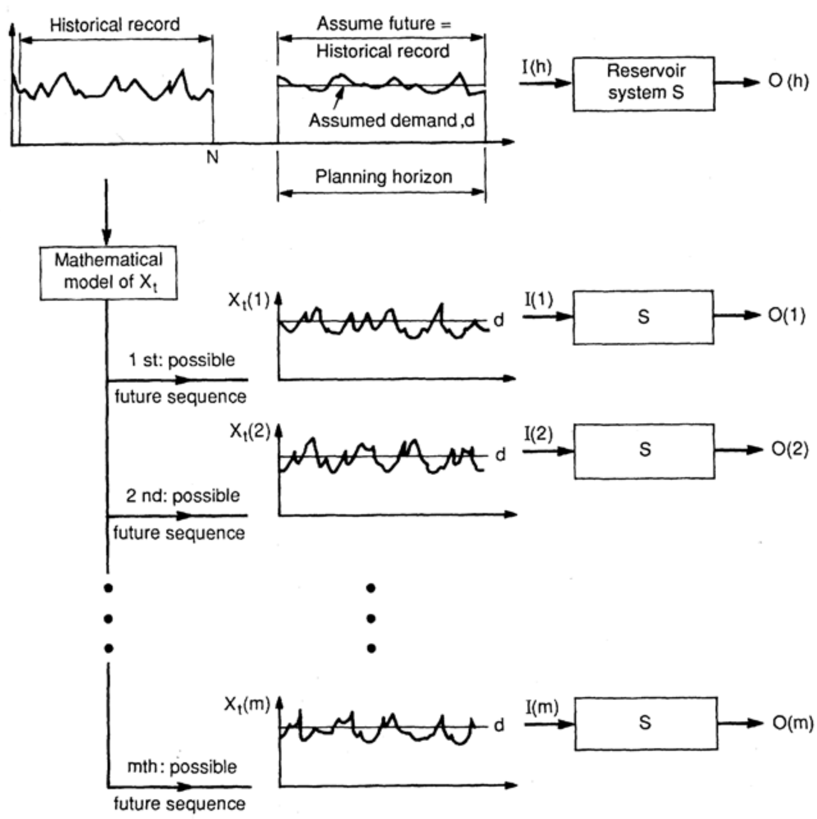


Figure 9: Schematic representation of a hydrologic system for Monte Carlo simulation

Fuzzy-set theory

Fuzzy sets are used to describe uncertainty in a non-probabilistic framework. They group classes of data with boundaries that are not sharply defined. The benefit of extending crisp theory and analysis methods to fuzzy techniques is the strength in solving real-world problems, which inevitably entail some degree of imprecision and noise in the variables and parameters measured and processed for the application. Accordingly, linguistic variables are a critical aspect of some fuzzy logic applications, where general terms such a "large," "medium," and "small" are each used to capture a range of numerical values.

2.6 Discussion and Recommendations

2.6.1 Water Resources Availability Forecasting

Forecasting the water resources availability is perhaps the most difficult task in long-term river basin management as it is influenced by a number of factors that cannot be quantified.

It is therefore suggested to use a scenario approach similar to the method that is called water-year method in the WEAP package that defines water resources availability as percentage of long-term water average values.

In any case, the scenarios of water availability should reflect the seasonal variability of the water supply, e.g. by applying a long-term pattern of monthly availability to the scenarios.

This approach has the advantage that it incorporates long-term observed values as well as monthly fluctuations that can be changed individually. It is suggested that a simple decomposition approach similar to the one presented in chapter 0 is used.

2.6.2 Demand forecasting

Given the anticipated planning period in the WaterStrategyMan projcet (20 to 30 years) and the high level of uncertainty in forecasting urban demand described in the previous chapter, the only approach that can be reasonably well justified is a scenario-based ("What if..") method that models the water resources system for a given set of future conditions (e.g. "business as usual" plus 10, plus 20 percent etc.) that is similar to the water-year method used for hydrological scenarios. The seasonality of demand can be well represented by the approach presented in chapter 0. In doing so, a given scenario cannot only be based on water availability for the whole year but can also take into account different patterns of demand seasonality.

A simple model that uses activity levels and future scenarios of population development can be implemented. In any case, the model should be able of representing some water management interventions such as leakage detection and control programmes or measures to decrease the unaccounted for water (UFW). For industrial water demand a distinction should be made between consumptive and non-consumptive use.

Econometric models that link the consumption of water to prices do not seem to be suitable for the WSM purposes (see report on WP 4.2)

2.6.3 Modelling approach

The modelling approach should encompass supply as well as demand-site issues and environmental, economic and social aspects. The model-independent network representation of the river basin compromising of arcs and nodes should be implemented. The nodes in this network represent physical as well as non-physical entities such as abstractions, intakes etc. that can be linked to the corresponding objects (water works, demand nodes, control nodes, storage nodes etc.). Each node is characterised by its location and its connectivity to the downstream and upstream nodes.

The water allocation model should consist of a network analysis tool that is capable of assigning both demand site and supply preferences.

A temporal resolution of one month does seem to be appropriate and the spatial scale should be oriented on the hydrologic boundaries (i.e. river basin) wherever possible.

As the model is to be used in a number of different river basin, the model has to be capable of coping with a wide range of possibilities. However, given the amount of required data for detailed physically based models (rainfall-runoff models, gw models etc.) such models do not seem to be applicable for the purpose of the WSM DSS. The

DSS should rather use conceptual models that integrate all aspects of river basin management. Models and variables have to be defined in a way that any water management intervention and its implication on the systems can be represented.

It is therefore of crucial importance to procure the minimum data indicated in and in Table 3 in order to represent the aspects of water management described above.

3 Section II: Methods for Estimating Costs

3.1 Introduction

The cost of environmental services is not just the cost of the goods and services that are required in order to make the environmental resource available for use, but also the costs that society has to bear by means of reduced opportunities of using the "natural capital" in alternative ways, and the costs that are necessary to maintain and improve the quality and quantity of the "natural capital" itself up to a level that is considered sufficient in terms of long-run sustainability (Antonio Massarutto - 2002).

Strategies to bring the prices and recovery of bills closer to the cost will be developed. In this part, the concept of full-cost recovery (FCR) will be described and analyzed. This concept is now the guiding principle for setting prices for water services and financing the water industry. European legislation has adopted the concept of full-cost recovery (FCR) among the guidelines for water resources management policies. This concept is intended to be an application of the polluter-pays principle (PPP) and provide a basis for charging the use of environmental resources in such a way that provides users with appropriate incentives for a sustainable use of the environment (European Commission - 2000 / Antonio Massarutto - 2002).

In the conclusion we make a proposal of nine simple indicators and methodologies that could be used in the Decision Support System.

Appendix 1 and 2 give the different indicators which are recommended by WATECO and indicators for a qualitative assessment of the economic situation in relation with water resource management. The aim of the following sections is to proposed a methodological approach for cost estimation. Three aspects of the Economics of water supply are described and analysed:

1. Water supply cost – Methodology for the estimation of the sum needed to reach sustainability of technical systems;
2. Non accounting opportunity costs;
3. Methods for the estimation of environmental costs.

3.2 Water supply cost – Methodology for the estimation of the sum needed to reach sustainability of technical systems

For the estimation of cost recovery, physical and financial data related to the infrastructure and water production costs should be presented along with methods about the estimation of the present value of the infrastructure, the monetary value of water services and environmental costs. The sum needed to reach sustainability of technical systems and the revenues from water billing should be computed. Finally, the rate of cost recovery will be estimated. The following figure illustrates this procedure.

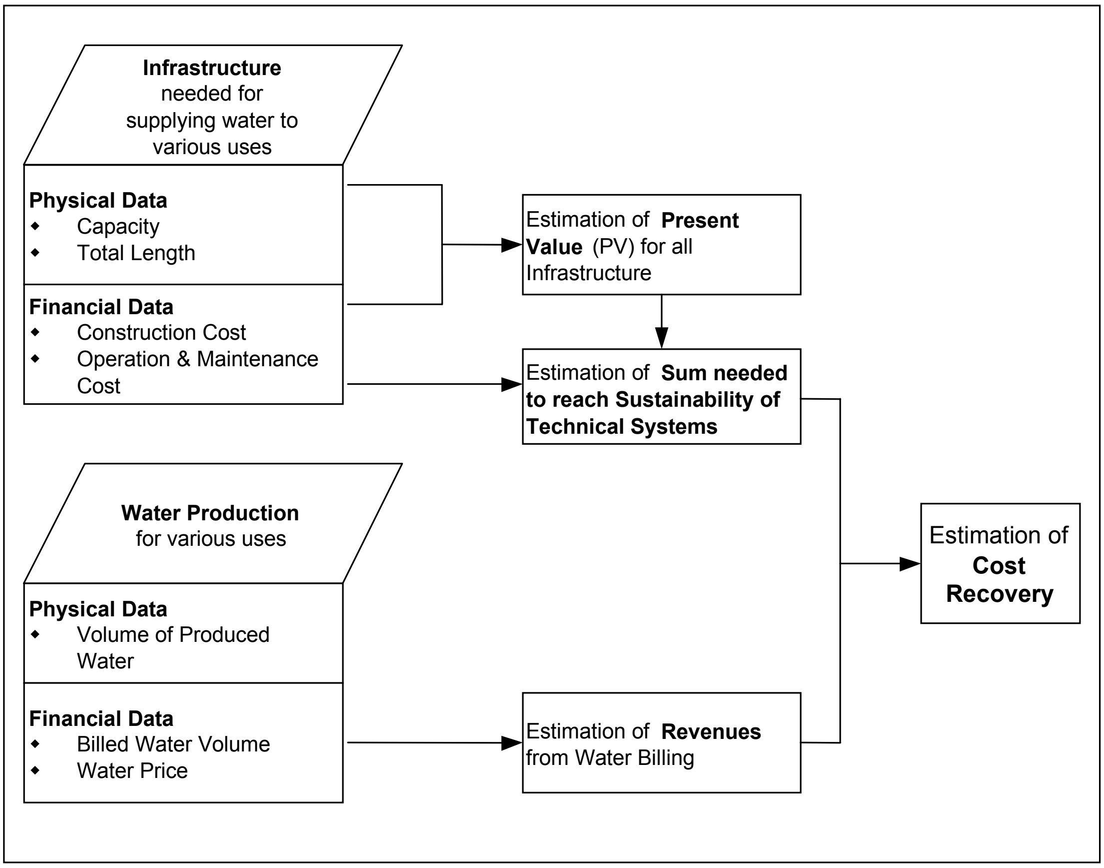


Figure 10: Cost recovery estimation

3.2.1 Average Cost and Rate of Cost Recovery from Water Billing:

The process used to estimate the average cost and the rate of cost from water billing is based on the WATECO Guidance document (see proposed indicator of WATECO in appendix 1).

Table 4 summarizes the key tasks and the relevant questions on cost recovery estimation according to this document.

Table 4: Key Tasks and questions in analyzing and reporting on cost-recovery (WATECO – Guidance document – Draft

Key Tasks	Questions
1. Define scale of assessment	What are the spatial and hydrological characteristics of the water body? Who will be affected by the measures? To what extent? Directly or Indirectly?
2. Identify types of cost and benefits	What types of cost and benefits can be derived from the measures? What types of cost and benefits can reliably be estimated? Are they quantitative, qualitative or monetary? Which cost and benefits appear significant?
3. Choose methodology	Which cost and benefits should be derived quantitatively, qualitatively and monetarily? Is it necessary to apply different methods? What resources are available for original research (time and finance)?
4. Collect data	What studies have been done before? Do we need to create first hand data or can we rely on other sources?
5. Assess cost and benefits	Are quantitative, qualitative or monetary impacts important? Have all types been given sufficient weight? How can all these different impacts be presented in a way that facilitates decision-making?

Water Uses

The available water volume must be allocated to the different users in order to satisfy the demand. The main users can be categorised as:

1) Permanent Population
2) Tourism
3) Irrigation
4) Industry
5) Power Generation

Data requirements related to infrastructure

Data related to infrastructure contain information about water supply systems, distribution networks for domestic and agricultural use and treatment plants. The physical data analysis includes information about the technical characteristics of infrastructure needed for the different uses. The financial data analysis provides information related to the different parts of infrastructure such as construction costs, present value, depreciation period and operation and maintenance costs of the water produced per year for the satisfaction of the water needs of each sector (*Table 5*).

Table 5. Physical and Financial Data Requirements Related to Infrastructure

Physical data		Financial data		
Parts of Infrastructure	Capacity / Length	Construction Cost	Depreciation Period	O & M Costs
Dams	m^3	€ / m^3	Years	€ / m^3 /year
Water catchment	m^3/days	€ / m^3	Years	€ / m^3 /year
Water treatment plant	m^3/days	€ / m^3	Years	€ / m^3 /year
Distribution systems of freshwater treatment	m^3/days	€ / m^3	Years	€ / m^3 /year
Waste water treatment plant	Population equivalents	€ / population equivalents	Years	€ / population equivalents /year
Distribution systems of waste water treatment	Population equivalents	€ / population equivalents	Years	€ / population equivalents /year
Water distribution net	m	€ / m	Years	€ / m /year
Sewer network	m	€ / m	Years	€ / m /year
Irrigation network	m	€ / m	Years	€ / m /year

3.2.2 Methods for the estimation of present value

The present value of the different parts of infrastructure can be estimated by using one of the following methods:

a) *Historical value* (used as method of valuation of capital asset): It is the value of assets at the price they were originally purchased. Because of inflation, this value bears no relation with what it would actually cost today to replace those assets – therefore, it is not the best measure for estimating economic costs.

b) *Current value* (used as method of valuation of capital assets): It is the historical value multiplied by an inflation index. Calculating this value raises a number of issues:

- Estimating the inflation index may be open to interpretation [should the general inflation or the construction (consumer ?) price index be used];
- This method does not take into account technical progress: a water treatment plant that costs a given amount 10 years ago might cost half today thank to technical progress.

However, this method is relatively easy to apply and is more appropriate than the historical value method.

c) *Replacement value* method (used as method of valuation of capital asset). This method estimates the present value of an asset from the current cost of replacing it for an identical service level. The advantage of this method is that it allows taking into account technical progress. However, it might be difficult, costly and time-consuming to apply to all the capital stock.

3.2.3 Methods for monetary valuations of water services

a) *Discount rate.*

b) *Operating cost*: all costs needed to maintain the operation of an environmental facility (e. g. material and staff cost).

c) *Maintenance cost*: cost for maintaining existing (or new) assets in good functioning order until the end of their useful life.

d) *Capital cost*:

- New investments: costs for new investment expenditures and associated costs (e.g. site preparation costs, start-up cost, legal fees)
- Depreciation: the depreciation allowance represents an annualized cost of replacing existing assets in future. The estimation of depreciation requires the definition of the value of existing assets and a depreciation methodology.
- Cost of capital: It is the opportunity cost of capital, i.e. an estimation of return that can be earned by alternative investments. The cost of capital applied to the asset base (new and existing, give the profits that investors are expecting to gain from their investments).

e) *Administrative cost*: administrative cost related to water resource management.

f) *Other direct cost*: this mainly consists of the costs of productivity losses due to restrictive measures.

3.2.4 Annual water cost with sustainable use of technical systems

The annual water cost of water when there is a sustainable use of the technical systems (Cost of sustainability of technical systems) can be calculated using the following formula:

$$C_{CSTS} = \sum_{ij} \left\{ C_{i,j} \cdot \left[\frac{(CostPV_i)}{t_i} \right] + CostAO_i + CostAM_i \right\} + \sum_{kj} \left\{ TL_{k,j} \cdot \left[\frac{(CostPV_k)}{t_k} \right] + CostAO_k + CostAM_k \right\}$$

Where *i*, *k*, *j* the parameters that are presented in Table 6.

Table 6. Parts of infrastructure and different water users

Plant *i*	**Total network length *k***	**Users *j***
Dams	Water distribution net	Permanent population
Water catchment	Sewer network	Seasonal population
Water treatment plant	Irrigation network	Irrigation
Distribution systems for freshwater treatment		Industry
Waste water treatment plant		Power generation
Distribution systems for waste water treatment		

and:

C_i:	Capacity of i
$CostPV_i$:	Present value of i
$CostAO_i$:	Average Operating Costs of i
$CostAM_i$:	Average Maintenance Costs of i
t_i:	Depreciation period (useful life) of i
TL_k:	Total length of network k
$CostPV_k$:	Present value of network k
$CostAO_k$:	Average Operating Costs of network k
$CostAM_k$:	Average Maintenance Costs network k
t_k:	Depreciation period (useful life) of network k

3.2.5 Data requirements for water billing

In order to calculate the revenues from water billing, data about the water volume that is produced every year for the different uses is necessary. Also, the amount of the produced volumes that are charged and the different prices for the various uses have to be calculated (Table 7).

Table 7. Required data for water billing

Use	Volume of water produced	Billed volumes of water	Water price
Permanent population	m^3 / year	m^3 / year	€ / m^3
Seasonal population	m^3 / year	m^3 / year	€ / m^3
Irrigation	m^3 / year	m^3 / year	€ / m^3
Industry	m^3 / year	m^3 / year	€ / m^3
Power generation	m^3 / year	m^3 / year	€ / m^3

3.2.6 Revenues from water billing

Using the collected data the sum of revenues from water billing (Revenues from water billing) is calculated following the formula:

$$\mathrm{Re}\,venues\ from\ water\ billing = \sum_j \left(Billed\ Volume_j \cdot \mathrm{Pr}\,ice_j \right)$$

Where j = the different water uses (Table 3).

3.2.7 *Rate of cost recovery from water bill to reach sustainability of technical systems*

The following formula calculates the rate of cost recovery from water billing to reach sustainability of technical systems (RCR):

$$RCR = \left(\frac{Revenues\ from\ water\ billing}{Cost\ of\ sustainability\ of\ technical\ systems} \right) \cdot 100$$

Nota Bene: Options to rapidly increase RCR:

1. decrease water volume used by different users to get closer to billed water volume for each use, if the difference is due to leaks.
2. Increase billed water volume of the different uses to reach the water volume that is actually used by the different users, if the difference is due to the fact that some water is used but not invoiced.
3. Increase water price of different uses.

3.2.8 *Cost recovery: the issue of subsidies*

The Guidance document draw up by WATECO working group (WATECO working group-2001) points out the importance of the issue of subsidies.

The polluter pays principle requires that users pay according to the cost they generate. However , subsidies reduce user's contribution to the full cost of water services and, according to the theory, disable price incentives to use resource in a sustainable manner.

Subsidies are allocated to either providers, users or polluters in different way. They can be paid directly by the central or local government:

to the providers of water services in the form of investment subsidies (capital subsidies, lowering fixed costs);

to the providers of water services in order to co-finance the operation of the infrastructure (operational subsidies, lowering variable costs);

to water users (income transfers, lowering the price / charges paid by the user).

In addition, subsidies can be paid indirectly by users / Polluters paying the cost of other users / Polluters. Cross subsidies may arise between different users (household, agriculture, industry, different regions (dry and wet, populated or less populated) and / or different users (rich or poor, small or large users,…) (WATECO working group-2001).

When user groups pay only part of the costs of a water service, the balance of the cost will have to be paid or subsidied by other. These others can be the public with a large contribution through general taxation (tax revenue being used by the central government to subsidise the supply of water services in ways described above) or other groups that pay a larger fraction of the total costs (including resource and environmental cost) than they generate (WATECO working group-2001).

It will be very important that the Decision Support System integrates an indicator on the subsidies practices.

We propose the following indicator:

Rate of subsidisation of water service = R_{sub}

$$R_{sub} = \left(\frac{\text{Cost supported by water services} - \text{Revenues from water billing}}{\text{Cost supported by water services}} \right) \cdot 100$$

Or

$$R_{sub} = \left(\frac{\text{Subsidies}}{\text{Cost supported by water services}} \right) \cdot 100$$

Nota Benne:

- Costs supported by water services ≤ Cost of sustainability of technical systems;
- Costs supported by water services = Revenues from water billing + subsidies;
- Cost of sustainability of technical systems – Costs supported by water services = Sum missing to reach the sustainability of the technical systems

3.3 Non accounting opportunity costs

Opportunity costs are a fundamental element of economic decision making. An opportunity cost is the value of forgone option resulting from a decision. Opportunity costs are accounting costs in different sectors like labor, energy… In these cases, there is no difference between the accounting and economics perspectives of what counts. But in the case of water service some opportunity costs do not become accounting cost. When these non accounting opportunity costs are significant, accounting –based prices can seriously misvalue water or nonwater aspect of the rate structure (Griffin Ronald – 2001).

In the following section we describe and analyse three distinct non accounting opportunity costs:

- Marginal Value of Raw Water;
- Marginal User Cost;
- Marginal Capacity Cost

3.3.1 Marginal Value of Raw Water

For the most part, water prices are the consequence of value added by the utility for the:

- administration,
- conveyance,
- storage,
- pressurization,
- treatment

of water (freshwater and wastewater).

Most of the time, a zero or near zero value is assigned to raw water.

The fact that there is no value assigned to the raw water resource is not necessary bad, because, in most times or regions water may not be scarce. In these cases, a givenwater use involves no sacrificed alternative uses. But when or where water is scare, water prices should incorporate the marginal value of raw water (Griffin Ronald – 2001).

3.3.2 Marginal User Cost

In a situation relating to non-renewed water supplies , water use causes a sacrifice in alternative future uses. When scarce ground water is used now, those units of water will be unavailable for future use. The technical name for this opportunity cost is marginal user cost, defined as the value of sacrificed future use, discounted to present value (Griffin Ronald – 2001).

3.3.3 Marginal Capacity Cost

Capacity expansion is a very important facet of water planing. The choice of new facilities, the timing and the sizing of the facilities are crucial decisions. Such decisions often have large cost implications for the utility. A very important aspect of new investments in water supply capital is their lumpiness. That means, in many instances, it is efficient to undertake project of greater scale than necessary to satisfy the current water demand. For this reason, utilities tend to expand in spurts. As system capacity becomes limiting, utilities undertake the next project. Upon project completion, excess capacity exists for accommodating more demand growth, but growth eventually consumes this capacity and the next project is engaged (Griffin Ronald – 2001).

The economic view is that optimal project timing maximizes the present value of net benefit. This is a more demanding criterion than requiring project to pass a cost benefit test. Pursuit of this goal typically implies that it is not rational to build project in advance of demand. Moreover, it is often economically efficient to not build projects when demand for their capacity is slight. Premature construction is costly due to the time value of money and capital depreciation. Following an efficient path for the timing of lumpy project therefore means that there are periods during which water supply capacity is less than water demand. During such periods, there is a third non accounting opportunity cost that must be taken into account : marginal capacity cost (Griffin Ronald – 2001).

The effect of including marginal capacity cost in price is to efficiently ration available capacity during capacity-constrained periods. If price omits marginal capacity cost during these periods, the demand will exceed supply. **According to the theory**, two inefficient consequences can then occur:

- the shortfall will have to be accommodated through some non price allocation policy, implying that the marginal value of finished water will not be equivalent across all clients;
- second, deficient pricing will create a prevailing opinion, among both clients and responsible utility managers, that projects should be initiated to rectify the perceived shortage. If such action is taken, the investment regime will be accelerated beyond an efficient pace and the present value of net benefits will be decreased.

The problematic aspect of incorporating marginal capacity cost in price is that it's changing level over time. A given supply capacity becomes more constraining over time due to growth in demand, but upon completion of a supply-enhancing project, marginal capacity cost commonly falls to zero and remains there until further demand growth eliminates the excess capacity (Turvey, 1976). For this reason, marginal capacity cost rises and fills over time. Its incorporation in rates brings about a long-term cycling in rates which may be objectionable to customers. Recognition of this issue has led to the creation of smoothing substitutes for marginal capacity cost (Mann et al., 1980), with the acknowledgement that some sacrifice in economic efficiency is made whenever rates do not embed true marginal capacity cost (Griffin Ronald – 2001). According to this theory, in the absence of marginal capacity cost – inclusive price, "we need only know demand elasticity in order to estimate a price increase sufficient to assuage the excess demand. This rate hike is marginal capacity cost" (Griffin Ronald – 2001).

3.3.4 Concepts of opportunity costs and scarcity rent and consequences on optimal allocation of a limited water resource among competing users (FEINERMAN Eli – 2002)

This chapter aimed at shedding some light on the concepts of opportunity costs and scarcity rent. The explanations are based on graphical analysis. The graphs are presented at the end of the chapters.

⇨ **Introductory Comments**

● Water is not *generally* scarce in terms of quantity: the availability of seawater desalination means that there is abundant water for the world as a whole and for any country that has a seacoast. But, of course, seawater desalination is expensive and, conveyance facilities to locations far from the sea may be non-existent or themselves expensive. There are two lessons to be learned here. First, water scarcity is a matter of cost and value, not merely of quantity. Second, the value of water and also its scarcity will be different in different locations.

● Microeconomy is basically about the allocation of scarce resources and about the relation of the value of those resources to their scarcity and their allocation. The fact that water is essential for human life makes water and its allocation very important, but it does not exempt it from the principles of microeconomics.

● No matter how important water is and no matter what special values are believed to attach to water in certain uses (drinking water are essential for human life), it is irrational to value water at more than the cost of replacing it. Hence **the possibility of seawater desalination places an upper bound on the value of water**.

⇨ Water Allocation by Prices

(a) The case of costless water supply.

The value of water does not merely consist of direct costs such as extraction, treatment, and conveyance. It is convenient to start with a single water lake, with limited amount of available water, of, say $\bar{Q}$ cubic meter that is used to irrigate two different agricultural plots.

Suppose, for simplicity, that **it costs nothing** to extract and use water from the lake in terms of direct costs and that the **total** annual demand for water by the two plots, at zero cost, is greater than the available supply $\bar{Q}$ (say the annual renewable amount in the lake). In that case, although the **direct costs** of utilizing the water are **zero** (by assumption) the **value** of water in the lake is **not zero**; if there were additional water, there would be additional positive benefits to the farmers and they would be willing to pay for those benefits. Before describing the optimal allocation scheme of $\bar{Q}$ between the two plots, it is important to note that:

(i)The fact that the value of the water in the lake is greater than the direct costs of supply (zero in this example) means that the lake's water has a positive scarcity rent (a concept that will be defined later); and

(ii) The scarcity rent is a measure of scarcity. Indeed, with very small agricultural fields, the same water, equally essential for human life, might not be scarce at all. In that case, its scarcity rent would be zero.

To derive the optimal allocation of the scarce water between the two plots, say plot A and plot B, it is convenient to use a graphical analysis. Assume that plot A is very fertile and plot B is less so. The higher quality of plot A is reflected in its value of marginal product of water (VMP_A), which exceeds the value of marginal product of plot B (VMP_B). Does this difference in quality means that only plot A should be irrigated? The answer is negative and the optimal allocation of $\bar{Q}$ between the two plots is illustrated in Figure 2. The horizontal axis shows the two amounts of water, $\bar{Q}$, available to the two plots (at zero costs, by assumption). As we move from zero to the right on this axis, water used on plot A, increases. Plot B has its horizontal axis reversed -- more water allocated for plot B means moving from the right to the left.

Q_A and Q_B $(Q_A + Q_B = \bar{Q})$ are the optimal (profit maximizing) allocations of water to plots A and B, respectively. Profit maximizing allocation requires the value of marginal product of water on each plot to be the same (i.e. $VMP_A(Q_A) = VMP_B(Q_B)$). Notice that water are used on both plots of land, but relatively more on plot A—the most fertile site. With zero costs, **total** profit from plot A is the area *abcd* and from plot B total profit is the area *cdef*. In practice, the optimal allocation can be obtained by either: (a) Allocation of water quotas: Q_A cubic meters to the farmer cultivating plot A and Q_B cubic meters to the farmer cultivating plot B; (b) Allocation by price: setting the price of water at a level of $P\ (= VMP_A(Q_A) = VMP_B(Q_B))$. At that price, the farmer cultivating plot A (farmer A) will purchase Q_A cubic meters and the farmer cultivating plot B (farmer B) will purchase Q_B cubic meters. The price *P* can be viewed as a market cleaning price.

Opportunity Costs and Scarcity Rent

Note that if farmer A will increase water use by 1 cubic meter, the amount of water available for farmer B will decrease by 1 cubic meter. The reduction in the **value** of production on farm B is the cost to society resulting from the increased water use on farm A. Similarly, the reduction in the value of production on farm A is the cost to society resulting from the increased water use on farm B. This cost, which is equal to P dollars per cubic meter, is an **opportunity cost** – the benefits forgone when a scarce resource is used for one purpose instead of the next best alternative.

An important comment: It should be emphasized that ALL costs in the economy are always "opportunity costs": energy, capital and labor used to extract and convey water to farm A are not available to serve farm B and do not contribute to its production. But, for the sake of presentation, we will adopt the commonly used term of "direct costs" to represent here the costs of the relevant inputs (like energy, capital, labor, etc.) that can be bought in the markets and their prices are known. The input "natural stock of water in the lake" is the only input in our example that cannot be purchased in the markets.

In this example, where the direct pumping and conveyance costs are assumed to be zero, the opportunity costs is also the **scarcity rent** of water -- rent (per unit) of a scarce resource (water in our case) is a surplus, the difference between the opportunity cost of water (equal to the market cleaning price P) and the per unit (marginal) direct costs (such as extraction, treatment, environmental and conveyance) of turning that natural resource into relevant products (agricultural crops in our example). The scarcity rent is the result of the fact that the total amount of water in the lake is scarce (the **total** annual demand for water by the two plots, at zero cost, is greater than the available supply) and is limited to $\bar{Q}$. In our example, the optimal price, P, is also the **scarcity rent** of water since reduction of 1 cubic meter of the limited supply $\bar{Q}$ will reduce the value of products in the (two-farm) economy by P dollars. **In case that water in the lake is not limited—the opportunity costs and the scarcity rent are zero. See an example in Figure 2a.**

It should be emphasized that **opportunity costs, like any other concept of costs, has a meaning only when water allocation is optimal, like in Figure 2**. If , for example, water allocation is arbitrary and farm A receives more water than Q_A and farm B receives less water than Q_B, we cannot talk about opportunity costs since we can increase the total value of production in the two-farms economy without adding water above $\bar{Q}$, just by transferring water from farm A to farm B.

Before proceeding, it is convenient to present the result of Figure 2 in 3 separated figures—Figures 3a and 3b represent the demand for water in farms A (VMP_A) and B (VMP_B), respectively and Figure 3c represents the aggregate demand in the two-farms economy ($D = VMP_A + VMP_B$).

Let us now discuss the more real case under which the direct costs are positive.

(b) The case of positive direct costs.

Let's assume for simplicity that the direct costs of extracting, treating and delivering water from the lake to the plots (including labor and capital) are equal to MC_1 $\$/m^3$ (when marginal costs are constant and independent of the level of water supply, as we assume here for simplicity, they are also equal to the average costs). Assuming that $MC_1 < P\ (= VMP_A(Q_A) = VMP_B(Q_B))$, all the available water from the lake will be used. Optimal allocation of $\bar{Q}$ will be obtained by setting the price of water at a level of $P\ (= VMP_A(Q_A) = VMP_B(Q_B))$, which is equal to the optimal price when direct costs are zero, (see Figures 4a-4c), -- farmer A will purchase Q_A cubic meters and farmer B will purchase Q_B cubic meters (which are equal to the respective quantities presented in Figures (3a) and (3b)).

The **scarcity rent** in this case, denoted by λ (see Figure 4), is defined by $P - MC_1$, i.e., the (market cleaning) price of water (which is equal to $VMP_A(Q_A)$ as well as to $VMP_B(Q_B)$) minus the marginal direct costs. If we will reduce the quantity of water available for one of the farmers (either A or B), his marginal benefits will be reduced by VMP_A (or VMP_B) but, at the same time, the cost of MC_1 will be saved, implying a net loss of $\lambda = P - MC_1$. Or equivalently, increasing the amount of water in the lake by 1 cubic meter will increase the **net** marginal benefits for the two-farms economy by λ dollars. If the additional cubic meter will be delivered to farm A or farm B or will be divided between the two farm, its contribution to the total benefits of the economy is equal to the value of marginal product of water, $VMP_A(Q_A) = VMP_B(Q_B)$, minus the marginal direct costs associated with the supply of this cubic meter, MC_1.

The opportunity costs (*P*) of water is equal to the sum of the marginal direct cost and the scarcity rent, i.e., $P = MC_1 + \lambda$ **.**

Note: the full cost of water in this example is marginal direct cost + the scarcity rent and NOT marginal direct cost + opportunity costs (which means double counting of the marginal direct costs). In this chapter we have ignored environmental costs to simplify the presentation. However, it can be easily added to the analysis as being a component of MC_1**.**

If the direct marginal costs per cubic meter are larger than

$P \ (= VMP_A(Q_A) = VMP_B(Q_B))$, e.g., MC_2 in Figures 4, than the optimal price should be $\tilde{P} = MC_2$ and the optimal total amount of water diverted from the lake to agricultural production is $\tilde{Q}(< \bar{Q})$, see Figure 4c. In that case, the water in the lake is not scarce (if we will decrease the total amount of water in the lake by one unit it will not affect agricultural production since this unit is idle), **the scarcity rent is zero** (i.e., $\lambda = 0$) and the opportunity costs are equal to $\tilde{P} = MC_2$.

Note: now the full cost of water is only the marginal direct costs be MC_2 **and NOT marginal direct cost + opportunity costs.**

(c) Water supplied at different locations.

We end this note by assuming now that farm A is very close to the lake (as was implicitly assumed before) while farm B is located far away from it and the cost to transport water from the lake to farm B is *t* dollars per cubic meter. The marginal direct costs of extracting and treating the water at the lake is MC_1. The conveyance and application costs to plot A are assumed to be zero for simplicity while the conveyance costs to plot B are *t* dollars per cubic meter. Allocation of $\bar{Q}$ by prices requires different water prices for plots A and B, denoted by P_A and P_B, respectively. It can be shown via straightforward mathematical analysis, that under optimal allocation (see Figure 5 for graphical illustration):

(i) $P_A = VMP_A(\hat{Q}_A);\ P_B = VMP_A(\hat{Q}_B);$ where $\bar{Q} = \hat{Q}_A + \hat{Q}_B;$

and the scarcity rent in this case, denoted by λ^* is given by:

(ii) $\lambda^* = P_A - MC_1 = P_B - t - MC_1 \Rightarrow P_B = P_A + t$.

Comparison of the market cleaning prices under this case with *P* of the previous cases (i.e., when A and B are both located by the lake shore) and comparing λ^* with λ of case b yield:

(iii) $P_A < P(\rightarrow \hat{Q}_A > Q_A);\ P_B > P(\rightarrow \hat{Q}_B < Q_B);\ \lambda^* < \lambda.$

Intuitive Explanation. To see that $P_B = P_A + t$ must hold when water allocation is optimal, begin by assuming that $P_B > P_A + t$ at the optimal solution. Then transferring one more cubic meter of water from *A* to *B* would have the following effects: First, since there would be one cubic meter less at *A*, net benefits would decline by $P_A = VMP_A(\hat{Q}_A)$, the opportunity costs of water on farm A. Second, since conveyance costs of *t* would be incurred, there would be a further decline in net benefits of that amount. Finally, however, an additional cubic meter at farm B would produce an increase in net benefits of $P_B = VMP_B(\hat{Q}_B)$, the opportunity costs of water at B. Since, by assumption, $P_B > P_A + t$, the proposed transfer would increase net benefits; hence, we cannot be at an optimum.

Similarly, assume that $P_B < P_A + t$. Then too much water has been transferred from A to B, and transferring one less cubic meter would increase net benefits. Hence, again, we cannot be at an optimum.

It follows that, at an optimum, $P_B = P_A + t$.

Nota Benne: in this example, the opportunity costs of water at farms A ($(P_A = MC_1 + \lambda^*)$ and B $(P_B = MC_1 + t + \lambda^*)$ differ. One may ask how opportunity costs associated with the same type of water are not identical, at the optimal allocation, for the two consumers? It is obvious that two sources of water of different qualities (e.g., salinity levels) should be treated as two different inputs (if they are used for irrigation or industrial purposes) or different products (if they are used for domestic consumption), and have different values and prices. The same logic applies to water of homogenous qualities which are applied at different locations or at the same location at different periods of the year (i.e., summer versus winter). The values of water at different locations (or periods of the year) are different.

Figures

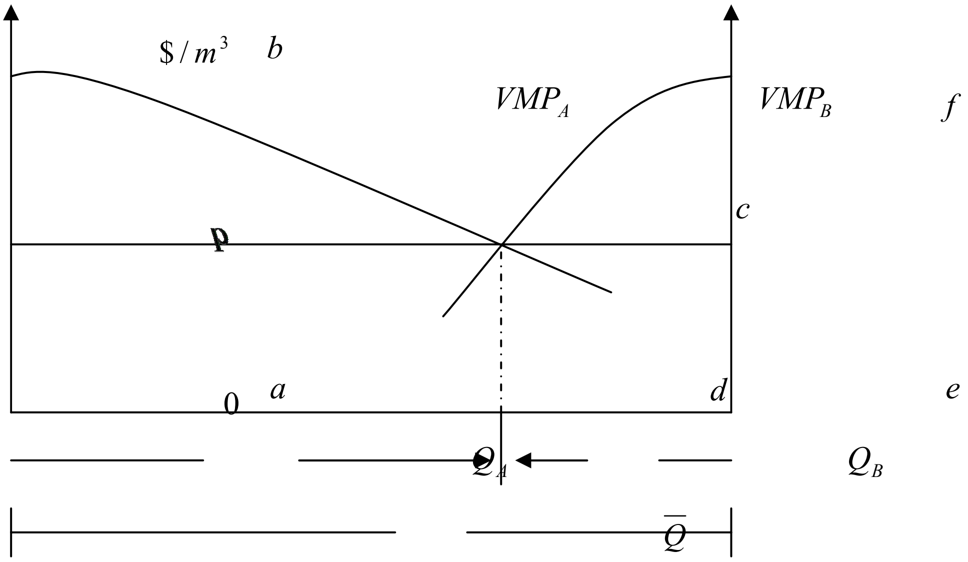


Figure 2: Optimal water allocation on two plots when water are scarce and direct costs are zero. P = opportunity costs.

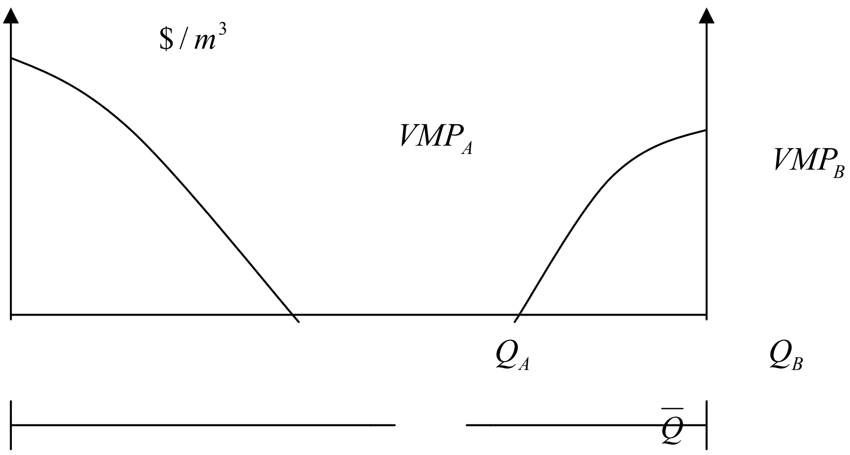
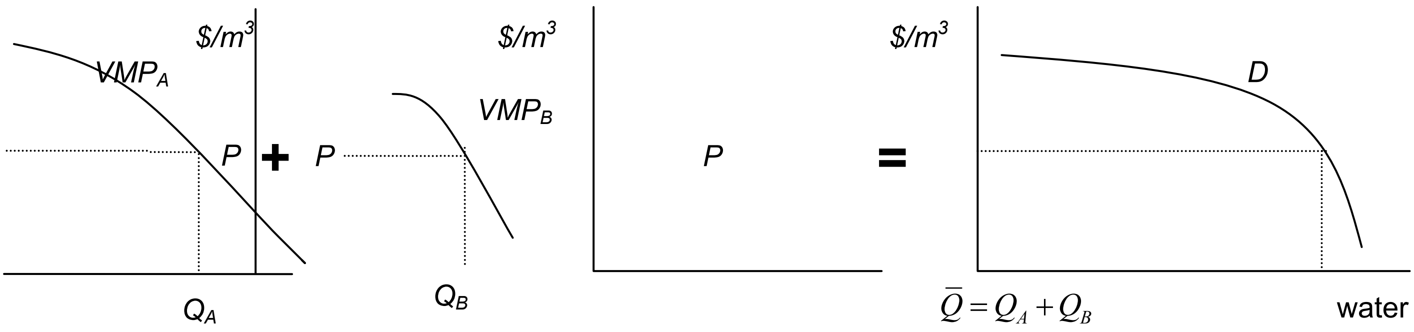
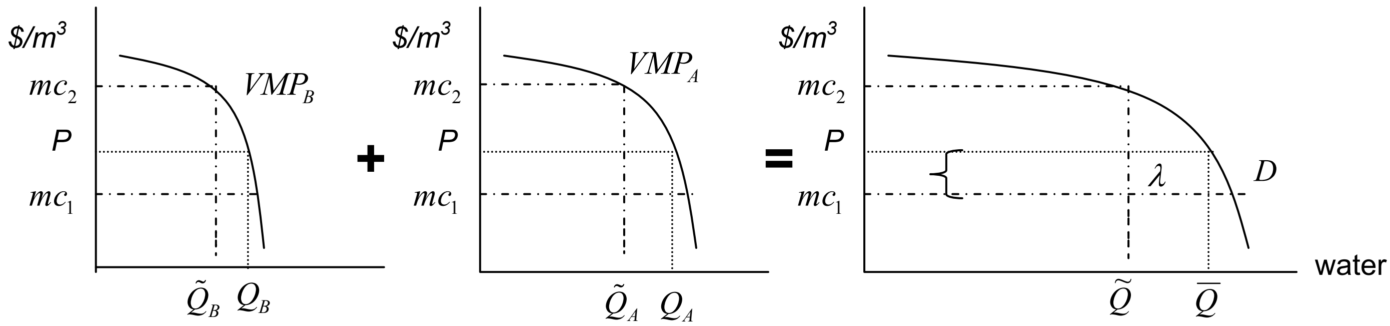


Figure 2a: Optimal water allocation on two plots when water are not scarce $(Q_A + Q_B < \overline{Q})$ and direct costs are zero. Opportunity costs = 0.



(3a) Farm A (3b) Farm B (3c) Economy

Figures 3a-3c: An alternative presentation of the results of Figure 1



(4b) Farm B (4a) Farm A (4c) Economy

Figures 4:

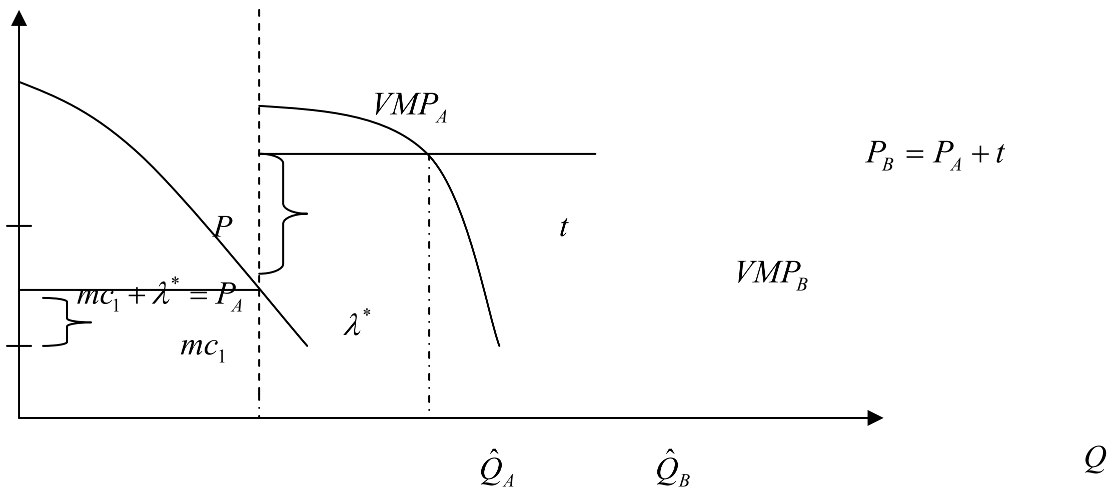


Figure 5:

3.3.5 Efficient allocation of water, water prices and scarcity rents: an example (Gadi ROSENTHAL – 2003)

The following chapter take an example to illustrate the notion of Scarcity rent. The presentation of this example has been done according to a simulation model that will be used for the drawing up of the WaterStrategyMan Decision Support System. Before going in detail in the presentation of the example a short note will present the principles of this simulation model.

⇨ **Note on the WaterStrategyMan Decision Support System (Manoli E., Arampatzis G., Pissias E., Xenos D., Assimacopoulos D. - November 2001 and Progea S.r.l.- January 2003)**

In the WaterStrategyMan Decision Support System, water allocation is achieved through a simulation model. A network representation of the hydrological basin is derived from the database (Fig 22). Nodes represent the connection between these entities. To capture the features of the water systems' function, different types of node are incorporated. These include springs, wells, boreholes, water treatment plants, demand sites, etc. The links correspond to the man-made or natural water conduits, such as pipelines, canals, river reaches, etc. The framework of the network is constructed by connecting the nodes and links according to their physical locations in the water resource system.

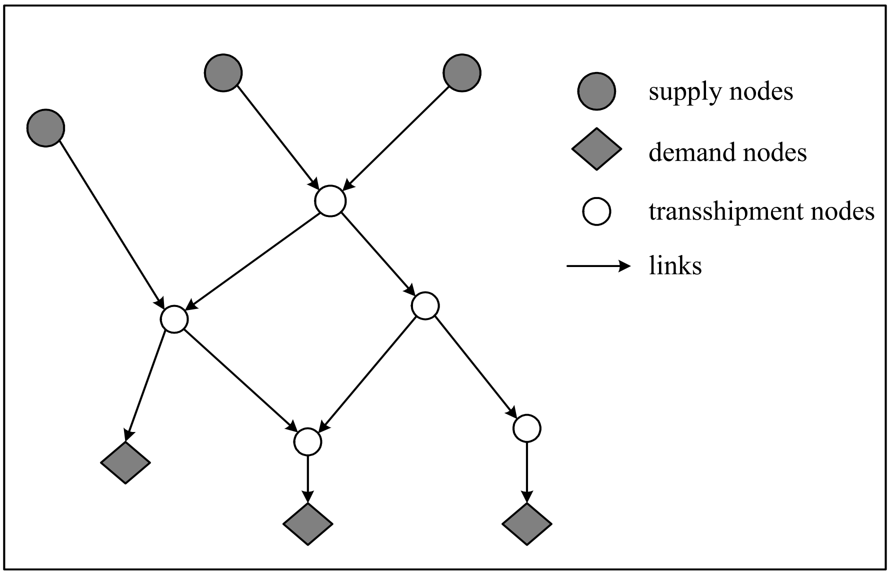


Fig. x Network representation of a water resource system

Each node i can be classified into one of the following three categories (i) supply node which is characterized by a positive monthly supply rate s_i, (ii) demand node which is characterized by a monthly demand rate d_i, and (iii) transhipment node. For each link j

two characteristic variables are introduced: (i) the link capacity c_j which represents the maximum monthly flows allowed (unbounded links can be defined by assigning a sufficiently large capacity), and (ii) the link monthly flow rate f_j (the decision variables of the problem).

In situations of water shortage, a conflict arises of how to distribute the water available at supply nodes, among the demand sites that are connected to them. The model can solve this problem using two user defined priority rules. First, competing demand sites are treated according to their priorities. Each demand site is characterized by a priority, ranged from 1 (highest priority) to 10 (lowest priority). During a water shortage, higher priority demand sites are satisfied as fully as possible. These priorities are useful in representing a system of water rights. On the other hand, supply priorities can be used when a demand site is connected to more than one supply node. These priorities are attached to the links and are useful in ranking the choices of a demand site for obtaining water.

A) Water sources (units / year)

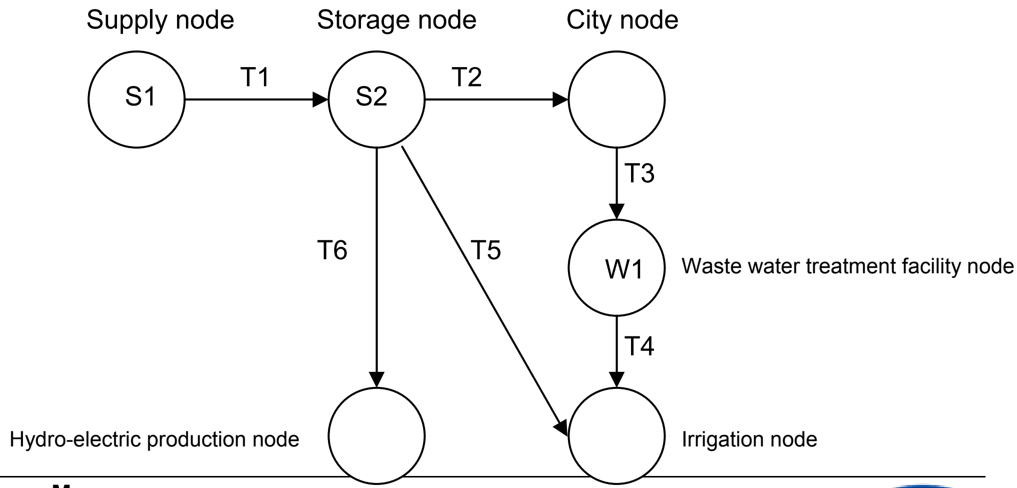
a. Supply node (fresh water): 140
b. Seawater desalination (location near supply node): unlimited
c. Waste water for irrigation: 50

B) Water demand

USER	MAXIMAL DEMAND (units / year)
City	100
Agricultural irrigation	100
Hydro-electric production	100

C) Water supply

a. Scheme of nodes and links



b. Direct costs (production, storage and transportation)

NODE / LINK		COST (cents / m^3)	REMARKS
1)	Supply (S1)	10.0	
2)	T1	5.0	
3)	Storage (S2)	2.0	
4)	T2	4.0	
5)	T3	3.0	Additional cost for agricultural use
6)	Waste water (W1)	1.5	
7)	T4	2.5	
8)	T5	3.5	
9)	T6	2.0	
10)	Desalination (production only)	50.0	Possible location: near S1, no storage needed

c. Average direct costs

TYPE		LOCATION	AVERAGE DIRECT COST (cents / m^3)
Fresh water	From S1	Exit S2	S1 + T1 + S2 = 17.0
	Desalinated	Exit S2	50 + T1 = 55.0
Waste water		Irrigation node	T3 + W1 + T4 = 7.0

D) Assumed average water values (at the user's location)

USER		VALUE (cents / m^3)	DEFINITION
(V1)	City	59	According to desalination cost + transportation (no storage needed)
(V2)	Fresh water agricultural irrigation	28	Product (net added value / m^3)
(V3)	Waste water Agricultural irrigation	21	Product (net added value / m^3)
(V4)	Hydro-electric	20	Product (net added value / m^3)

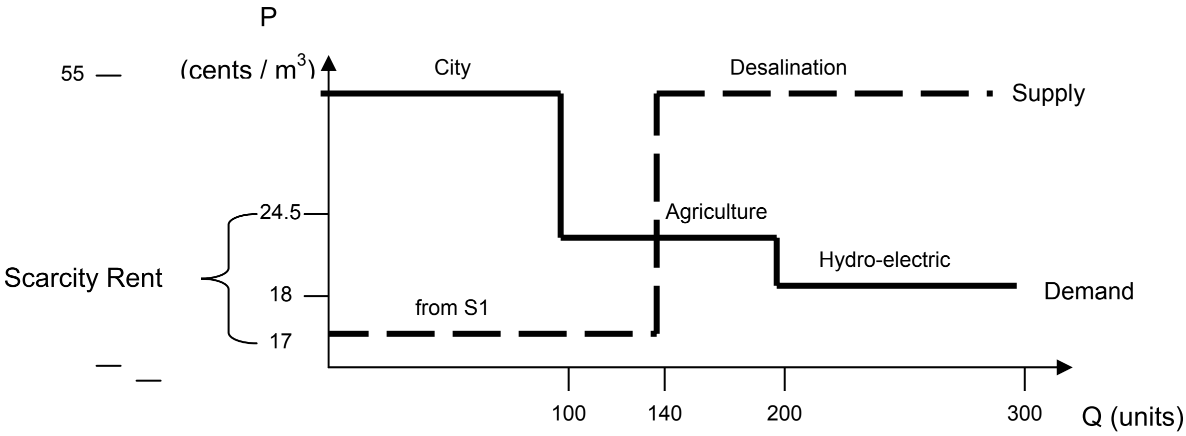
1) All values are defined <u>before</u> any payments of water
2) Values reflect priorities

E) Net value of water

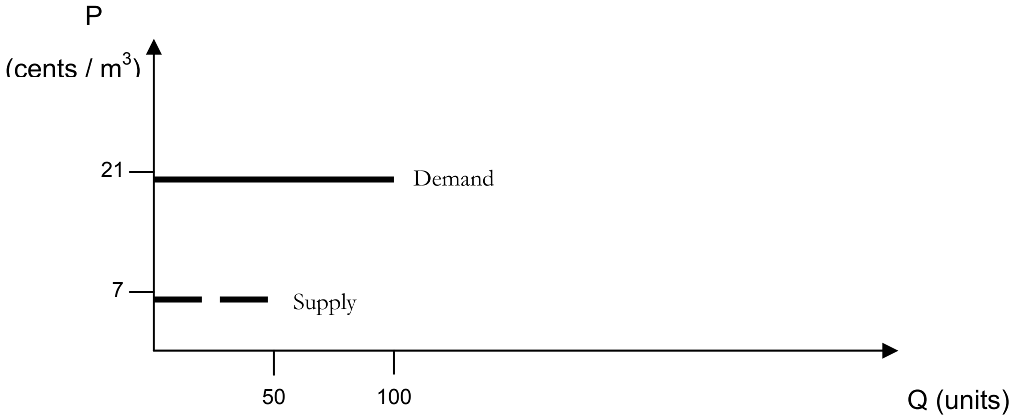
Definition: Net Value = Value at S2 exit = Value - Direct cost "after S2"

TYPE	USER	NET VALUES (cents / m^3)
Fresh water	City	V1 – T2 = 55.0
	Irrigation	V2 – T5 = 24.5
	Hydro-electric	V4 – T6 = 18.0
Waste water	Irrigation	V3 – (T3 + W1 + T4) = 14.0

F) Supply and demand curves of fresh water



G) Supply and demand curves of waste water



H) Results of allocation

Allocation according to net value / m^3 order:

USER	QUANTITY (fresh water)	WASTE	DESALINATION
City	100	-	-
Agriculture	40	50	-
Hydro-electric	-	-	-

I) Prices and scarcity rents

Definition: Equilibrium (24.5 cents) + transportation to user

1) Fresh water prices:

USER	PRICE (cents / m^3)
City	24.5 + T2 = 28.5
Agriculture	24.5 + T5 = 28.0
Hydro-electric	24.5 + T6 = 26.5

Fresh water scarcity rent = V_{EQ} – Average Cost = 24.5 – (S1 + T1 + S2) = 7.5 cents / m^3

2) Waste water price: depends on negotiations between city and farmers, 7 – 21 cents / m^3

Waste water scarcity rent = price – 7 cents / m^3

3.4 Methods for the estimation of environmental costs

In the last few decades economists have devoted significant professional attention to develop and apply methods to place monetary values on environmental services. Economic valuation, at the conceptual level is said to be a measure of the preference people hold for different states of the environment. Valuation, as an empirical exercise, rests on the argument that choices individuals make in market exchanges provide the data that analysts can use to translate people preferences into money terms.

Different methods can be used:

a) *Market methods* (used as a technique for the valuation of environmental costs and benefits): These methods use values from prevailing prices for goods and services traded in markets. Values of goods in direct markets are revealed by actual market transactions and reflect changes in environmental quality: for example, lower water quality affects the quality of shellfish negatively and hence its price in the market.

b) *Cost-based* valuation methods (used as a technique for the valuation of environmental costs and benefits): This method is based on the assumption that the cost of maintaining an environmental benefit is a reasonable estimation of preventive and / or mitigation measures. This assumption is not necessarily correct. Mitigation may not be possible in all cases, for example, in cases where actual mitigation cost could be an underestimation of true environmental cost. On the contrary, mitigation measure might not be cost-effective and these costs might be an over-estimation of environmental costs. A distinction needs to be made between:

- The costs of measures already adopted, which are theoretically already included in financial cost category. These costs should be reported as a distinct financial cost category. Counting them as environmental costs would be double counting.

- The costs of measures that need to be taken to prevent environmental damages up to a certain point, such as the Directives' Objectives. These costs can be a good estimate of what society is willing to forego.

c) *Revealed preference methods* (used as a technique for the valuation of environmental costs and benefits): The underlying assumption is that the value of goods in a market reflects a set of environmental costs and benefits and that it is possible to isolate the value of the relevant environmental values. These methods include recreational demand methods, hedonic pricing models and averting behavior models:

- Hedonic Pricing: Hedonic pricing methods explain variation in price (in the price of goods) using information on "qualitative and quantitative" attributes. They are used in the context of water to value how environmental attributes and changes affect property prices. In addition to structural features of the property, determinant of property prices may include proximity to, for example, a river or lake. The change in property price corresponding to an environmental degradation, for example the pollution of a river or lake is the cost of this degradation.

- Averting Behavior: this method derives from observations of how people change defensive behavior – adapt coping mechanisms – in response to changes in environmental quality. Defensive behavior can be defined as measures taken to reduce the risk of suffering environmental damages and actions taken to mitigate the impact of environmental damages. The costs of mitigating the impact may entail expenditure on medical care needed as a consequence of drinking poor quality water. The expenditure produces a value of the risk associated with the environmental damage.

- Recreation Demand Models: Improvements or deterioration in the water quality may enhance or reduce recreation opportunities, for example swimming, in one or more sites in a region. However, markets rarely measure the value of these changes. RDM can be used on the choices of trips or visits to sites for recreational purposes and the level of satisfaction, time and money spent in relation to the activity. By assuming that the consumer spends time and money as if he was purchasing access to the goods, for example a river stretch, patterns of travel to particular sites can be used to analyze how an individual values the site and, for example, the water quality of the river stretch. Reductions in trips to a river due to deterioration of water quality and associated changes in expenditures reveal the cost of this deterioration.

d) *Stated preference methods* (used as a technique for the valuation of environmental costs and benefits): These methods are based on measures of willingness to pay through directly eliciting consumer preference on either hypothetical or experimental market. For hypothetical market, data are drawn from surveys presenting a hypothetical scenario to the respondents. The respondents make a hypothetical choice, which is used to derive consumer preferences and value. Methods include contingent valuation and contingent ranking. It is also possible to construct experimental market where money changes hand, e. g. using simulated market models. In the questionnaire, it is possible to ask respondents how much they would pay for avoiding an environmental cost or how much they value a given environmental benefit.

e) *Contingent Valuation*: Contingent Valuation is based on survey results. A scenario including the good that would be delivered and how it would be paid for (e.g. through an increase of the water bill) is presented to the respondent. Respondents are asked for their willingness to pay (WTP) for the specified good. The mean willingness to pay is calculated to give an estimated value of the good. One of the difficulties with this approach lies in ensuring that respondents adequately understand the environmental change that is being valued.

f) *Use of Value Transfer* (alternative option to direct valuation of environmental costs or benefits - more commonly known as benefit transfer in the case of benefits): This method uses information on environmental costs or benefits from existing studies and uses this information for the analysis in the river basin under consideration. As a result, a data set that has been developed for a unique purpose is being used in an application for a different purpose, i.e. it transfers values from a study site to a policy site, i.e. from the site where the study has been conducted to the site where the results are used. Above all, benefit transfer is suitable when technical, financial or time resources are scarce. However, among other problems, it is important to note that since benefits have been estimated in a different context they are unlikely to be as accurate a primary research. A step-wise approach should be developed in order to ensure that the transfer of values derived in other contexts can minimize the potential for estimation errors.

3.5 Cost-Effectiveness Analysis (CEA)

Cost-Effectiveness Analysis (CEA) is a technique for comparing the relative value of various strategies. In its most common form, a new strategy is compared with current practice (the "low-cost alternative") in the calculation of the cost-effectiveness ratio (CE ratio):

$$CE\,ratio = \frac{Cost_{New\,Strategy} - Cost_{Current\,Practice}}{Effect_{New\,Strategy} - Effect_{Current\,Practice}}$$

The result might be considered as the "price" of the additional outcome purchased by switching from current practice to the new strategy (e.g., 10,000 € per life year). If the price is low enough, the new strategy is considered "cost-effective."

3.6 Benefit-cost ratio = BCR

$$BCR = \frac{\left(\sum_{t=0}^{T} \frac{B_t}{(1+d)^t}\right)}{\left(\sum_{t=0}^{T} \frac{C_t}{(1+d)^t}\right)}$$

Where :

- ⇨ the planning period begins in the current year, t = 0, and extends to some future planning horizon T (in years) ;
- ⇨ B = total benefit in the subcripted year (in Euros) ;
- ⇨ C = total cost in the subcripted year (in Euros) ;
- ⇨ d = discount rate expressed in decimal form

A better indicator of the cost benefit balance is given by the following ratio (BCBalance)

$$BCBalance = \frac{\left(\sum_{t=0}^{T} \frac{B_t}{(1+d)^t}\right) - \left(\sum_{t=0}^{T} \frac{C_t}{(1+d)^t}\right)}{\left(\sum_{t=0}^{T} \frac{C_t}{(1+d)^t}\right)}$$

Where :

- ⇨ the planning period begins in the current year, t = 0, and extends to some future planning horizon T (in years) ;
- ⇨ B = total benefit in the subcripted year (in Euros) ;
- ⇨ C = total cost in the subcripted year (in Euros) ;
- ⇨ d = discount rate expressed in decimal form

3.7 Critic issues of the economical theory assumptions

3.7.1 Difficulties with opportunity cost evaluation (Bernard BARRAQUE – 2002)

At the root of all the approaches relating to marginal opportunity cost, economists have implicitly made many assumptions which are considered as questionable by water suppliers and other members of the water policy community. For instance, the notion of user cost is based on the fact that water used for potable uses is "forgone" for future uses. But this may means that they consider water as a mineral like oil, and not as a renewable resource. It is justified if utilities decide to tap fossil aquifers where water resource are largely trapped and non-renewable. But as concerns surface water or shallow aquifers, they will be renewed every year in a variable manner depending on climatology, and, as long as overexploitation does not lead to land subsidence or other irreversible effects, future users' cost can be maintained at a rather low level if waste water is collected and treated, thus making water available again downstream; besides it will be very difficult to calculate a user cost anyway, since the future availability of water to share between users is variable (droughts, wet years), while usually capacity expansion is an irreversible decision, made once, but having very long term consequences in terms of debt service.

Another critique is that it is much more important to calculate the Marginal User Cost of irrigation, which in Mediterranean areas often takes the lion's share, and is evaporated or consumed by the plants for the most part. Strikingly, irrigation systems are often at best hardly covering operation costs, and forget about depreciation of the investment. They are obviously miles away from full cost pricing, and yet economists tend to focus more on potable water services full cost pricing, without realising that at their full cost valuation and pricing, water services should be able to buy as much water as they need from irrigation.

A second assumption made by economists is that water demand is responsive to price changes. Yet, even in the United States, most studies on demand elasticity to prices remain not conclusive, and even show very little price elasticity. More intriguing is that elasticity is not much higher in the US than in Europe while prices are in average two to three times smaller, and consumption two to three times larger. Yet according to the typical demand curve, elasticity should be much higher in the US. One probable explanation is that water supply has always been under-tariffed because the external (public health) benefits were so great that it had to be so. And tariffs would be so low that domestic water abstractions would be made unconsciously, in a routinely if not careless manner. But the social question is whether water supplies should raise water prices just for the sake of seeing elasticity appear?

Another difficulty is due to the unique character of water services, with more than 80% of the private or internal cost made up by the investment, and the rest only by operations and maintenance. Besides, depreciation of the heaviest investment (water mains and sewer pipes) should be made on a very long period of time (more than 50 years), beyond the horizon of bankers and their interest rates. There is then a great temptation to "sink" the investment, and to consider that marginal private cost is just operation and maintenance. In any case, marginal cost is usually always inferior to average cost, and conversely to the assumption of economists it cannot equate it. This then reinforces the tendency for a water supplier to try to sell more water all the time, until he has reached full capacity of existing system. Then suddenly water prices increase tremendously, which makes any capacity expansion very risky, since elasticity models are not made to evaluate the effects of changes which are all but marginal. If the supplier includes depreciation of the initial investment and debt service in the price, what happens is that in case of a negative demand evolution, he will have to increase his prices to still be able to cover the fixed costs. And that may well be found unacceptable by domestic water users who would have been induced to conserve water.

This kind of criticism has been made also by D. Brookshire et al. (2001), who have reviewed the literature of attempts to estimate present and past urban water demands in the arid West of the U.S. as a function of prices. They first conclude that present pricing, even after scarcity of the resources has been acknowledged, are far from reflecting this scarcity. They also find out that this literature is of little help to imagine consumer response to prices which would do so: extrapolation well beyond the observed prices range is quite difficult, because then user cost is in fact linked to future consumption patterns which depend on future prices. In order to limit the uncertainty on MUC, on has to make precise assumptions on population and residential growth, and on the evolution of other, non residential demands. However, projections have always been made by planners who had global urban expansion rather than implied investment in mind; which usually led to over-invest in systems expansion, and begging for subsidies. But let us quote the conclusion of the authors: "Empirical demand studies of residential and urban water uses in the arid Southwest appear inadequate for current and future policy analysis for a number of reasons. First, reported water charges in these analyses are generally based on cost recovery in municipal water supply systems, which fail to reflect the true scarcity of water. A second, but unique, problem is a direct outcome of the cost recovery pricing methodology. In particular, water charges are, in many cases, so low (less than a tenth of a cent per gallon - 0.30 Euros/m3), that it is unclear whether consumers have the appropriate information with which to make informed decisions concerning water use. This calls into question both the reliability and robustness of elasticity estimates for extrapolating reported demand curves. In fact, it raises questions as to the reliability of the demand estimates themselves. Related to and complicating this phenomenon is the fact that in the studies reported, water is considered as an end use commodity, Yet, as observed above, there is reason to believe that in many cases water is simply an input in a production process.

"For all these reasons, there is substantial doubt as to the reliability and applicability of extant demand estimates *outside of observed price ranges* as well as substantial doubt as to the efficacy of estimated demand elasticities in extrapolating existing relationships. Moreover, analyses must incorporate the fact that as water charges increase, not only will the quantity of water demanded change, but the patterns of water use will change as well. This means that empirical studies must also allow for structural change in demand functions. Finally, there is the issue of user cost, whose value must be ascertained in order to establish the scarcity value of water. While costs are usually thought of as supply or production related, to the contrary, user cost, the discounted present value of the best foregone use, is clearly demand-based; i.e. the value of this best foregone use is determined by the level and nature of future demand (…) This apparent conundrum is just the usual simultaneity that attends most economic decision-making. The problem is intellectually trivial for the theorist, but informationally challenging for the policist (manager/policy-maker)…"

Last remark: despite its underpricing in many countries (and particularly in the US), potable water is still sold at a very high price compared to other uses. Therefore it will have the possibility to compensate largely other users of water if they take their share. In particular irrigation, which conversely has a rather low value, but abstracts and above all consumes much more, which should be sanctioned by much higher prices.

3.7.2 Critics of methods for the estimation of environmental costs

Economic valuation, at the conceptual level, is said to be a measure of the preferences people hold for different states of the environment. Valuation, as an empirical exercise, rests on the argument that choices individuals makes in markets exchange provide the data that analysts can use to translate people's preference into money terms (SHABMAN Leonard and STEPHENSON Kurt – 2000).

The logic of the argument is straightforward. In market exchange money income is sacrificed (a price is paid) in order to secure a good or service. By arguing that preferences guide market choices, analyst conclude that the money value of a good or service is at least equal to the amount of income a person spends to obtain the good or service. Thus market prices are the raw data for preference measurement. The often-explicit premises of this revealed choice framework are that individuals know their preference for goods and services (states of the world) before being confronted with choice, that people are willing to pay to satisfy those preferences, and whatever an individual chooses is in the interest of that individual (RANDAL and PETERSON – 1984). It is the benefit-cost analysts' responsibility to measure those preferences in money terms (RANDAL – 1999).

Non all economists support the expanded use of nonmarket valuation calculation in policy. These critics are supported by concerns about nonmarket valuation expressed by psychologist, philosophers, and political scientists who are familiar with the valuation research program. In general, the critics question one, or both, of two core assumptions:

- that choices made in real or hypothetical market can be interpreted as a reflection of preferences or value;

- that such interpretations should direct decision making (SHABMAN Leonard and STEPHENSON Kurt – 2000).

Free Market Environmentalist Critique

Free Market Environmentalist have a particular understanding of the social purpose of market exchange. In this conception of market exchange people do not bring their preferences to the choices they make, but come to know their preference when faced with particular choice opportunities. Then even in making these choices, people have a limited capacity to observe, process, and make use of all available information. Decisions are made with significant ignorance – completely overlooked or unknown opportunities. Therefore, market exchange is process for coming to know, discover and revise preferences. At the same time entrepreneurs act to offer new preference-changing choice opportunities by discovering new technology and resources (HAYAK – 1948). The function of market prices is to stimulate change, coordinate decentralized adaptation to that change, synthesize disperse and fragmented knowledge and promote individual incentives, discretion, and responsibilities.

In the Free Market Environmentalist economist's view, market prices cannot be examined to discover a fixed set of preference, because not only preferences but other determinant of price are always in flux making (SHABMAN Leonard and STEPHENSON Kurt – 2000).

The work of psychologists provides support for the Free Market Environmentalist view that preferences are dictated by choice circumstances and are context- and time – sensitive. Psychological research has found that in many choices situations people do not retrieve preferences from previously formed preferences, but preferences are constructed at the time and in the context of a choice opportunity (SLOVIC et al. – 1977; TVERSKY et al. – 1988; GREGORY et al. – 1993; SCHKADE and PAYNE – 1994; SCHADE 1995)

The Free Market Environmentalist economist compares nonmarket valuation with central economics planning in the former Soviet bloc countries. For example, the inability of soviet planners to calculate prices that would determine how many bushel of wheat to produce or how many nails to produce is not different than a government analyst attempting to determine water quality standards, the allocation of water between municipalities and agriculture, and how many acres of timbers of harvest (ANDERSON and LEAL – 1991; SMITH – 1995) to engage in valuation is to pretend without justification that the preference-revealing and discovering process of markets can be replicated. The Free Market Environmentalist economists direct their attention away from valuation and toward the establishment of market and market like processes for value discovery.

Institutional Economics Critique

> *"To believe that market determines value is to believe that milk comes from plastic bottles"* (BROMLEY – 1985).

Institutional economist working in the tradition of John R. COMMONS and Thorstein VEBLEN note that benefit-cost analysis begins with the assumptions that preferences, resource endowment (income), and technological opportunities are fixed and not subject to inquiry or questioning. Prices that emerge in market processes are a function of these conditions. For institutional economist, however, the conditions establishing market exchange should themselves be subject to social debate, scrutiny, and policy change (BROMLEY – 1989). One concern is that choices and prices reflect income (market power and economic opportunities) as much as preferences and value, and distributional issues are a legitimate social concern. A second concern is that decisions are made within a context that shape people's preferences (HODGESON – 1988; VATN and BROMLEY – 1994) and that current preferences may reflect outdated social habits and ignorance (HODGESON – 1998).

With this perspective, institutional economists argue that nonmarket valuation inappropriately elevates the preferences of current individuals and those with the greatest income (ability to pay) to the touchstone for environmental decision making (JACOBS – 1994). They note that the provision of most environmental services is undertaken in the political arena and it is within this context that people form and express values about environmental services. Noting that environmental issues are dominated by a moral dimension and expressed in political processes, they argue that nonmarket valuation expects people to take issues out of moral or social context and places them in an exchange (willingness to pay) context (VATN and BROMLEY – 1994). In fact, the large number of protest bids often reported in the Contingent Valuation Methods studies provides empirical evidence that willingness to pay is not the way some people think about the environment.

The institutional economists argue that preferences are malleable and should be subject to ongoing scrutiny, and not treated as datum for governing public decisions. Eschewing valuation, institutional economists advocate analyses that preferences a subject of investigation and social debate (SHABMAN Leonard and STEPHENSON Kurt – 2000).

> *"Proponent and critics of environmental valuation are reflecting a more general debate over the appropriate place of analysis and analysts in the making of any public policy. The proponent of benefit-cost analysis (and valuation) suggest that they are providing neutral information that can mitigate any unacceptable influence of special interest in the choice process. The critics fear that analysts themselves, and the "questionable" information they produce, may gain unwaranted influence over resource allocations more properly left to market or to democratic political choice. In making their arguments, critics of environmental policy. Because their concerns are one example of a wider and long-standing debate on the role of quantification and analysis in public policy, the debate*

over the value of valuation will net be resolved." (SHABMAN Leonard and STEPHENSON Kurt – 2000).

Full-cost recovery and consideration for inclusion of water cost to price

⇨ **What is full cost pricing?**

For neo-classical economics, the full cost of a water service is not just the cost of operations and maintenance plus depreciation of initial investments. This is average cost. Full cost pricing must also reflect the opportunity costs of water system resources, and even equal the marginal opportunity cost (CARTER and MILON – 1999; BARRAQUE - 2000).

⇨ **What can we suggest? – A proposition for a strategy relating to inclusion of water cost to price (BARRAQUE – 2002)**

The economic part of the WaterStrategyMan decision support system cannot just stay with traditional average cost pricing, since it leads to wrong investments (as illustrated by the model case of Barcelona). However, calculating the Marginal Opportunity Cost, even in medium term, is very difficult, and impossible directly. Moreover, we have to introduce a new criterion if we are to reach sustainability: a sustainable water policy must integrate economic and environmental aspects, as illustrated above, but also keep an ethical dimension, i.e. remain socially and politically acceptable. This third dimension is the least well studied, and yet it is crucial. In particular, it is quite probable that in most Mediterranean areas under study, prices of water services are far from reflecting average costs, since they do not include repayment of investments. In turn, the quality of the services is not very good, and then domestic users are tempted to keep individual water sources like private wells and cisterns. Doing so they fragilise even more the Potable Water Service, because their demand is only a secondary demand. In order to break this vicious circle and to bring water services to a better standard of continuity and "constance" (Zerah, 1997), there is a need to give a lot more information to water users than is usually the case now, so as to reduce transaction costs, and also leaks …Basically, it results from the above that the first need is to have a strong and representative water supplier, with the legitimacy to bargain with its customers and with other water users. Water being scarce, some form of metering has to be introduced, to follow the patterns of consumption and to be able to track the leaks quicker.

The first step of the economic analysis must be to calculate the present average cost and the present rate of cost recovery from water bills. The calculation should be made including sewage collection and treatment in conformity with the Urban Waste Water Directive (EC 271/91), i.e. with either centralised system (and in this case water costs should more or less double if there is no more precise estimate), or with decentralised systems like septic tanks (for which a yearly average cost can be obtained locally). Calculation should also clearly include figures for the seasonal population, because tourism will usually be important in the area. The analysis must be supplemented by a description of the state of the technical system, its need for investment to reach its own sustainability (replacing ageing parts). It should be possible to calculate the rate of leaks, and the cost of water conserved by various schemes of leaks control.

Then one has to turn toward forecast of potable water demand. This must be done much more carefully than in the supply side tradition. For any given population growth, or tourism or irrigation development, on can calculate what will be the yearly and seasonal water need, all things equal, and compare with the existing data on resources yearly available, and also in decennal drought. If there is a deficit, one has to interrogate the development scheme, or rather try to solve the equation: how much of this development can be sustained if domestic or tourist water demands go down through conservation schemes? If it is enough, next step is to develop a strategy to bring the prices and recovery of bills closer to the costs.

If it is not enough, but only in dry years, this will be a case for trying to have contracts with local farmers who irrigate: what is their opportunity cost if they cannot irrigate as much as they wished, and incur a loss in crops ? This loss should be compensated, and on top of this price, should be calculated the cost of "wheeling" the water to the PWS, through a transfer system.

Note that it is the same for qualities as for quantities: one can always calculate the loss incurred by farmers who abandon fertilisers and pesticides for the sake of protecting groundwater for future potabilisation, and build up a compensation scheme. In many instances studied in Germany and in France, the compensation is of the order of 150 Euros/ha.

But there is a constraint, which is easier to describe than to calculate: if water markets between PWS and irrigators generalise, there is a real risk of disappearance of agriculture, which in the Mediterranean can mean erosion and other environmental impacts due to desertification. Then there is a need to study how could agriculture adapt and partly revert to dryland or no-intrants farming without disappearing, and what would be the best compensation scheme. Another possibility is to irrigate with treated effluents, at least in areas downstream of cities of with no abstraction areas.

If there is not enough water to accommodate urban and tourist development, but this time in a structural manner, this is a case for extra-capacity investment, and of course water transfers from longer distances. But before deciding this extra-capacity investment, again conservation strategies and non-conventional water supplies must be considered and compared in at least their own average costs, and at best their MOC. In particular, as we have already pointed, sea-, or better, brackish-water desalination, depending on what is available and how far, will serve as the basis for benchmarking and compare the various schemes.

In any case, it seems clear that if there is not enough water to implement a development plan based on both urban and irrigation growth, the second should be sacrificed (and irrigation areas maintained as they are) if the plan implies to build long distance water transfers: never could irrigation pay for the MOC of these new supplies, and they can only have water if a system which has been designed for urban uses does not reach its full operational capacity. There thus should be a learning process to bring irrigators to accept that their water use has low priority and should follow urban use.

The following diagram illustrates the different propositions that have been made in this chapter

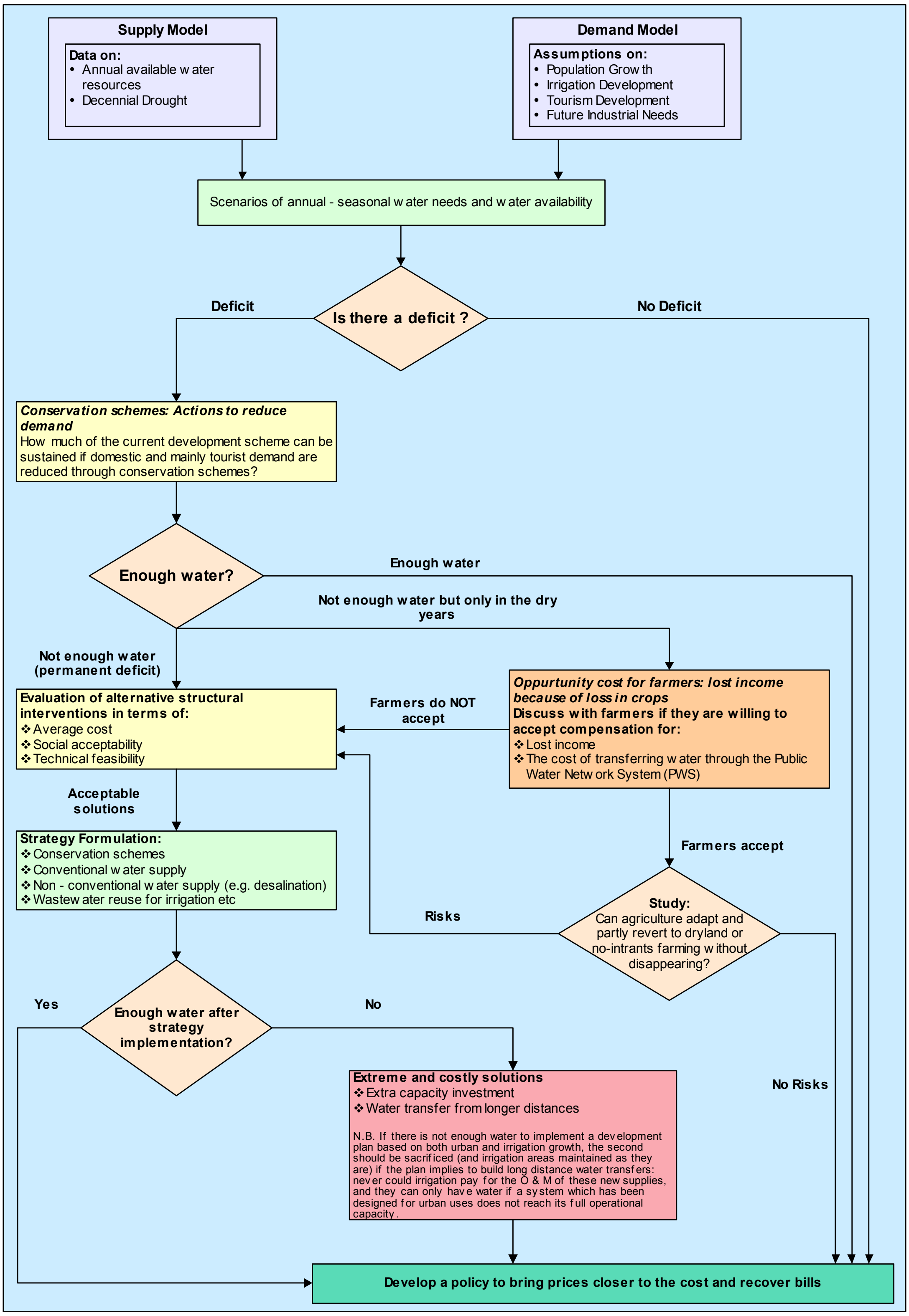


Figure 6. Building Strategies on the relationship of potable water demand and economics in the DSS

3.8 Recommendations: proposed indicators for the DSS

3.8.1 Indicators 1 : Total present value of the direct costs

$$TVPC = \sum_{t=0}^{T} \frac{P.Q_t}{(1+r)^t}$$

where :

- ⇨ the planning period begins in the current year, t = 0, and extends to some future planning horizon T (in years) ;
- ⇨ P = the price per unit of supplied water charged by the water company (in Euros) ;
- ⇨ Q_t = annual quantity of supplied water (in m^3) ;
- ⇨ r = the annual **real** interest rate relevant for the investor.

3.8.2 Indicators 2 : Cost of sustainability of technical systems = C_{STS}

$$C_{sts} = \sum_{ij} \left\{ C_{i,j} \cdot \left[\frac{(CostPV_i)}{t_i} \right] + CostAO_i + CostAM_i \right\} + \sum_{kj} \left\{ TL_{k,j} \cdot \left[\frac{(CostPV_k)}{t_k} \right] + CostAO_k + CostAM_k \right\}$$

Where *i*, *k*, *j* the parameters that are presented in Table 8

Table 8: Parts of infrastructure and different water users

Plant *i*	**Total network length *k***	**Users *j***
Dams	Water distribution net	Permanent population
Water catchment	Sewer network	Seasonal population
Water treatment plant	Irrigation network	Irrigation
Distribution systems for freshwater treatment		Industry
Waste water treatment plant		Power generation
Distribution systems for waste water treatment		

and:

C_i:	Capacity of i (in m^3)
$CostPV_i$:	Present value of i (in Euros)
$CostAO_i$:	Average Operating Costs of i (in Euros)
$CostAM_i$:	Average Maintenance Costs of i (in Euros)
t_i:	Depreciation period (useful life) of i (in years)
TL_k:	Total length of network k (in m)
$CostPV_k$:	Present value of network k (m^3)
$CostAO_k$:	Average Operating Costs of network k (in Euros)
$CostAM_k$:	Average Maintenance Costs network k (in Euros)
t_k:	Depreciation period (useful life) of network k (in years)

3.8.3 Indicators 3 : Rate of subsidisation of water service = R_{sub}

$$R_{sub} = \left(\frac{\text{Cost supported by water services} - \text{Revenues from water billing}}{\text{Cost supported by water services}} \right) \cdot 100$$

Or

$$R_{sub} = \left(\frac{\text{Subsidies}}{\text{Cost supported by water services}} \right) \cdot 100$$

Nota Benne:

- Costs supported by water services ≤ Cost of sustainability of technical systems;
- Costs supported by water services = Revenues from water billing + subsidies;
- Cost of sustainability of technical systems – Costs supported by water services = Sum that missing to reach the sustainability of the technical systems

3.8.4 Indicator 4 : valuation of environmental costs

We proposed to use *cost-based valuation methods* for the valuation of environmental costs.

These methods present different limits that end user of the DSS should know:

1) This method is based on the assumption that the cost to maintain an environmental benefit is a reasonable estimation of preventive and / or mitigation measures.
2) This assumption is not necessarily correct. Mitigation may not be possible in all cases, for example, in cases where current mitigation cost could be an underestimation of true

environmental cost. On the opposite, mitigation measure might not be cost-effective and these costs might be an over-estimation of environmental costs.

3) A distinction needs to be made between:

- The costs of measures already adopted, which are theoretically already included in financial cost category. These costs should be reported as a distinct financial cost category. Counting them as environmental costs would be double counting.
- The costs of measures that need to be taken to prevent environmental damages up to a certain point, such as the Directives' Objectives. These costs can be a good estimate of what society is willing to forego.

The following equation should be used.

C_{Env} = costs of preventive and / or mitigation measures ≅ environmental costs

$$C_{Env} = \sum_{xy}\left\{C_{x,y}\cdot\left[\frac{(CostPV_x)}{t_x}\right]+CostAO_x+CostAM_x\right\}+\sum_{zy}\left\{TL_{z,y}\cdot\left[\frac{(CostPV_z)}{t_z}\right]+CostAO_z+CostAM_z\right\}$$

Where x, y, z the parameters that are presented in the following table.

Parts of infrastructure needed to maintain the environmental benefit of keeping the same quantity of water available per capita (for each user) with a quality in compliance with legislation

Plant x needed to maintain the environmental benefit of keeping the same quantity of water available per capita (for each user) with a quality in compliance with legislation	**Total network length *z* needed to maintain the environmental benefit of keeping the same quantity of water available per capita (for each user) with a quality in compliance with legislation**	**Users *y***
Dams	Water distribution net	Future permanent population (based on a scenario) – present permanent population
Water catchment	Sewer network	Future seasonal population (based on a scenario) – present seasonal population
Water treatment plant	Irrigation network	Future needs for irrigation (based on a scenario) – present needs for irrigation
Distribution systems for freshwater treatment		Future needs for industry (based on a scenario) – present needs for industry
Waste water treatment plant		Future needs for power generation (based on a scenario) – present needs for power generation
Distribution systems for waste water treatment		

and:

C_x:	Capacity of x (in m^3)
$CostPV_x$:	Present value of x (in Euros)
$CostAO_x$:	Average Operating Costs of x (in Euros)
$CostAM_x$:	Average Maintenance Costs of x (in Euros)
t_x:	Depreciation period (useful life) of x (in years)
TL_y:	Total length of network y (in m)
$CostPV_y$:	Present value of network y (m^3)
$CostAO_y$:	Average Operating Costs of network y (in Euros)
$CostAM_y$:	Average Maintenance Costs network y (in Euros)
t_y:	Depreciation period (useful life) of network y (in years)

Nota: If there is local data providing, by a scientific survey, willingness to pay for the conservation of the resource, theses data should be used as an indicator of the environmental costs.

3.8.5 Indicator 5 : Opportunity cost estimation

To find the best approach to quantify opportunity costs, we should try to build a model following example described in the chapter 2.2.5.

The research to determine this approach to quantify opportunity costs must go on, but it seems to be very difficult to incorporate all aspects of opportunity costs in one simple indicator.

If we do not succeed in the construction of a model for the quantification of opportunity costs, a very simple and rustic method should be proposed. It can be asked to the end user of the Decision Support System to fill in the following table to qualify the level of the scarcity rent.

		The inconvenience for following uses will be…				
		Permanent population	Tourism	Irrigation	Industry	Power generation
It has been decided to allocated X m^3 to the following user	Permanent population		☐ very high ☐ high ☐ average ☐ low ☐ very low	☐ very high ☐ high ☐ average ☐ low ☐ very low	☐ very high ☐ high ☐ average ☐ low ☐ very low	☐ very high ☐ high ☐ average ☐ low ☐ very low
	Tourism	☐ very high ☐ high ☐ average ☐ low ☐ very low		☐ very high ☐ high ☐ average ☐ low ☐ very low	☐ very high ☐ high ☐ average ☐ low ☐ very low	☐ very high ☐ high ☐ average ☐ low ☐ very low
	Irrigation	☐ very high ☐ high ☐ average ☐ low ☐ very low	☐ very high ☐ high ☐ average ☐ low ☐ very low		☐ very high ☐ high ☐ average ☐ low ☐ very low	☐ very high ☐ high ☐ average ☐ low ☐ very low
	Industry	☐ very high ☐ high ☐ average ☐ low ☐ very low	☐ very high ☐ high ☐ average ☐ low ☐ very low	☐ very high ☐ high ☐ average ☐ low ☐ very low		☐ very high ☐ high ☐ average ☐ low ☐ very low
	Power generation	☐ very high ☐ high ☐ average ☐ low ☐ very low	☐ very high ☐ high ☐ average ☐ low ☐ very low	☐ very high ☐ high ☐ average ☐ low ☐ very low	☐ very high ☐ high ☐ average ☐ low ☐ very low	

3.8.6 Indicator 6 : Rate of recovery for the Total Cost = RCR$_{tc}$

$$RCR_{tc} = \left(\frac{Revenues\ from\ water\ billing}{C_{CSTS} + C_{Env}} \right) \cdot 100$$

Where :

$$\mathrm{Re}venues\ from\ water\ billing = \sum_j \left(Billed\ Volume_j \cdot \mathrm{Pr}ice_j \right)$$

C_{CSTS} and C_{env} => see below indicators 2 and 5

RCR_{tc} can be estimated also through:

$$RCR_{tc} = \left(\frac{\mathrm{TR} - Subsidy}{TC} \right) \cdot 100$$

Where:

TR: total revenues of water service (depending on the cost recovery mechanism - this figure could be based on either fixed or variable charges in € / years)

Subsidy: the total amount of subsidies paid to the water service

TC: the total economic costs (in € / year) of water service provided (C_{CSTS} + C_{Env})

Nota benne : Because of the difficulties for finding a simple indicator relating to non accounting opportunity costs, the rate of recovery for the total cost does not take them into account.

3.8.7 Indicator 7 : Benefit-cost Balance = BCBalance

$$BCBalance = \frac{\left(\sum_{t=0}^{T} \frac{B_t}{(1+d)^t}\right) - \left(\sum_{t=0}^{T} \frac{C_t}{(1+d)^t}\right)}{\left(\sum_{t=0}^{T} \frac{C_t}{(1+d)^t}\right)}$$

Where :

- ⇨ the planning period begins in the current year, t = 0, and extends to some future planning horizon T (in years) ;
- ⇨ *B* = total benefit in the subcripted year (in Euros) ;
- ⇨ *C* = total cost in the subcripted year (in Euros) ;
- ⇨ *d* = discount rate expressed in decimal form

3.8.8 Methodology for the allocation of prices

User	Water current price per cubic meter for the user (Fill in as a data in the DSS)	Billed volume of water for different uses (Fill in as a data in the DSS)	Current revenues from water billing for the different uses (calculated by the DSS)	Current part of the total cost recover by the user (calculated by the DSS)	Projected part of the total cost recover by the user (the end user of the DSS propose a scenario to find a more equitable repartition)	Projected price with a constant billed volume (calculated by the DSS as a consequence of the scenario defined by the end user of the DSS)
Permanent population	A Euros	BVpp = Billed volume for permanent population	BVpp x A	$\left(\frac{BVpp \times A}{C_{CSTS} + C_{Env}}\right) \times 100$	AA %	$A_{pr} = \frac{AA \times (C_{CSTS} + C_{Env})}{100 \times BVpp}$
Tourism	B Euros	BVt = Billed volume for tourism	BVt x B	$\left(\frac{BVt \times B}{C_{CSTS} + C_{Env}}\right) \times 100$	BB %	$B_{pr} = \frac{BB \times (C_{CSTS} + C_{Env})}{100 \times BVt}$
Irrigation	C Euros	BVirr = Billed volume for irrigation	BVirr x C	$\left(\frac{BVirr \times C}{C_{CSTS} + C_{Env}}\right) \times 100$	CC %	$C_{pr} = \frac{CC \times (C_{CSTS} + C_{Env})}{100 \times BVirr}$
Industry	D Euros	BVind = Billed volume for industry	BVind x D	$\left(\frac{BVind \times D}{C_{CSTS} + C_{Env}}\right) \times 100$	DD %	$D_{pr} = \frac{DD \times (C_{CSTS} + C_{Env})}{100 \times BVind}$
Power generation	E Euros	BVpg = Billed volume for power generation	BVpg x E	$\left(\frac{BVpg \times E}{C_{CSTS} + C_{Env}}\right) \times 100$	EE %	$E_{pr} = \frac{EE \times (C_{CSTS} + C_{Env})}{100 \times BVpg}$
Total	Not relevant	Not relevant	Total	RCR_{tc}	Total = RCR_{tc} projected	Not relevant

Nota Benne 1: The projected price is with a constant billed volume => this approach does not take into account the price elasticity of demand.
Nota benne 2: Because of the difficulties for finding a simple indicator relating to non accounting opportunity costs, the current part of the total cost recover by the user does not take them into account.

Legend

—— fill in as a data in the DSS
—— calculated by the DSS
—— the end user of the DSS propose a scenario to find a more equitable repartition

4 Section III: MCDM Approaches

4.1 Introduction and Terminology

Traditionally, making decisions in business and engineering was based on a single objective to be optimised. Decisions in water resources management in particular are typically characterised by a large set of alternatives and multiple, conflicting and incommensurate evaluation criteria.

The need of considering multiple objectives when making decisions has been widely recognised in the last decades.

A class of operations research (OR) techniques that are used frequently for this type of problem are multi criteria decision making (MCDM) tools. Multi Criteria Decision Making techniques provide powerful tools for engineers who are faced with increasingly complex decisions and conflicting objectives.

Hipel (1982) states the following five benefits of modelling techniques in MCDM:

- They furnish a logical structure in which the problem can be organised and displayed
- They constitute a common language for discussing the problem with experts and laypeople,
- they help improving the communication within society
- They take into account multiple objectives of a project
- They allow for an extensive sensitivity analysis to ascertain the consequences of meaningful parameter changes upon the optimal solution

This report presents some of the Multi Criteria Decision Making techniques that could be used as the basis for assessing water resources systems within the Decision Support System (DSS).

A MCDM problem can be described using a decision matrix characterised by *m* alternatives, each of them being assessed using *n* attributes. Thus, the decision matrix is a *m x n* matrix with each element being the *j*-th attribute value of the *i*-th alternative. A simplified framework for MCDM is depicted in Figure 11.

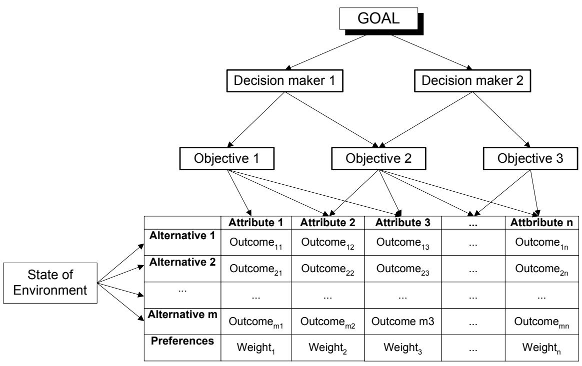


Figure 11: Framework for MCDM (Malczewski, 1999)

MCDM problems can be characterised by the following features:

- Conflicting objectives
- Hybrid nature of attributes
  Attributes typically have different units of measurement. The yield of a reservoir may be assessed by volume of water but the esthetical value of the dam can only be described in linguistic terms.
- Uncertainty
  Uncertainty in MCDM problems may be due to uncertainty in subjective judgements by the people involved as well as due to missing or incomplete data and/or information of some attributes.

First, the goals of a certain project or action have to be defined. These objectives are defined by decision variables that are established in the course of decision making process. Decision variables are denoted x_k, k=1,…K. A particular alternative is a set of decision variables and the achievement of objectives is measured using objective functions $Z_j(x)$

Mathematically, the multiobjective decision making problem can be expressed by a p-dimensional vector of objective functions

$$z(x) = \left[z_1(x), z_2(x), ..., z_p(x) \right]$$

that is to be maximised subject to constraints

$$g_i(x) \leq 0 \qquad i = 1, 2, ..., m$$

The decision variables $x = (x_1, x_2, ..., x_n) \in R^n$.

The *feasible region* is denoted X as defined as follows:

$X = \{ x : x \in R^n, g_i(x) \leq 0, x_j \geq 0 \}$ for all i and j. As a single optimal solution does not exists, MCDM techniques seek for a set of ***non-dominated*** solutions *S* which are a subset of the feasible region. Non-dominated solutions are characterised by the fact that for each solution outside the set of non-dominated solutions (but within the feasible region) there is one non-dominated solution for which all objective functions are unchanged or better and at least one objective function is improved. Formally, the non-dominated solutions can be expressed by

$$z_q(x') > z_q(x) \qquad \text{for some } q \in \{1, 2, ..., p\}$$

and $z_k(x') \geq z_k(x)$

Non-dominated solutions are often refereed to as ***pareto optimal*** solutions ***or efficient solutions***.

A ***superior solution*** (***ideal solution***) is a solution that maximises all of the objectives at the same time. Formally, a solution is superior if and only if

$z(x^{'}) > z(x)$ for all i.

Because of the conflicting nature of many objectives it is obvious that such a solution will hardly exist in water resources management. A most preferred or best-compromise solution is a non-dominated solution that is finally chosen by the decision maker based on his preference structure. A best-compromise solution chosen by one decision maker will most likely be different from a best compromise solution chosen by another decision maker.

A number of approaches have been suggested to classify the various MCDM techniques. The classification that is used here is based on the timing of the articulation of the preferences by the decision maker and the optimisation of his preference structure relative to one another. Consequently, MCDM approaches can be subdivided into the following three classes:

- Methods based on the prior articulation of preferences
- Methods based on the progressive articulation of preferences
- Methods based on the posterior articulation of preferences

4.2 Weighting of indicators

MCDM problems typically involve a number of criteria that are not equally important to the DM. Consequently, one important step in MCDM is the articulation of weights to the criteria that reflect the DM's preference structure with regard to the objectives.

Each attribute *j* is assigned a weight that represents the preference structure of the decision maker (DM). The weights typically sum up to one.

Formally, a set of weights is defined as follows:

$w = (w_1, w_2, ... w_j, ... w_n)$ and $\sum w_j = 1$.

The weight values assigned to the criteria account for two factors:

- Changes in the range of variation for each evaluation criteria
- Different degrees of importance being attached to these ranges of variation

There exist a number of methods for calculating the values of the normalised weight value based on the information given by the DM. The most commonly used approaches are briefly described below:

Ranking methods

The simplest way of assigning a numerical weight value to any of the objectives is to rank the objectives in order. The most commonly approach for assigning weight values is the *rank sum* method, in which each criterion is weighted and the normalised by the sum of all weights. Formally,

$$w_j = \frac{n - r_j + 1}{\sum (n - r_k + 1)}$$

where w_j is the normalised weight value for the j-th criterion, n is the number of criteria to be considered, and r_j is the rank position of the criterion.

Alternatively, the weight can be derived from the normalised reciprocals of a criterion's rank. The following formula is used to compute *rank reciprocal* weights:

$$w_j = \frac{1/r_j}{\sum 1/r_k}$$

Rating methods

Rating methods are based on the DM's estimation of weights on a predefined scale. If the simplest rating method, the *point allocation approach* is used, the DM will express his preference structure for the attributes on a predefined scale of, say 0 to 1 or 0 to 100. A weight value of zero indicates that the criterion can be ignored and a value of 100 represents a situation where only one criterion is considered.

4.3 Transformation of indicators

In order to compare and to aggregate indicators, it is necessary to transform and normalise them respectively. There exists a number of different transformation functions for a variety of different indicators, the most commonly applied way is to determine a desirable and least acceptable (best and worst) values and to normalise the measured value between the two threshold values linearly, so that

$$I_{ij} = \begin{cases} 0 & if \;\; z_{ij} < w_j \\ \dfrac{z_{ij} - w_j}{b_j - w_j} & if \;\; w_j \le z_{ij} \le b_j \\ 1 & if \;\; z_{ij} > b_j \end{cases} \qquad \text{and } z_{ij} = \begin{cases} b_j \;\; if \;\; z_{ij} > b_j \\ w_j \;\; if \;\; z_{ij} < w_j \end{cases}$$

where I_{ij} is the degree of achievement of objective j in alternative i, z_{ij} is an indicator value of objective j in alternative i, b_j and w_j denote the best and worst values of the indicator for objective j. This type of transformation function is depicted Figure 12 (last diagram).

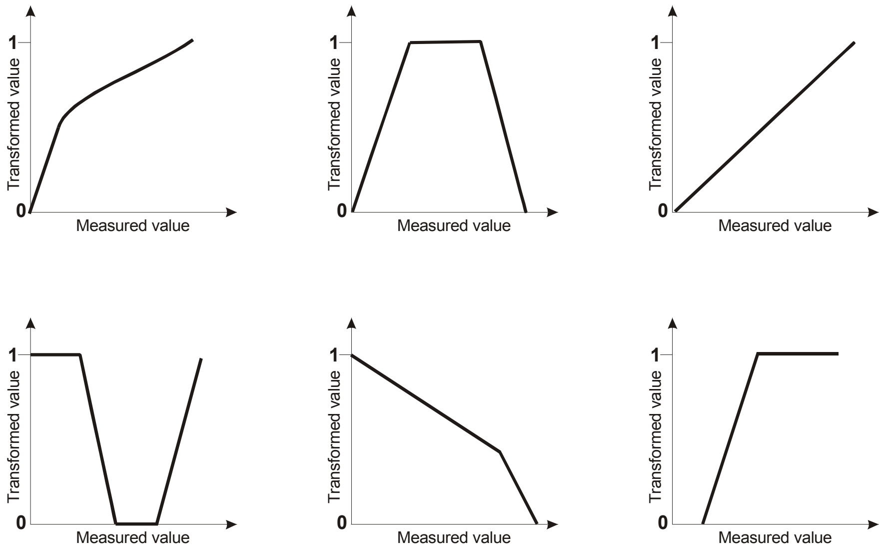


Figure 12: Typical transformation functions for indicators

The type of transformation function depends on the indicator under consideration and the preferences of the decision maker.

The first diagram in the figure above shows some non-linear positive relationship between the measured and transformed value. The transformation depicted in diagram two may be applicable for situations in which a measured value above a given threshold is worsening the situation and is not desired (see diagram four for a negative normalisation).

4.4 MCDM Techniques

4.4.1 Based On The Prior Articulation Of Preferences

Introduction

Methods in this category based on the prior articulation of the preference structure with regard to the objectives. The advantage of this group of MCDM techniques is that the process of assigning preferences to the different objectives may help the DM in understanding the problem better. On the other hand, the process of determing the preference structure may be difficult and is often time-consuming. Most of the approaches are particularly suitable for situations where the outcome is know with certainty.

Scoring Methods

Scoring methods are one of the simplest and the most frequently used methods in multi-criteria decision making. Basically, these methods consist of three steps. First, the DM assigns weights to each of the attributes (e.g. on a scale of 1 to 10 or 1 to 100). Then, the a numerical value on a similar scale is assigned to the attributes that determines the degree of performance of each alternative. The worth of an alternative *j* is computed by a the following weighting sum:

$$v_j = \sum_{i=1}^{n} \alpha_i n_{ij}$$

Obviously, the alternative with the highest value of v_j is the best option.

Scoring methods permit tradeoffs between different criteria. That is a bad performance of one alternative in one attribute can be compensated by an enhancement in one or more others attributes, which in many cases cannot be accepted by the decision maker. The use of a linear weighting sum to compute the values of alternatives, however, has very little theoretical foundation.

The Analytic Hierarchy Process

The Analytic Hierarchy Process (AHP) is a widely used MCDM technique that has been developed by Saaty in the mid 70s. For the underlying theory, see [5].

It can be best classified as a scoring method that allows the consideration of both, objective and subjective factors in the decision making process. The approach has been implemented on a very popular software package called ExpertChoice. (http://www.expertchoice.com).

AHP is based on the three principles decomposition, comparative judgements and synthesis of priorities. The method assumes that it is in general easier for a DM to compare two alternatives than to compare more than two.

The decomposition principle breaks down the MCDM problem into a hierarchy in which the higher elements compromise the higher goals and objectives and the lower elements represent the attributes. The lowest elements in the hierarchy are the alternatives. Once the problem has been structured in that way, the relative importance of each of the elements has to be determined through a pairwise comparison ("How important is alternative A compared to alternative B") of the elements with respect to the element above. The result is a number of matrices for each of the alternatives. The local priorities (i.e. priorities on the criteria, subcriteria and alternatives are determined by computing the normalised principal right eigenvectors of the comparison matrices.

The main advantage of the AHP approach is that it provides a measure for the consistency in the DM's judgement. With the aid of consistency ratios that are determined with the decision making process, it is possible to measure the consistency of the DM in his preference structure.

Although the methodology is widely used, particularly in business applications, it has not been without criticism.

The main criticism is the "rank reversal problem", that refers to the reversal of the preference order when a new option is introduced in the process. Another major criticism refers to the pairwise comparison of alternatives which implicitly assumes that the DM is clear about how much of criterion A is compared to how much of criterion B which is not true in general.

With regard to the application in the DSS within the WSM project, the main flaw remains the lack of methodological transparency. The computation of eigenvectors in AHP is not very easy to perform and irreproducible for the DM.

Multiattribute utility theory (MAUT).

Multiattribute utility theory is a methodology that is aimed at selecting the best option from a number of alternatives in situations where the decision outcomes are not known with certainty. Unlike the MCDM methods discussed so far, in MAUT approaches, the probability density function over the attribute space is defined instead of an exact value indicating the outcome of an alternative. The conceptual basis for utility theory and the axioms the DM has to conform to will not be discussed in detail here. A comprehensive description the theory is given in Goicoechea et. al, 1982. The axioms imply that preferences of alternatives are defined in terms of expected utilities *u(x)*. The multiattribute function is usually decomposed into m singe-attribute functions which are constructed through interviews with the DM. As the outcome of a decision is uncertain, the DM is presented with lotteries to quantify his utility over a given alternative. The assessment of an utility function requires the assessment of m component utility functions which are determined by the risk attitude of the DM.

Although the method is capable of dealing with uncertainties and has a strong theoretical foundation, there are a number of difficulties related to that type of MCDM techniques. First, the assessment of the utility function requires information which is difficult to provide and secondly, the underlying assumptions of the method (i.e. preferential independence and utility independence) are not always easy to ensure.

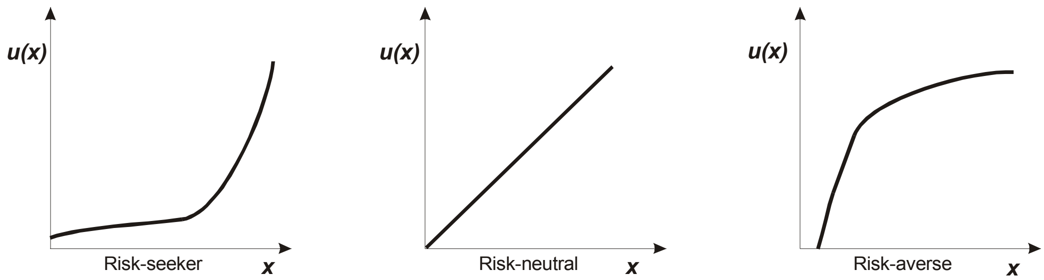


Figure 13:Basic shapes of utility functions

Outranking methods

Methods that provide an ordinal outranking of alternatives but cannot indicate how much one alternative is preferred to another.

ELECTRE I

ELECTRE I (Elimination et choix traduisant la realité) is a multicriterion algorithm that reduces the number of non-dominated solutions by comparing two alternatives as a whole. It is particularly suitable for MCDM problems with a discrete number of alternatives and can be classified as an outranking method as it provides and ordinal ranking of the alternatives. The algorithm is based on the idea to select an alternative that is preferred for most of the criteria but does not cause unacceptable level of discontent for the other criteria.

The pairwise comparison of the alternatives is based on concordance, discordance and threshold values.

The concordance matrix of two alternatives *i* and *j* is a weighted measure of the number of criteria for which alternative *i* is preferred (or equal) to alternative *j*:

$$c_{ij} = \frac{W^{+} + \frac{1}{2}W^{=}}{W^{+} + W^{=} + W^{-}}$$

Where

W^+ = sum of the weights for which *i* is preferred to *j*,

W^- = sum of the weights for which *j* is preferred to *i*,

$W^=$ = sum of the weights for which *i* and *j* are equally preferred

The discordance matrix expresses the maximum interval difference between alternative *i* to alternative *j*:

$$d_{ij} = \max_{k, n_{ik} < n_{jk}} \left(n_{jk} - n_{ik} \right)$$

The smaller the value of d_{ij}, the less bad is the comparison of *i* with *j*.

To calculate the outranking relationship between the alternatives *i* and *j*, the DM has to define threshold values *p* and *q*, both in the range 0 to 1. The concordance threshold *p* specifies how much concordance the DM wants; a value of 1 corresponds to full concordance; alternative *i* should be preferred to alternative *j* in all criteria. The discordance threshold q indicates the amount of discordance the DM is willing to accept; for *q=0*, the DM does not accept any discordance.

The outranking relation between the two alternatives is determined by combining the concordance and discordance matrices; alternative i dominated alternative j if and only if:

$c_{ij} \geq p$ and

$d_{ij} \leq q$

It is clear that by choosing certain combination of p and q and solution may not be feasible with the given alternatives. In this case, the threshold values have to be adjusted and the method has be applied again.

ELECTRE II

ELECTRE II is an extension of the ELECTRE I and has been developed by Roy (1971). Whereas ELECTRE I provides a partial ordering of the alternatives, ELECTRE II offers a complete reordering of the non-dominated set of alternatives. It is based on the same assumptions as the ELECTRE I but uses multiple levels of discordance and concordance to construct two extreme outranking relationships; a strong relationship R_s and a weak relationship R_w. The calculation of the elements c_{ij} of the concordance matrix differs from the calculation in ELECTRE II:

$$c_{ij} = \frac{W^+ + W^=}{W^+ + W^= + W^-}$$

The discordance matrix has the same definition as in ELECTRE I. The strong relationship is defined if and only if one or both of the following conditions hold:

$$c_{ij} > p^*; \;\; D_{i,j} < q^* \;\; and \;\; W^+ > W^-$$

$$c_{ij} > p^0; \;\; D_{i,j} < q^0 \;\; and \;\; W^+ > W^-$$

The weak relationship is defined if and only if the following conditions holds:

$c_{ij} > p^-;\ \ D_{i,j} < q^* \ \ and \ \ W^+ > W^-$

The result of these relationships are two graphs; one for strong and one for weak relationships which are used for ranking the alternatives in the next step. A complete description of the approach and an illustrative example are given in Goicoechea et al 1982.

Goal Programming

Goal programming is based on the assignment of predefined target values to each objective function by the DM. The optimal solution of the problem is then defined as the one the that minimises the sum of the deviations from the target values. The method can be formally described as follows:

$$\min \sum_{i=1}^{p} \left| F_i(x) - T_i \right|$$

where T_i denotes the target value of the objective function $F_i(x)$. The criterion to minimise is the sum of differences between target value and objective function value. The objective function is non-linear, so that the simplex method can only be applied if the function is transferred into a linear form. This transformation is done by introducing new slack variables d_i^+ and d_i^- so that

$$d_i^+ = \tfrac{1}{2}\left\{ \left| F_i(x) - T_i \right| + \left[F_i(x) - T_i \right] \right\}$$

$$d_i^- = \tfrac{1}{2}\left\{ \left| F_i(x) - T_i \right| - \left[F_i(x) - T_i \right] \right\}$$

d_i^+ is the positive deviation from the predefined target values (overachievement), d_i^- is the negative deviation from the target value (underachievement) and adding both equations yields:

$$d_i^+ + d_i^- = \left| F_i(x) - T_i \right|$$

Both, d_i^- and d_i^+ have to be non-negative and, since it is not possible to have underachievement and overachievement of one goal at the same time, the product has to be zero which is automatically fulfilled by the simplex-method. The non-linear optimisation problem can therefore be formulated as

$$\min W_0 = \sum_{i=1}^{p} \left(d_i^+ + d_i^- \right)$$

subject to

$$x \in X$$

$$F_i(x) - d_i^+ + d_i^- = T_i$$

$$d_i^+, d_i^- \geq 0 \quad , i = 1, \ldots, p$$

which can be solved using a simplex method.

The DM may wish to assign weights that express his preference with regard to overachievement or underachievement of the respective objective functions. In addition, he rank the goals according to his preference structure. In this case, the goal programming model can be written as

$$\min S_0 = \sum_{i=1}^{p} P_i \left(w_i^+ d_i^+ + w_i^- d_i^- \right)$$

subject to

$$x \in X$$
$$F_i\left(x\right) - d_i^+ + d_i^- = T_i$$
$$d_i^+, d_i^- \geq 0 \quad , i = 1, \ldots, p$$

4.4.2 MCDM Based On The Progressive Articulation Of Preferences

Introduction

Techniques based on the progressive articulation are characterized by an iterative process that involves the DM. First, a subset of the non-dominated solutions is identified and the DM is asked to provide his preference structure for these alternatives. The problem is then modified accordingly and the two steps are repeated until the DM accepts one best compromise solution.

Compromise Programming (CP)

Compromise programming is an interactive method that identifies non-dominated solutions which are closest to the ideal solution by some distance measure.

The underlying idea of compromise programming can be easily explained for a simple case where only two objectives are to be achieved. The degree of achievement of objective Z_1 is displayed on the y-axis and the degree of achievement of objective Z_2 is displayed on the x-axis. The indicators are transformed using the convenient definition that zero denotes the least acceptable value (no achievement) and one represents full achievement of the objective.

The ideal point of optimal achievement is obviously the upper right corner with the coordinates (1,1) (Figure 14). The degree of meeting both objectives d can be calculated by the distance between the ideal point and the points of achievement for a given alternative. :

$$d_i = \left[\sum_{j=1}^{2} \left(1 - n_{ij}\right)^2 \right]^{\frac{1}{2}}$$

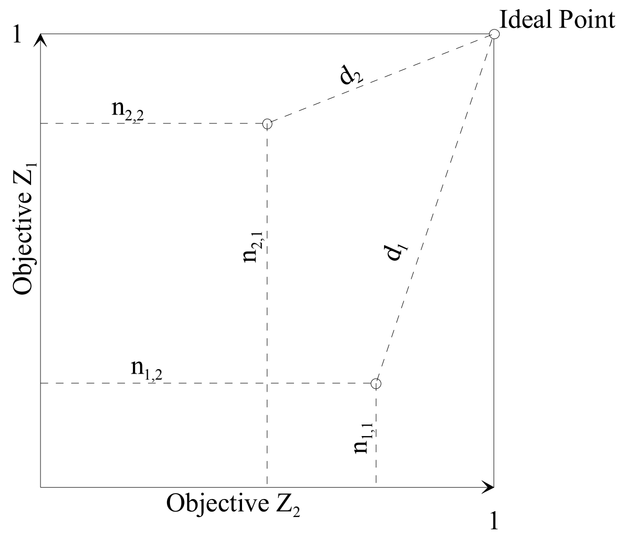


Figure 14: 2-D geometrical interpretation of distance-based methods

for the two-dimensional case (s . By introducing a compensation factor p and the weights α for each alternative, the distance from the ideal point in an i-dimensional space is computed using

$$d_i = \left[\sum \alpha_i \left(1 - n_{ij}\right)^p \right]^{\frac{1}{p}}$$

The parameter p reflects the DM's concern with respect to the maximum deviation and determines how a poor achievement of one objective can be compensated with a good performance in another. For p=1, the Hamming distance is calculated and all deviations are weighted equally (i.e. a perfect compensation). For p=2, the Euclidean distance penalises large deviations from the ideal point. The larger p, the larger is the weight for the largest deviation. For the Chebychev distance ($p=\infty$), there is no compensation between criteria. The assessment depends on the largest deviation from the ideal point. The sensitivity of the power factor is depicted in Figure 15.

The weight α_i reflects the DM preference or relative importance of the *ith* objective. Usually, only three points of the comparison set are computed, p=1,2 and ∞. The alternative with the minimum distance to the ideal point with respect to p is selected as the compromise solution.

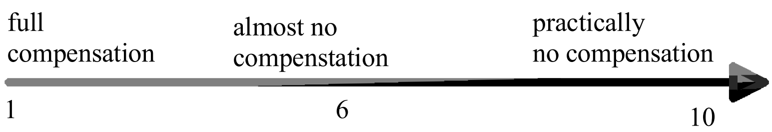


Figure 15: Sensitivity of the power factor p

Composite Programming (CTP)

Composite Programming has first been introduced by Bárdossy et. al.(1985) as an extension of compromise programming and can be described as employing a hierarchical methodology to compromise programming. Based on the factual relationship of objectives and the ability of compensation of objectives, the objectives are grouped. The groups, in turn are arranged in hierarchical order on cascading levels. Each group is assigned a compensation factor p and a weight α that determines the relative importance of the group of objectives.

Feasible Goals Method (FGM)

Feasible Goals Method was developed by Lotov et. al. (1997) and can be classified as a goal-oriented selection of decision alternatives based on computer graphics. The method has been implemented in a software package called Visual Market. It basically presents interactive decision maps (IDM) that present the DM the efficiency of a given criterion, depending on another. The underlying theory is mathematically sophisticated and will not be presented here.

In this approach, the decision maker is given the opportunity to explore graphically presented criterion performance values and proxy tradeoffs among them. The method provides for fast display of proxy criterion tradeoffs concerning a large number of alternatives. Proxy tradeoffs are displayed on stacked charts, which can also be animated. This information helps to identify a *reasonable goal* – an acceptable tradeoff value among the decision criteria, which is close to feasible criterion performance values.

An extension of the method, the Reasonable Goals Methods (RGM) is particulary helpful when a large number of alternatives are analysed visually.

Since the results of the optimisation are presented to the DM in an iterative way, it is possible to assess the effect of a certain change in the alternatives has on the overall performance. The underlying theory however is not easy to explain to laymen.

4.4.3 MCDM Based on the Posterior Articulation of Preferences

These methods are the least commonly used of the MCDM techniques. In general these methods generate the set of non-dominated solutions which are then presented to the DM to select the preferred one. The main problem with this approach is that the choice of the preferred alternative is not always easy for the DM and may be time-consuming and cumbersome. An example for this class of MCDM techniques is the Data Envelopment Analysis (DEA).

Data Envelopment Analysis (DEA)

Data Enveloppment Analysis was developed by Charnes, Cooper and Rhodes in 1978 and is based on three main elements:

- Inputs (where less is better), e.g. assets in production
- Outputs (where more is better), e.g. performance indicators
- Decision making units; an entity for which measurable inputs can be assigned (alternatives)

The approach is based on the assumption that an increase in an input is expected to yield an increase in an output and that it is desirable to minimise inputs as they result in costs.

The algorithm used to provide a solution of MCDM problems using the DEA is a pair of dual linear programming models. The output is a so-called envelopment surface (sometimes referred to as production function) which allows the DM to determine which DMU is efficient and which is not. Furthermore, the can identify the sources of inefficiency of an alternative and provides a list of alternatives that can be used for comparison.

The DEA is not always classified as a method based on the posterior articulation of preferences but has been classified here because it is used to compute the efficiency of alternatives.

4.5 Uncertainties

A number of uncertainties are inherent in the decision making process. These can be classified as follows:

- Uncertainties in expert opinions
- Uncertainties in decision making
- Uncertainties associated with the DM

The approaches present so far (with MAUT being the only exception) are based on the assumption that the DM can express his preferences over the criteria precisely. Clearly, this is not true in general and the entire decision making process has a number of uncertainties that can be classified as follows:

Uncertainties in expert opinion refers to the inherent uncertainties when estimating the impacts of a given set of action on a water resources system. The environmental, social, esthetical and other consequences of a given water management intervention cannot be predicted with certainty. MAUT being the only method that can cope with uncertainties.

Uncertainties in the decision making process refer to the individual and societal consequences . Finally, uncertainties that are related to the decision maker can have a large influence on the selection of alternatives; situations might occur in which the DM is not able or unwilling to determine the relative importance of the evaluation criteria. His unwillingness may be due to imprecise information and/or knowledge. In addition , it inconsistencies in the DM's choice can be found .

There are several ways to overcome this problem, some of them will be briefly discussed below.

Sensitivity analysis

One way to deal with uncertainties is to use a sensitivity analysis that is aimed at investigating the sensitivity of the objectives. Typically, the criterion outcome is computed for number of weights and the range of possible variation of the weights is determined. The objective of a sensitivity analysis is to find out how the output of the MCDM procedure (i.e. the recommendation of an alternative) is affected by the DM's preference.

Alternatively, a certain problem can be solved using an average weighting and the result can be compared with the one reflecting the preferences initially assigned. In many cases, a degree of confidence for each criterion is specified for a given preference value.

Fuzzy approaches

Another approach that is widely applied in situations where one is confronted with uncertainties is to use fuzzy approaches. Many of the above described methodologies have been extended using fuzzy sets. These include fuzzy compromise programming (Bender and Simonovic, 2000) as well as fuzzy compromise programming (Bárdossy and Duckstein, in Hipel, 1982). and a fuzzy extension of AHP.0

Fuzzy approaches have proved to be very useful in water resources planning and has been implemented in a number of decision support systems. The main advantage of those approaches is that more realism is added to the process since many criteria in water resources planning are fuzzy by their very nature. In the same way, the criteria weights as well as the DM's interpretation of the degree of compensation between criteria which all together warrants scepticism when traditional MCDM techniques are used.

4.6 Summary and Recommendations

The final choice of what MCDM to use for a given problem is not always easy and straightforward, so that this problem itself could actually be classified as an MCDM problem. There are several factors to be considered when selecting an MCDM technique (Mollaghasemi, 1997):

- Characteristics of the decision making problem
- Characteristics of the DM
- Characteristics of the solution technique

Given specific task for which MCDM approaches will be applied in the context of the WaterStrategyMan (WSM) project, it is recommended that the selection of a MCDM technique should be based on the following criteria:

- Comparability of alternatives
- Methodological Transparency
- Mathematical Sophistication
- Interactivity for preference structure
- Not stakeholder specific
- Involvement of the DM in the decision making process

Consequently, the method to be selected should be an interactive method based on the progressive articulation of preferences, although methods based on the posterior articulation may be applicable if they meet the above recommendations.

The advantages of methods based on the progressive articulation of preferences can be summarised as follows:

- A better understanding of the problem is achieved through involvement of the DM
- The outcome may be more easily accepted
- Less restrictive assumptions are required.

5 Section IV: Indicator Approaches

5.1 Introduction

The importance of reasonable indicators to assess the impacts of natural systems has been widely recognised in recent years.

The improvement of data collection and development of indicators is strongly recommended in chapter 40 ("information for decision making-bridging the data gap") of Agenda 21.

Chapter 40.4 of the agenda, reads *Commonly used indicators such as the gross national product (GNP) and measurements of individual resource or pollution flows do not provide adequate indications of sustainability. Methods for assessing interactions between different sectoral environmental, demographic, social and developmental parameters are not sufficiently developed or applied. Indicators of sustainable development need to be developed to provide solid bases for decision-making at all levels and to contribute to a self-regulating sustainability of integrated environment and development systems.*

Accordingly, a number of international organisations such as the United Nations (UN), the European Union (EU), the European Environment Agency (EEA), the World Bank and others have recently defined or are currently defining indicators to "measure" sustainable development. This report gives an overview of the methodologies to compute indicators for the assessment of water resources systems and presents some indicators that could be used in the framework of the WaterStrategyMan (WSM) project.

The list of indicators presented is neither comprehensive nor final but should be understood as a first draft of core indicators to be used within the project.

The following sections of this deliverable contains a brief summary of existing indicator approaches with a special focus on water-related indicators. In the first part, some basic definitions and criteria for selecting indicators are given. Next, commonly used indicator approaches by international organisations are described.

The appendix contains a list of candidate indicators that could form the basis for assessing water management interventions with regard to the objectives stated.

5.2 Basic definitions and notations

An *indicator* is generally understood as a value that describes a condition. An attribute is a measurable inherent characteristics of a set of actions. In a broader sense, OECD defines an indicator as "a parameter, or a value derived from parameters, which points to, provides information about, describes the state of a phenomenon/ environment/ area, with a significance extending beyond that directly associated with a parameter value." Indicators are typically tracked over time. Indicators can be aggregations of different data or can be composed of complex characteristics. The classical example for an indicator is the Gross Domestic Product (GDP).

OECD points out two major functions of environmental indicators; first, they reduce the number of measurements and parameters are required to give and "exact" representation of the situation and secondly, they simplify the communication process by which the results of a measurement are provided to the user.

An *index* is a combination (a mathematical aggregation) of two or more indicators.

Defining an index of a set of indicators is not always an easy task because it involves assigning weights to diverse parameters which depends of course on the user's preference. The aggregation procedure itself can be linear or on-linear, additive, multiplicative etc. and it is clear that the index may vary largely depending on the selected approach.

One or more indicators can partly describe an attribute; the indicator BOD partly describes the attribute water quality in rivers.

Indicators are selected with a goal or objective in mind and thus they describe the value of a system and the bettering or worsening of the conditions over time. The information derived from indicators can therefore be used to develop appropriate actions.

With regard to the assessment of water resources systems in the context of the European Water Framework Directive (WFD), the indicators must describe the three broadest objectives for achieving sustainability, i.e. environmental integrity, economic efficiency and equity (Young, 1992).

5.3 Criteria for selecting environmental indicators

Before actually defining indicators for the purpose of evaluating different scenarios with regard to the criteria to be defined, it is necessary to define general criteria for selecting indicators.

OECD classifies the criteria for an ideal indicator into the three main criteria policy relevance and utility for users, analytical soundness and measurability. Policy relevance requires that an indicator should provide a representative picture of the conditions, pressures and responses of the environment and that it should be able to show trends over time in relation to the change it is intended to represent.

It further requires that an indicator must be comparable on an international scale and have a threshold or reference value so that users can simply assess the significance of the values associated with it.

Analytical soundness of an indicator involves its technical and scientifically sound foundation and international consensus about its validity. Furthermore, the data for the indicators should be readily available on the given scale and should be updated on a regular interval in accordance with reliable procedures.

Despite the very wide range of issues that have to be asses using indicators it tends to be more effective to have a small set of well-chosen indicators rather than a large number of interrelated indicators. The World Bank stipulates the following criteria for appropriate environmental indicators:

- Direct relevance to project objectives
- Limitation in number

- Clarity of design
- Realistic collection or development costs
- clear cause and effect links
- High quality and reliability
- Appropriate spatial and temporal scale
- Targets and baselines
- Little or no interrelation

Clearly, the major constraint to meet the above criteria for indicators is the availability of data.

5.4 Limitations to Indicators

Most of the indicators used by international organisations such as OECD, World Bank and others are conceived for a geographical scale that corresponds to the national or country level. Some of the indicators however are applicable on the regional or catchment scale that is used in the WaterStrategyMan project. The best spatial scale for an indicator is one at which the indicators shows least stochastic variations and little variations if the spatial scale is slightly changed.

The indicator water availability per capita if measured on a national scale is not able to depict important water shortages in some parts of the country.

The appropriate time scale for the calculation of an indicators value is of utmost importance; if the time scale is chosen too long, the variability of the objective under consideration will be lost; the supply with drinking water in a region may be sufficient if annual average values are considered, but may be unacceptable if based on monthly values.

Although the interest for water-related indicators is rapidly growing, one must be aware of some important problems with indices.

The most important limitation of water-related indicators is due to data availability and quality. This can be caused by inaccurate regional resolution, gaps in data on water availability, difficulties in measuring water use data and a lot of other reasons.

Another problem is related to the multidisciplinary and multifaceted nature of water issues. It is often attractive to aggregate different indicators or measure into one single index. This single index however can be misleading and uninformative as the indicators can describe different spatial scales.

Care must be taken to clearly define the different measures from which an indicator is defined.

The indicator "Access to safe drinking water and sanitation systems" is widely used by international organisation such FAO; World Bank, WHO etc. Gleick (2002) notes that the definitions of the terms "access, clean water and sanitation services" have changed remarkably since the measure was first used in the seventies. "Access" for instance is defined as "Water source at a distance from home at … km" but there is no agreement on how many kilometres this distance can be.

It goes without saying that as soon as these definition change it is extremely difficult to get a clear picture of the entire system and its changing conditions over time.

Whatever indicator approach is used one must recognise the fact that an indicator only describes part of the complex field of water resources management. The interests are often conflicting and an index can only measure on aspect of the whole system.

5.5 Structuring of indicators

There exist a number of different approaches for structuring indicators in a way that the structure reflects indicators describing the condition of a system and indicators describing the response of the system to a given condition. The most commonly used methodologies are briefly described below.

5.5.1 The P-S-R approach

A widely used approach to structure indicators is Pressure-State-Response (P-S-R) approach that was first introduced by OECD in 1994 and can be applied at the national, sectoral, community, or individual firm level.

It is based on the assumption that human activities exert a *pressure* on the environment and thereby affect the quality and quantity of the natural resources (its *state*). The pressure, in turn, cause a *response* of the society that can be through environmental, economic and sectoral policies. Pressures cover both direct and indirect pressures. Direct pressures exert from the use of a resource or a discharge of pollutants, whereas indirect pressures result from the activity itself or from trends of environmental significance. The construction of a new port has direct impacts by displacing natural areas and may have indirect impacts by increased traffic and hence pollution.

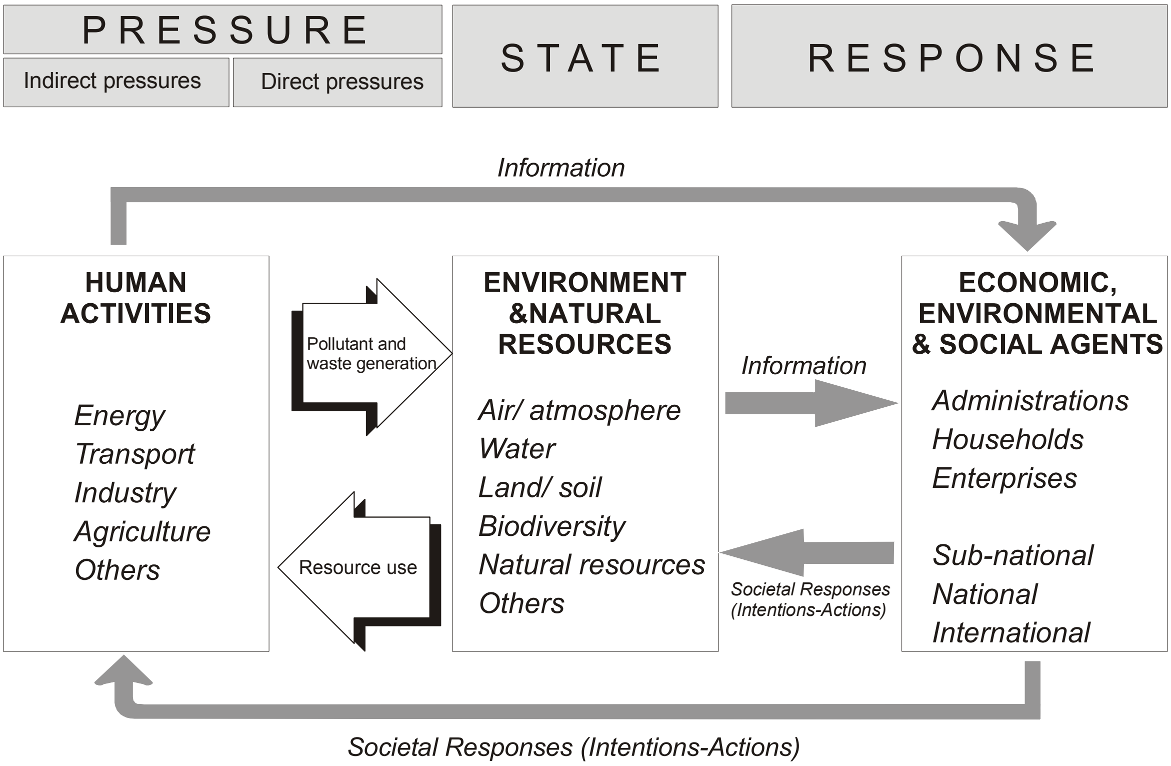


Figure 16: Pressure- State- Response (P-S-R) model (OECD)

The original concept of the P-S-R approach has experienced some modifications and adjustments; examples are the Driving force-State-Response (DSR) model that was formerly used by UNCSD or the Driving force-Pressure-State-Impact-Response (DPSIR) model that is used the European Environment Agency (EEA).

The main advantage of the PSR model is that it may help the decision-maker as well as the public to see the interconnections between the various issues on the system under consideration.

Provided the data availability, the major indicators may be disaggregated at secotral level for analysing the pressures exerted by different economic sectors and distinguishing responses from government, private households and the business sector.

5.5.2 The DPSIR model

The Driving force-Pressure-State-Impact-Response (DPISR) model is an extension of the PSR model and was developed in the 70s by Anthony Friend. The approach has been adopted by the EEA.

Drivers can be for example the economic activities in the country and its spatial distribution or the market prices for fuel and transport. Pressure indicators describe the parameters that directly cause environmental problems. Examples are toxic emission, heavy metal pollutants, etc.

Impact indicators describe the ultimate effects of a change of the state. Examples are the number of people affected by polluted drinking water etc.

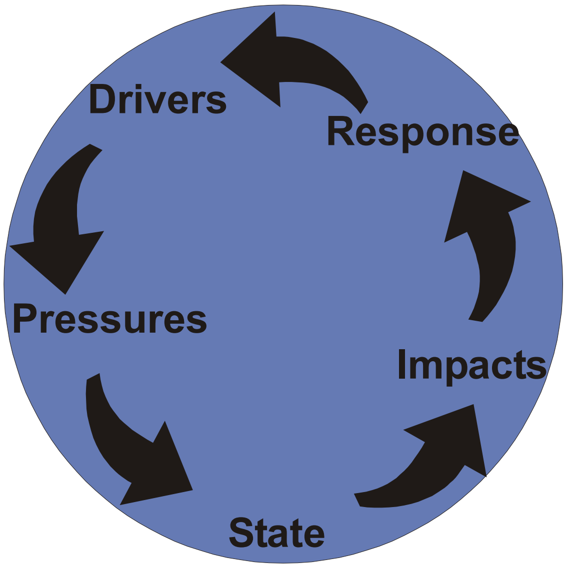


Figure 17: DPSIR model used by the European Environment Agency

5.5.3 Project-based approach

In a project-based approach, indicators are designed to measure both, the long-.term or pervasive results of a project and the immediate, short-term impacts of a given project. The project-based approach of structuring indicators is commonly used for certain projects with anticipated impacts on a well-defined area, where the specification of project-level objectives is feasible. In situations that have a broader perspective, however, the PSR approach is the obvious choice.

5.5.4 Well-being assessment

The International Union for Conservation of Nature and Natural Resources (IUCN) has developed a method for assessing the human and environmental conditions and progress toward sustainable development.

It is based on the assumption that ecosystem surrounds and supports society as much as the white of an egg surrounds the yolk and that society can consequently only be well and sustainable if both the people and the ecosystem are well.

Both, ecosystem well-being and environment well-being are given equal weights and are characterised by five subsectors, each of them characterised by one indicators. Participants agree on performance criteria, representing the indicators and factors such as the estimated sustainability rates or targets.

Indicator scores are combined into element scores which are, in turn, combined into dimension indices, either as unweighted average, weighted average or lower/lowest value.

The dimension indices are given equal weight and are aggregated into a Human Well-Being Index (HWI) for the human dimensions and a Ecosystem Well-Being Index (EWI) for the environmental dimensions.

5.6 Commonly used indicators for water availability

Up to now, there have been several assessments of water resources at different levels (local, national, global) using a number of indicators. The purpose of this chapter is to briefly describe the different indicators and indices that are used to track and compare environmental conditions.

5.6.1 Access To Drinking Water and Sanitation Services

These two are probably the most commonly used index to describe a country's condition with regard to water resources. Their definition goes back to the seventies, but as described above, the definition of access for instance has changed over time so that a direct comparison of countries is not possible in any case.

5.6.2 Falkenmark Water Stress/Water Competition Index

Besides the access to drinking water and sanitation services index, the Falkemark Water Stress Index (or Competition Index) is probably the most widely used index for describing availability of water. Falkenmarks water stress index relates population to the availability of water resources by using Israel as a benchmark for a society's ability to develop in arid regions. He identifies 2000 people as the "maximum number of people that an advanced society is able to support and manage" from a flow unit of 1 Mio m^3. The more people are trying to survive of such a flow unit, the greater the water scarcity.

Later, the index was based on a minimum per capita amount of 100 litres per day that is needed for basic needs. Based on this requirement, the threshold levels in Table 9 which are widely accepted to define water stress are obtained.

Table 9: Water Stress Definitions (Falkenmark)

Annual Renewable Fresh Water (m^3/cap*year)	**Level of water stress**
>1700	Occasional or local water stress
1000-1700	Regular water stress
500-1000	Chronic water scarcity (lack of water begins to hamper economic development and human health and well-being)
<500	Absolute water stress

Although the index is widely used to rank countries with regard to water stress it has some shortcomings. First of all, water availability measure the natural endowment of a county but nothing about how the resource is actually used. A second flaw is that the measure is taken constant over time and does not consider temporal fluctuations of water resources. The indicator may be over 2000m^3/cap* year for a given country but if the country receives all its resources in only one rainy season but might still have serious water stress situations in the dry season.

The index has also been criticised for not including precipitation that directly supports both natural and agricultural vegetation. The latter problem is due to the fact that national estimates on freshwater resources usually do not consider renewable surface water and groundwater flows.

5.6.3 Basic Human Needs Index

Gleick (1990) made an attempt not to measure the availability but to measure some aspects of water use. He defined a basic water requirement (BWR) that is needed per capita of 50 litres a day and then estimates the population that has no access to this BWR.

Limitations of this indicator include data problems at regional level and problems due to the aggregation on country level.

5.6.4 World Development Indicators (WDI)

The world development indicators (WDI) was developed by the World Bank and is updated yearly. It features 600 indicators in the following six sections: Overview, people, environment, economy, states and markets and global links. The indicators are referred to the national and country level respectively, so that they are not directly applicable in the WSM project.

5.6.5 The IWI approach (EPA)

The Index of Watershed Indicators (IWI) was developed by the United States Environmental Protection Agency (EPA) in 1997 and is an overall indicator for the health of aquatic systems in the United States.

It is based on fifteen indicators for characterising the condition and the vulnerability of water resources. Seven of the indicators are related to the condition of water resources systems, eight are related to the vulnerability of the systems.

Table 10: Indicators for the Watershed index (EPA)

Condition indicators	*Vulnerability indicators*
Assessed rivers meeting all designated uses	Aquatic/Wetland species at risk
Fish and Wildlife consumption advisories	Pollutant loads discharged aboe permitted limits-toxic pollutants
Indicators of source water quality for drinking water systems	Pollutant loads discharged above permitted discharge limits- conventional pollutants
Contaminated sediments	Urban runoff potential
ambient water quality data (toxic pollutants	Index of agricultural runoff potential
Ambient water quality (conventional pollutants)	Population change
Wetland loss index	Hydrologic modification- Dams
	Estuarine Pollution Susceptibility index

5.6.6 Vulnerability of Water Systems (Gleick)

This indicator was developed to assess the impacts of a climate change on watersheds in the United States. The index is based on five quantitative measures describing the sensitivity of water resources to water demand, floods, droughts, groundwater exploitation, reliance on hydroelectricity and variability. In detail, the measures are:

- Storage volume relative to renewable supply
  S/Q; With large storage to supply ratio, the short term droughts are less likely to cause a water shortage. Small ratios indicate that floods and droughts can have severe impacts
- Consumptive water use relative to available supply
  This indicators is described above. When the ratio is high, pressure on water resources will be high
- Proportion of hydroelectricity relative to total electricity
  The intention of this indicator is to measure a region's dependence on hydroelectricity which is affected by fluctuations in water availability.
- Groundwater overdraft relative to total groundwater withdrawals
  This indicator measure the ratio of groundwater overdraft (i.e. water pumped in excess of the natural recharge rate) to total groundwater withdrawal. When this ratio is high, water availability is already a problem.
- Streamflow variability
  When a region has a high variability in streamflow, the risk of being affected by floods and

droughts is high. The indicator is defined by The Q_5 flow (flow exceeded in 5 percent of the time) over the Q_{95} flow (the flow exceeded in 95 percent of the time)

5.6.7 Water Resources Vulnerability Index (SEI)

The Water Resources Vulnerability Index (WRVI) has been proposed in 1997 by the Stockholm Environmental Institute (SEI) and is calculated from three sub-indicators, which in turn may be composed of other indicators. The WRVI is calculated by averaging the three sub-indices that are in turn calculated by averaging the indicators belonging to this index (see Figure 18). Each of the indicators is divided into four classes (no stress, low stress, stress and high stress).

A modification of the approach has been made where the WRVI is not computed from average values but from the highest value of any of the sub-indices.

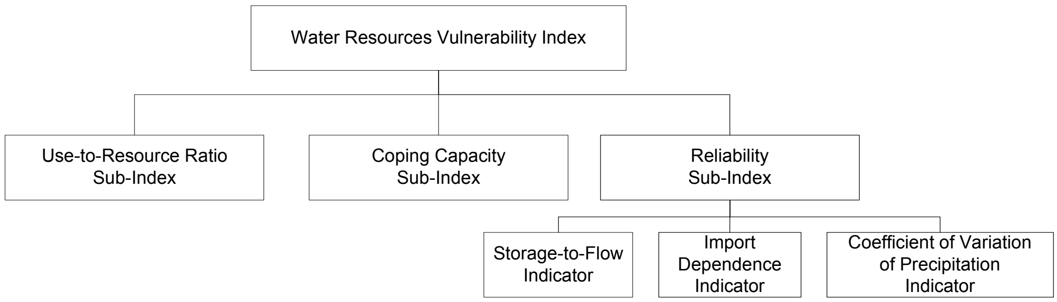


Figure 18: The SEI Water Resources Vulnerability Index (Gleick, 2002)

5.6.8 Relative Water Scarcity (IWMI)

This index was proposed by the International Water Management Institute and describes the water resources of a country in some future-perspective. The Index of Relative Water Scarcity (IRWS) measures (1) how fast a country's water use is growing and (2) how close it is to its total available limit. The indicator is calculated from the percentage increase in water withdrawals over the 1990-2025 period and the projected water withdrawals in 2025 as a percentage of the annual water resources (AWR).

It is somewhat hypothetical as the projections of water withdrawals for a period of more than 20 years are highly uncertain.

5.6.9 CSD Working List of Indicators of SD

In 1996, the commission on Sustainable Development of the United States (CSD) published a working list of indicators on Sustainable Development that are structured according to the Driving Force-State-Response model.

The list follows the chapters of agenda 21 and can be seen a flexible list from which countries can choose indicators according to their priorities and targets. The indicators cover social, economic, environmental and institutional aspects of SD and mostly refer to a national or country level and therefore require some modifications when applied in the framework of the WSM project.

Some environmental indicators from the CSD working list are compiled in the table below.

Table 11: Water-related environmental indicators from the CSD working list of indicators

Category/Chapter	Driving Force	State	Response
Chapter 18: Protection of freshwater resources	Annual withdrawals of ground and surface water Domestic consumption per capita	Groundwater reserves Concentration of faecal coliform in freshwater BOD in water bodies	WWT coverage Density of hydrological networks
Chapter 17: Protection of the oceans, all kinds of seas and coastal areas	Population growth in coastal areas Discharges of oil into coastal water Releases of N and P into coastal waters	Maximum sustained yield for fisheries Algae index	

5.6.10 Plan Bleu

The Mediterranean Commission on Sustainable Development (MSCD) defined a set of 130 indicators for assessing the progress towards Sustainable Development in the Mediterranean countries. The indicators are structured according to the PSR model and cover the following topics:

- Population and society
- Territory and human settlements
- Economic activities and sustainability
- Sustainable development: actors and policies
- Exchanges and co-operation in the Mediterranean

It is the most comprehensive work carried out to assess progress towards sustainable development in the Mediterranean region.

Despite the fact that most of the indicators presented are defined on a national scale and the data is not measurable and/ or meaningless on catchment or region level, the list provides a good basis for a candidate list for indicators within the WSM project.

5.6.11 OECD water related indicators

OECD has developed a set of more than 200 indicators that measure environmental performance and progress towards sustainable development. The indicators are organised by issues including climate change, air pollution, biodiversity, waste and water resources and structured according to the PSR model. The OECD work focuses primarily on indicators to be used on national and international level. The water related core indicators are subdivided into freshwater quality indicators and indicators for water resources and are summarised below.

Table 12: OECD core indicators for freshwater quality

Issue	Indicator	Type
Eutrophication		
	Emissions of N and P in water and soil	P
	N and P from fertiliser and livestock	P
	Nutrient balance	P
	BOD/ DO in inland waters	S
	Concentration of N and P in inland waters	S
	Population connected to secondary and/or tertiary WWTP	R
	User charges for WWT	R
	Market-share of phosphate free detergents	R
Toxic contamination	Emissions of heavy metals	P
	Emissions of organic compounds	P
	Consumption of pesticides	P
	Concentration of heavy metals and organic compounds in env. Media	P
Acidification	Exeedance of critical loads of pH in water	S

Issue	Indicator	Type
Water resources		
	Intensity of use of water resources (abstractions/available resources)	P
	Frequency, duration, extent of water shortages	S
	Water prices and charges for sewage treatment	R

The core set of indicators is supplemented with a set of sectoral indicators such as transport-environment indicators, energy-environment indicators and others.

5.6.12 UNESCO/IHP Sustainability Criteria

The task committee on Sustainability of the American Society of Civil Engineers and Working group M.4.3 of the UNESCO/IHP project jointly presented some approaches to measure sustainability for water resources systems (ASCE, 1998).

Efficiency, Survivability and Sustainability

Pezzey (1992) distinguished between three planning objectives to include sustainability in planning models. These objectives are

- Efficiency,
- Survivability
- Sustainability

The underlying assumption of the approach is that the degree of achievement of the three planning objectives is measured to asses the contribution of the system to sustainability. It is assumed that the net welfare value of any decision made today can be predicted for any time y in the future.

Efficiency

Assume a minimum level of welfare W_{min} is needed for survival. A decision k will be efficient if it maximises the present value of current and all future welfare values for each period y. Considering a discount rate r, the objective function for the welfare is

$$Max \sum_{y} \frac{W(k,y)}{(1+r)^{y}}$$

As the discount rate r is increasing, the values of the future become less and less important for those living today.

Survivability

A decision can be considered survivable if the net welfare W(k,y) is greater or equal than the minimum required for survival, W_{min}

$$W(k,y) \geq W_{\min} \qquad \text{for all periods y}$$

Sustainability

A development is sustainable if it assures that the average welfare of future generations is no less than the average welfare available to previous generations.:

$$W(k,y+1) \geq W(k,y) \qquad \text{for all periods y}$$

In other words, a non-negative change in welfare has to be assured:

$$\frac{dW(k,y)}{dy} \geq 0$$

The duration of the period y have to be chosen in a way that natural fluctuations in water resource are averaged out over the period.

Clearly, the crucial problem with this approach is to determine the net welfare value.

Weighted Criteria Indices

The weighted criteria indices is a procedure that has be proposed by the Delft Hydraulics Institute in the Netherlands in 1994. In this approach, five main criteria that contribute to a sustainable development are distinguished. Each of the five criteria is further subdivided into 4 sub-criteria

Table 13: Main criteria and respective sub-criteria for sustainable development (Baan, 1994)

Socio-economic	**Use of Natural Resources**	**Conservation of natural resources**	**Public health well-being**	**Sustainability of infrastructure**
Effects of income distribution	Raw material and energy	Water conservation	Effects of public health	Opportunities for a phased development
Effects on cultural heritage	Waste discharges	Accretion of land or coast	Effects on safety (risks)	Opportunities for multi-functional use and management and to respond to changing conditions
Feasibility in socio-economic structure	Use of natural resources	Improvement and conservation of soil fertility	Effects on annoyance/hindrance	Sustainable quality of structures
	Effects of resilience and vulnerability of nature	Nature development and conservation of natural values	Effects on living and working conditions	Opportunities for rehabilitation of the original situation

The impacts of a given project on water resources systems are assessed responding to the checklist-like criteria.

All criteria are given equal weights and the sum of the numerical values given to each sub-criterion is the sustainability index for the project that expresses the contribution of the project to sustainable development. Obviously, the higher the sustainability index, the higher the projects contribution to sustainable development. Based on computed value, the decision maker will accept, reject or modify the project.

Weighted Statistical Indices

Using the weighted statistical indices approach, an index of sustainability is computed in two steps; first, a set of suitable economic, environmental, ecological and social criteria is defined. The criteria have to be defined quantitatively or at least linguistically (e.g. “poor”, ”good”, “excellent”).

For any of these indicators values, an acceptable range has to be defined by determing upper and lower threshold values for the given indicator.

Time series of all those parameters are then derived by simulating the water resource system under consideration using different inputs or scenarios. Figure 19 illustrates a time series plot of a simulated values over the simulation period.

For all indicators, the statistical parameters

- reliability,
- resilience, and
- vulnerability

are computed, weighted aggregated to one single index that describes the contribution of a given set of actions or scenario to sustainable development.

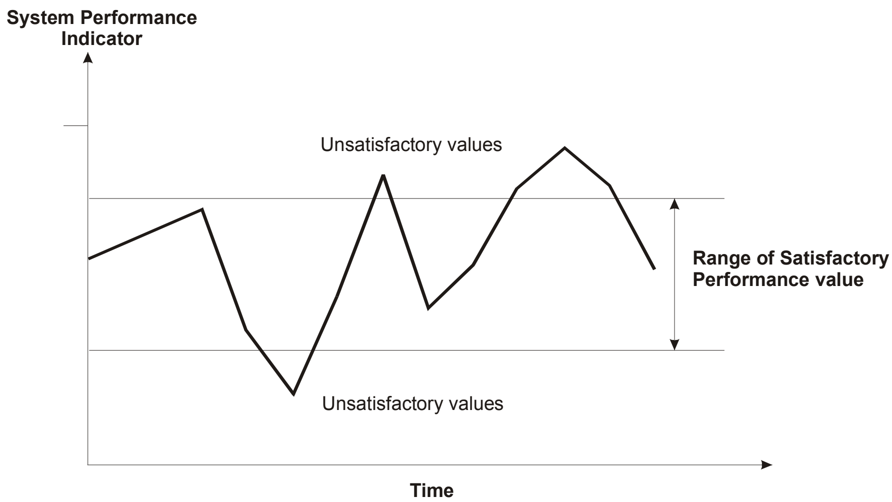


Figure 19: Measures of a system performance indicator (ASCE, 1998)

Reliability

Reliability is the probability that a criterion value will be with the predefined range of satisfactory values. Formally, it is defined by

$$Rel = \frac{N_S}{N}$$

where N_S denotes the number of values in a satisfactory range and N denotes the total number of simulated values.

Resilience

Resilience is an indicator for the speed of recovery of an unsatisfactory condition.

It is defined the number of times a satisfactory value follows an unsatisfactory value related to the total number of values.

Vulnerability

Vulnerability is a statistical measure of the extent or duration of failure. It is the amount a value exceeds the upper limit or the amount a value falls short of the lower limit, whichever is greater. Vulnerability can be related to the extend a value misses the satisfactory range or the duration of a continuous series of failure events.

The performance criteria are computed any simulated criteria, so that time series of reliability, resilience and vulnerability are produced. The system is improving (i.e. contributing to sustainability) over time if reliability and resilience are increasing and vulnerability is decreasing. One will find that the performance indices are improving for some criteria, while they may be worsening for other criteria. Again, weights can be assigned to the criteria to express the preference structure of the decision maker.

5.6.13 Other Sustainability Criteria

Consensus as a measure of sustainability

Simonovic (1997) proposes to use consensus, being defined as "a general agreement in opinion" as a measure of sustainability. Consensus is seen as a high level indicator that is measured at one moment of time, but it is implicitly assumed that the needs and values of future generations are some equitable combination of needs and values of today's generation.

The approach is an iterative process among the stakeholders, in which the degree of consensus is measured using the following five distance metrics to assess the degree of consensus.

Highest coincidence measure

$$\gamma^1 = 1 - \min_{i \neq j} \left| w_i x_i - w_j x_j \right| \qquad i, j = 1, \dots, n$$

Highest discrepancy measure

$$\gamma^2 = 1 - \max_{i \neq j} \left| w_i x_i - w_j x_j \right| \qquad i, j = 1, \dots, n$$

Integral mean coincidence measure

$$\gamma^3 = 1 - \frac{1}{n} \sum_{i=1}^{n} \left| w_i x_i - u \right.$$

Integral pairwise coincidence measure

$$\gamma^4 = 1 - \frac{2}{n(n-1)} \sum_{i=1}^{n-1} \sum_{j=1}^{n} \left| w_i x_i - w_j x_j \right|$$

Integral highest discrepancy measure

$$\gamma^5 = 1 - \max \left\| w_i x_i - u \right| \qquad i = 1, \dots, n$$

$$u = \frac{1}{n}\sum_{i=1}^{n} w_i x_i$$

where

n Number of decision makers

x_i distance metric value for decision maker i

w_i parametric control and weighting fo a decision maker

γ^k Degree of consensus for an alternative k

Distance metrics x and weights w are set in a way that the consensus measures γ are in the range [0,1]. "Measuring" the degree of consensus has the advantage that a numerical feedback is provided on discrepancies and coincidence and thus decision maker can be identified as supportive or otherwise. This, in turn provides some measure of the progress in negotiations in an iterative decision making process.

Fairness, Reversibility and Risk

Bender and Simonic (1997) argue that a number of issues is making sustainable decision making for water resources systems more challenging. Those issues include

- Expansion of spatial and temporal scales
- Risk and Uncertainty
- Multi-Criteria Analysis

They therefore formulate the following three criteria for sustainable project management and decision making:

- Intertemporal fairness
- Reversibility
- Risk.

Fairness

Intertemporal (also referred to as intergenerational) fairness considers both, the maintenance of social well being and the project acceptance by affected stakeholders.

Overall fairness is defined here as a combination of equity, equality and need-based fairness objectives.

Reversibility

Reversibility as a measure of sustainability is seen as the degree to which the aggregated set of anticipated and unanticipated impacts of the project can be mitigated. It is based on the assumption that a high degree of reversibility is related to a low disturbance of the natural environment.

Risk

The general definition of risk (product of the magnitude of negative effects and the probability of occurrence) is used here for projects with negative social, environmental and economic impacts. Risk is computed as an aggregated measure that is influenced by various components using historical and empirical data. The components are aggregated using weighting functions.

5.7 Candidate Indicators for the DSS

5.7.1 Indicator describing water quality and water quantity

The list below summarises a number of indicators that could serve the above described purposes of assessing water resources systems at catchment level. For indicators and data related to the economic analysis, see chapter 3.

A final list of appropriate indicators based on the selected paradigms is given in deliverable 8.

Water availability (GRWR[1]/cap)	S	Describes the availability of water per person per year in the region.
Exploitation index (per source)	P	Water production as percentage of GRWR
Precipitation Variability $Var=P_{high}-P_{low}/GRWR$	S	Describes the dependence of the region on the variability of precipitation.
Non-Sustainable Water Production Index	P	Measures the abstraction of non-recharging water resources (i.e. fossil groundwater) related to the total abstracted volume.
Agricultural water withdrawal	S	Gives the height of water on each m² of water managed area per year.
Dependence on upstream regions (Q_{in}/GRWR)	S	Gives the proportion of water coming from upstream regions or
GW abstraction rate	S	Volume of GW abstractions in the region over the estimated recharge volume per year.
Number of tourists per km coastline	P	Describes tourism activities along the sea.
Wastewater treatment rate	R	Measures the relationship of wastewater treated adequately in relation to the volumes of wastewater produced.
Share of irrigated agricultural land	P	Shows the dependence of the region on irrigated agriculture.
Water use efficiency for irrigation	R	The ratio between the volume of water actually reaching the plots and the total water volume allocated for irrigation. *May be difficult to assess*.
Number of nights per hundred inhabitants	P	Number of national and international overnight stays in hotels and similar establishments (H&A) compared with the number of inhabitants in the region.
Number of bed-places per hundred inhabitants	P	Number of beds in hotels and similar establishments (H&A) compared to the population annually. Number of beds x100/Total resident population
Share of collected water	R	Shows the proportion of water that is collected in the region
Economic efficiency of agriculture to water	S	Measures the agricultural yield achieved per volume of water.
Cost of water	R	Capitalised OMR cost plus energy consumed plus subsidies plus DMs investments related to the total volume of water produced (i.e. total supply to Agriculture, Industry and domestic users)
Price of water to user	S	Measures the price of water to users per sector.
Coverage of water demand per sector	R	Water supply as percentage of demand for all sectors
Share of treated water	R	Water treated as percentage of total waste water
WWTP connection rate	R	Measures the proportion of people connected to waste water treatment plants.
Generation of waste water by sector	P	Waste water (return flows) produced by sectors
Storage capacity/GRWR	R	Reservoir volume as percentage of GRWR
WQ of drinking water	P	Number of samples failing EU directive/total samples; if not linguistic description
Population growth rate	P	Growth rate of resident population (incl. migration)
Urban growth rate	P	Growth rate of urban population
Use of fertilisers/agrochemicals per ha	P	Data available?

[1] Global Renewable Water Resources (GRWR)= long-term average precipitation minus the long-term average evapotranspiration plus the long-term average inflows.

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7 Appendices

Indices Used in the Economic Analysis of Water Uses (in WATECO guidance document)

Drinking water supply

1) Population connected to public water supply
2) Population with self supply
3) Number of water supply companies

Wastewater treatment

1) Population connected to sewerage system
2) Population connected to waste water treatment plant
3) Number of wastewater treatment companies

Economic characteristics of key water uses

1) Agriculture
    - Total cropped area
    - Cropping pattern
    - Livestock
    - Gross production
    - Income
    - Total farm population
2) Industry
    - Turn over for the key sub-sectors
    - Employment for key sub-sectors
3) Hydropower
    - Installed power capacity
    - Electricity production
4) Navigation / transport (not relevant for the different selected areas for inland water)
    - Number of boats through key points per year
    - Employment linked to navigation
    - Quantity and value of goods transported
    - Quantity and value of goods through key harbors
    - Employment linked to harbor activities
5) Gravel extraction
    - Number of extracting companies

- Total employment
- Total turnover

6) Fish farming (not relevant for the different selected areas for inland water)

- Number of fish farm
- Total employment
- Total turnover

7) Leisure fishing (not relevant for the different selected areas for inland water)

- Number of person-days

8) Boating and wind-surfing (not relevant for the different selected areas for inland water)

- Number of person-days

9) Water related tourism (not relevant for the different selected areas for inland water but tourism in general is very important in these areas - it must be taken into account for the pressure it puts on water resources)

- Number of person-days
- Daily expense per tourism day
- Total employment in the tourism sector
- Total turnover of the tourism sector

10) Flood control

- Total population protected;
- Total turn-over of protected economics activities

Appendix 2 : Indicators Used in Environmental Economic Analysis

Indicators for a qualitative assessment of the economic situation (in relation with water resource management) (Global Water Partnership – 2002)

A change of state in the following indicators may have an impact on water needs and / or on the planning of water allocation.

1) *Access to safe drinking water*: number of litters per day and per person within 15 minutes walking distance. This indicator shows the proportion of the population with reasonable access to an adequate amount of safe drinking water (20 litters per day and per person as a minimum within 15 minutes walking distance).

2) *Urban population growth rate in years x and y*:

$$\left[(x-y)\cdot\sqrt{(Urban\ population\ in\ year\ y - Urban\ population\ in\ year\ x) - 1}\right]\cdot 100$$

This indicator is significant for sustainable development because it denotes increases in urban population pressures on natural resources, economy and society (unit: %)

3) *Tourism population growth rate in years x and y*:

$$\left[(x-y)\cdot\sqrt{(Tourism\ population\ in\ year\ y - Tourism\ population\ in\ year\ x) - 1}\right]\cdot 100$$

This indicator denotes increases in tourism population pressures on natural resources, economy and society (unit: %)

4) *Percentage of population on urban areas*: $\frac{Urban\ population}{Total\ population}\cdot 100$. This indicator shows the concentration of population in towns (unit: percentage)

5) *Share of irrigated agricultural land*: $\frac{Irrigated\ area}{Total\ cultivated\ agricultural\ area}\cdot 100$. This indicator shows the efforts of equipment for the intensive use of the water resources for the irrigation (unit: %)

6) *Population living below the poverty level in arid areas*: this indicator shows the level of development of the population

7) *Net migration rate*: this indicator is closely linked to urbanization and demographic indicators (unit: number of thousands inhabitants).

8) *Number of international tourists per 100 inhabitants*: $\frac{International\ tourist\ arrivals}{Population}\cdot 100$.

This indicator measures the intensity of international tourism in a country and the pressure exerted on the population. As far as possible, it would be interesting to have special data relating to peak of tourism times (unit: number per 100 inhabitants).

9) *Employment distribution (agriculture, industry, services)*: it shows changes in the distribution of the active population by economic sector. The proportion for tourism in services and agriculture is desirable but difficult to identify (unit: %)

10) *Share of tourism receipts in the exportations*: $\frac{International\, tourism\, receipts}{Exports\, of\, goods\, and\, services} \cdot 100$. It measures the importance of international tourism in exports and the contribution in currencies (unit: %).

11) *Annual energy consumption inhabitant*: this is the amount of energy (liquid, fossil, gas, or electricity) used by an individual in a given year in a given geographic area. (unit: ton oil equivalent per capita TOE).

12) Power price: (unit: €)

13) *Distribution of GDP (agriculture, industry, services)*: this indicator shows the contribution of each sector of activity in the Gross Domestic Product. It would be interesting to have the contribution of tourism in agriculture and services (unit: %).

14) *Foreign direct investments*: they represent capital inflows. They show the confidence of foreign investors in the country's economy.(unit: €)

15) $\frac{External\, debt}{GDP}$: This is the ratio of total foreign debt to Gross Domestic Product. It is the measure of the level of indebtedness, which helps to assess the foreign debt situation and debt servicing relative weight of a country. It needs to be interpreted carefully because it cannot, on its own, provide a complete picture of the debt situation of a country (unit: %)

16) $\frac{Public\, deficit}{GDP}$: This is the public deficit for the central government in a country in relation to the GDP (%).

17) $\frac{Current\, account\, balance}{GDP}$: The current account balance is considered to be a key indicator of the foreign power or weakness of a country. The ratio of the possible deficit to the GDP measures the size of the deficit (unit: %).

Indicators related to prices

1) *Average household income*: this indicator is essential for the evaluation of water cost acceptability and level of economic development (€ / year).
2) *Average household budget for domestic water* (€ / year)
3) *Average household budget for agricultural water* (€ / year)
4) *Average household budget for industry water* (€ / year)
5) *Average household budget for power generation water* (€ / year)
6) *Water price for household supply* (€ / m^3)
7) *Water price for agriculture supply* (€ / m^3)
8) *Water price for industry supply* (€ / m^3)

9) *Water price for power generation supply* (€ / m^3)

10) *Water price for tourism supply*: this indicator could be interesting in order to simulate a distinction between household water price and tourism water price (€ / m^3)

These seven previous indicators will allow the formulation of different strategic scenarios according to the adjustment of water price for each activity sector (industry, agriculture, household, tourism). A difference in prices for domestic water supply for permanent and seasonal population is expected, for example in cases when an increase in drinking water production is necessary to satisfy high tourist season. In practice, this different water price for seasonal population needs could be integrated in the tourism taxes (per night).