Water Management Methodologies for Water Deficient Regions in Southern Europe

Preface

Growing demand for water, uncertainties in natural water supply and new requirements imposed by environmental legislation are posing serious challenges at maintaining water quality and meeting demand for water resources.

An integrated management of water resources at river basin level ensures that social, environmental, technical dimensions as well as economic implications of water allocation are taken into account.

A traditional way to analyse and optimise such water resources problems it to apply Multi-Criteria Decision Making (MCDM) approaches based on scaled criteria that "measure" the degree of achievement of a given objective.

This volume on Methodologies for Water Management is based on a thorough investigation of fifteen regions in the Mediterranean and reviews some of the concepts and methods that are useful for river basin management. It is organised in the following manner:

Chapter 1 summarises models for the assessment and forecasting of water availability and demand, taking into account temporal and spatial variability and water shortage conditions.

The aim of C*hapter 2* is to present a methodological approach for the estimation of financial, resource and environmental costs associated with water management interventions and uses. The different sections of this chapter analyse the theoretical background for the estimation of the different components of the cost and propose a simplified, easy-to-implement approach for their computation. *Chapter 3* outlines different methods for MCDM in water resources management and describes basic requirements for indicators to "measure" the achievement of a given objective in the planning process.

The data requirements, structuring approaches and the most commonly used indicators in water resources management are presented in *Chapter 4*.

This volume has been compiled and edited by the Institute of Hydrology and Water Management at the University of Bochum, Germany. The contribution of Prof. Dionysis Assimacopoulos (section 1 of Chapter 2), Prof. Eli Feinermann (section 2 of Chapter 2) and Dr. Jean-Marc Berland (concluding section of chapter 2) are gratefully acknowledged.

Bochum, June 2004

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Chapter 1 Quantitative Analysis of Water Management Systems

Introduction

The management of water resources includes structural (physical works) as well as non-structural (conversation measures, efficiency improvments, economic instruments etc.) and should be conducted in a way that integrates technical, social, environmental and economical dimensions into a coherent framework.

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.

In this chapter, some of the methods and models for an assessment of water resources in terms of both, quantity and quality are reviewed.

The first section gives a broad overview of river basin systems and the interrelations of supply and demand components. Next, some models for assessing demand and supply at subsystem level are described. The final section discusses methods for estimating water demand and use for different sectors.

It should be noted that this chapter 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 Decision Support Systems (DSS) for water resources management will be described in a separate volume.

Water Resources Assessment

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.

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:

- Parameters, which are 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 define 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.

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 when water management interventions take 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).

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

The long-term natural water balance equation for any given catchment can be written as

$$P = ET + Q \pm L + \Delta S$$

where P is the total precipitation, ET is evapotranspiration, Q is total runoff including groundwater flow, L is leakage from and to the catchment area and ΔS represents a change in 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, combined balances of both groundwater and surface water to complex investigations of both groundwater and surface water and water quality and quantity.

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

A simplified schematic overview of quantitative water management analysis at catchment level is depicted in Figure 1.

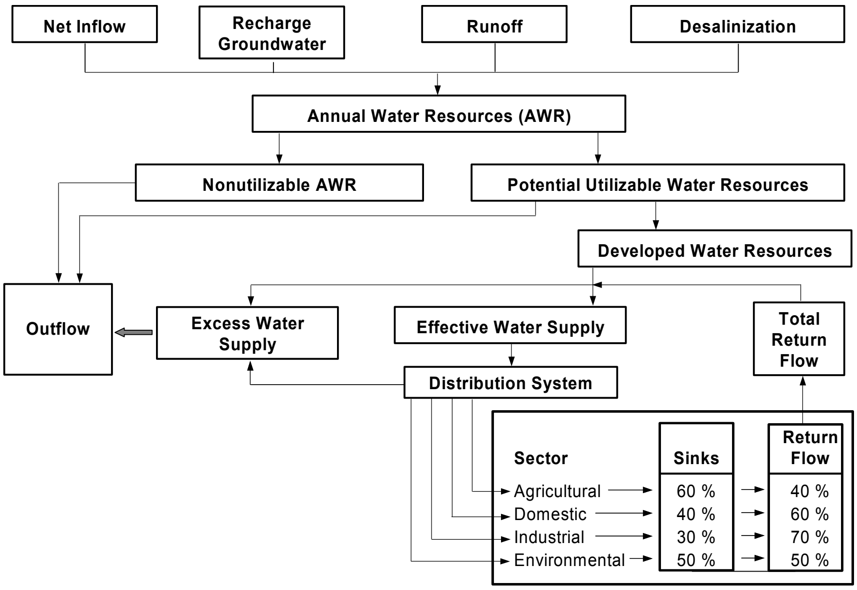

Figure 1: Schematised overview of water management analysis

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).

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.

The *spatial scale* of a water balance clearly depends on the objectives of the balance and the available data and may range from horizontal balance of a river reach to (sub-) basins or larger entities.

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.

A general framework for river basin management modelling is given in Figure 2.

The objective of the modelling exercise is to maximise the total socio-economic benefit of the river basin. Benefits include economical values of municipal and industrial (M & I) use, profit from irrigation as well as profit from in-stream uses. The entire system is controlled by institutional decisions on water management policies such as tariffs, allocation decisions, environmental constraints and others.

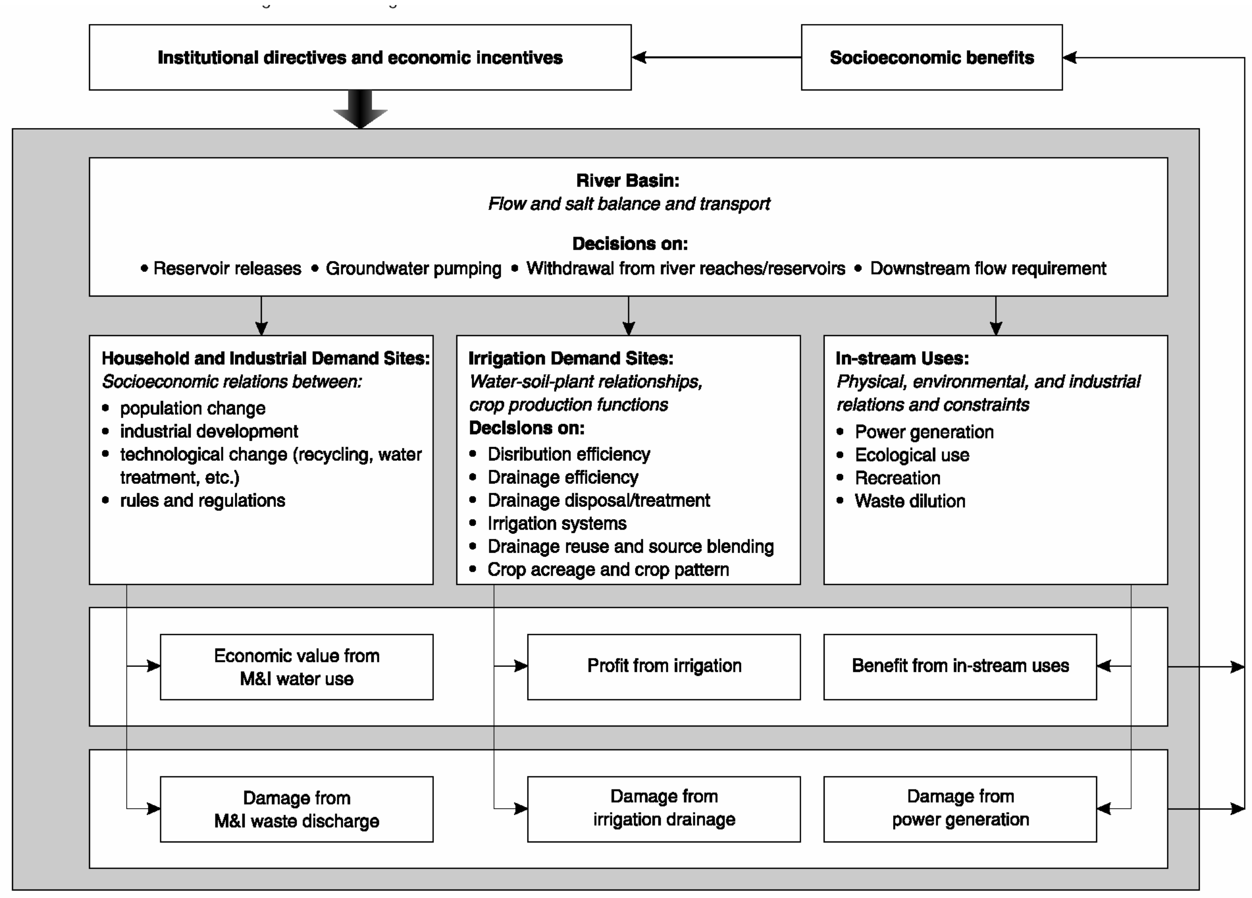

Figure 2: Framework for river basins management modelling (IWMI, 1999)

Water management structures are 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 the following section, a few concepts for assessing water resources availability at subsystem level will be described.

Groundwater resources

Groundwater resources are of high importance, especially in arid and semi-arid regions where surface water is limited and groundwater resources supply the bulk water for agriculture and domestic use.

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 require 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.

The question whether abstractions from fossil aquifers must be generally considered as non-sustainable is discussed controversy. Meadows (1992) notes that *The sustainable rate of use can be no greater than the rate at which a renewable source, used sustainably can be substituted for it*.

The physical structures related to groundwater management comprise of 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. In many 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 exceeds 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.

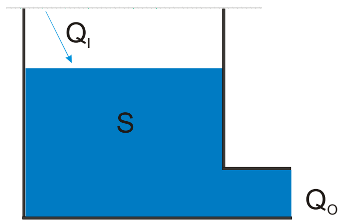

Figure 3: Conceptualisation of an aquifer as a Linear Reservoir Model

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 groundwater quality exist but such models can only give a vague picture of the quality of groundwater.

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.

Runoff at any point in a river is contributed by runoff from the catchment area upstream of that point and discharge from groundwater bodies. Rivers and streams may be perennial (runoff occurs throughout the year), intermittent (some reaches of the river temporarily dry up) and ephemercal (flows only after rainfall).

Functional relationships between characteristics of the catchment, amount and temporal distribution of rainfall can be used to estimate the runoff at a given point in the river for a given time.

Seasonal patterns in discharge can be indicated by mean monthly hydrographs. A common method for providing a graphical summary of discharge variability are flow duration curves that plot discharge as a function of the percentage of time that this discharge is exceeded.

Approaches for limiting the pollution in rivers and lakes can be divided into two classes: Effluent (emission limit values)-based approaches and quality based approaches.

Emission limit values (ELV) can be defined as regulatory measures aimed at the source of potential environmental pollution. They are used to restrict the level of permissible pollutant emissions to the environment by means of general or abstract limit values. This approach is guided by such concepts as 'state-of-the-art technology' or the highly economically oriented 'best available technology'.

Environmental quality standards (EQS) focus on the pollution target. They can therefore be described as rules relating to environmental quality. They are generally concerned with individual aspects of the environment, such as a particular medium (soil, water and air) or a specific target (e.g. human beings, ecosystems). For these targets, environmental quality standards outline a desirable quality level.

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

Total organic carbon (TOC), total dissolved solids (TDS), 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 temperature, the temperature should additionally be considered.

Marine and coastal waters

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

Conjunctive management of groundwater 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 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 4.

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.

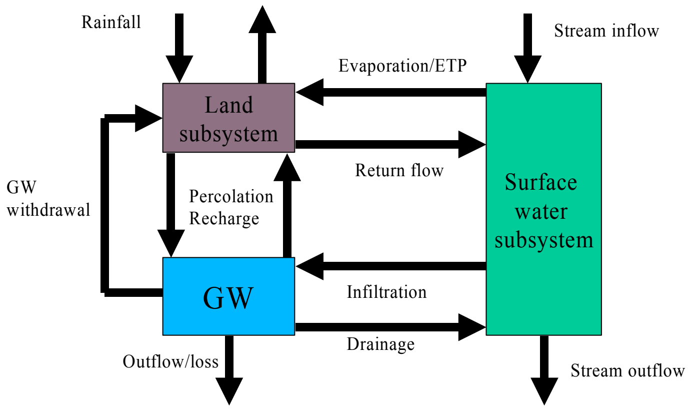

Figure 4: Conceptual model for conjunctive use (Stephenson, 1998)

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 into 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 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 is shown in Figure 5 (U_t and Y_t denote reduced and additional draft as a function of storage)

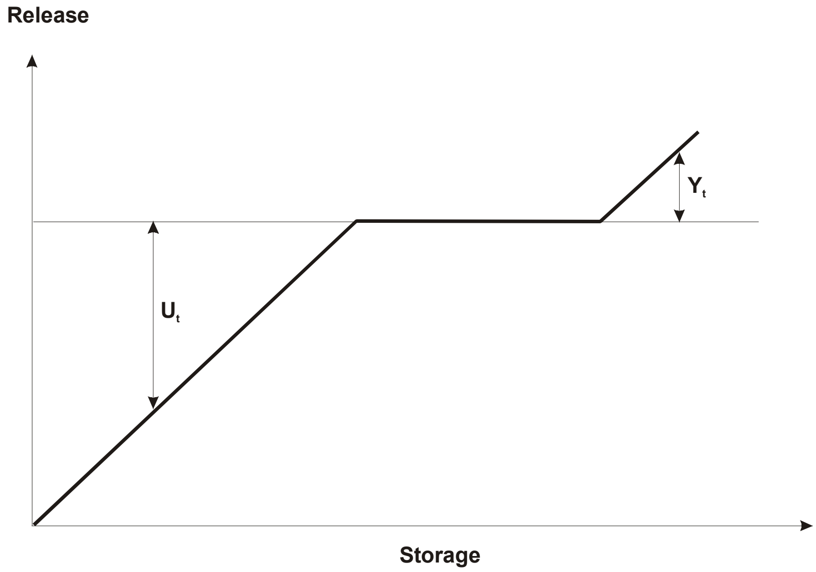

Figure 5: Schematic representation of the Hedging rule for reservoir management

Hydropower

The water demand for hydropower facilities connected to a reservoir structure can be calculated by estimating 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:

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

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). A desalination plant can also be 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.

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 Water Framework Directive (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.

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 estimate 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 Hampshire 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.

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 6 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 of 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.

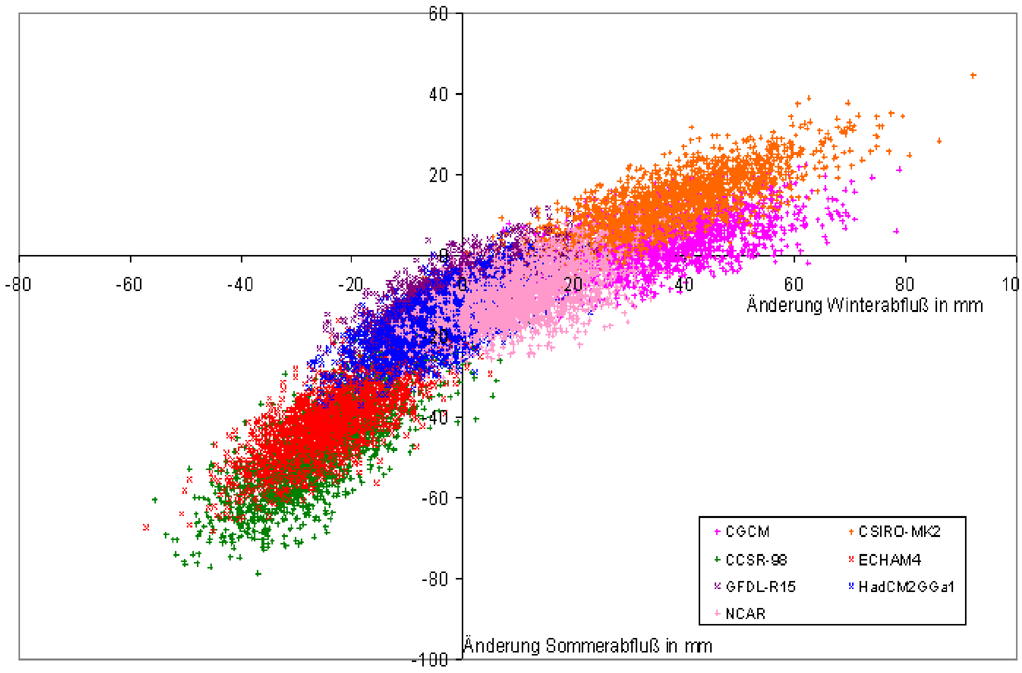

Figure 6: Impacts of climate change on the upper Danube river, deviations of summer and winter runoff in mm (Schumann and Antl, 2002)

Assessment and Forecasting Water Demand

In a very general way, water demand can broadly be classified in offstream and instream use. Instream use refers to the water that is used but not withdrawn from groundwater or surface water for purposes such as hydroelectricity, navigation etc while offstream use refers to water that is withdrawn and diverted from a source.

The following subsection describes the entities that demand water. Several demand types are distinguished: industrial demand, agricultural demand, domestic demand, demand for hydropower, environmental demand and demand for tourism. If these different demands are represented in a conceptual model it may be necessary to further subdivide the requirements. For example, it may be important to distinguish domestic demand that is largely influenced by tourism and residential domestic demand to take into account different underlying driving forces and demand patterns.

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).
- Mining

Factors affecting the water demand for industry include type of industry, tariffs, extent of water reuse and water saving technologies, water conservation programmes and others.

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. but cannot be computed independently of the above mentioned criteria.

Agricultural water demand

The term agricultural water demand here refers to four different types: 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 to be made between water that is used consumptively 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 K_c that accounts for crop characteristics, crop development, vegetation periods and others. Reference evapotranspiration ET_0 is defined as the evapotranspiration from an extensive surface of green grass cover of a height of 12 cm adequately watered. The net 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 water 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). It should be noted however, that the greatest uncertainty in estimating agricultural water demand comes from the efficiency estimate that is primarily controlled by the type of applied irrigation technique and can range from 10% to 90%.,

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.

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.

Further information on the above equation is given in the FAO's Irrigation and drainage paper no. 33 entitled "Yield response to water".

Domestic water demand

The water demand and use of human settlements (urban demand) includes demand in-house uses such as drinking, cooking, kitchen and toilet use, and out-of house for 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 (see Table 1).

As can be seen in Figure 8, leak size does not have to be significant. A 7 mm diameter leak already causes losses of some 1700 m³ of water per month (Computed for average conditions and for a pressure of about 15 bar).

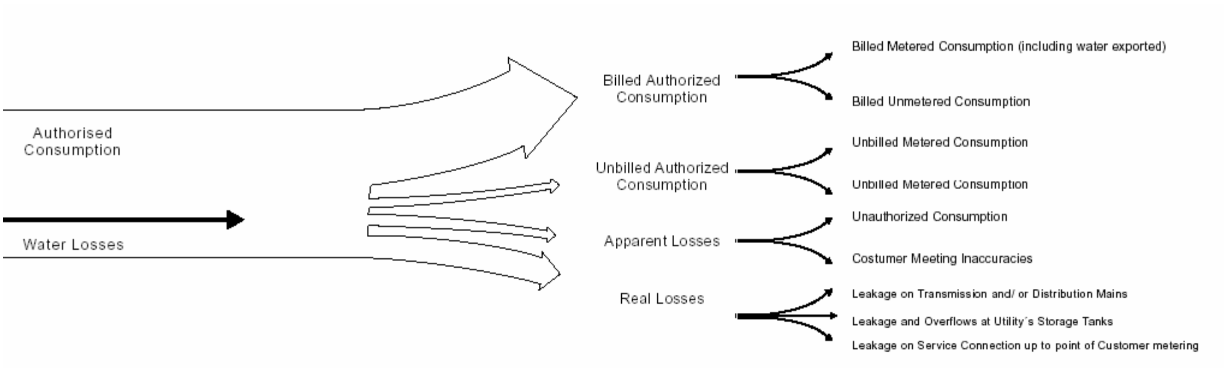

Figure 7: IWA Approach for water losses

Actual leak Size [mm]	l per		m³ per	
	minute	hour	day	month
0.5	0.33	20	0.48	14.4
1.0	0.97	58	1.39	41.6
1.5	1.82	110	2.64	79
2.0	3.16	190	4.56	136
2.5	5.09	305	7.30	218
3.0	8.15	490	11.7	351
3.5	11.3	680	16.3	490
4.0	14.8	890	21.4	640
4.5	18.2	1100	26.4	790
5.0	22.3	1340	32.0	960
5.5	26.0	1560	37.4	1120
6.0	30.0	1800	43.2	1300
6.5	34.0	2050	49.1	1478
7.0	39.3	2360	56.8	1700

Figure 8: Leakage size and losses (Stephenson, 1997)

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

Albania	up to 75
Armenia	50 - 55
Bulgaria (Sofia)	30 - 40
Bulgaria (other than Sofia)	More than 60
Croatia	30-60
Czech Republic	20 – 30
Denmark	4 – 16
Finland	15
France (national average, 1990)	30
France (Paris)	15
France (highly rural area)	32
Germany (former West Germany, 1991)	6.8
Germany (former East Germany, 1991)	15.9
Germany (average, 1991)	8.8
Hungary	30 - 40
Italy (national average)	15
Italy (Rome)	31
Moldova	40 - 60
Romania	21 – 40
Slovakia	27
Slovenia	40
Spain	24-34
Ukraine	Around 50
UK (England and Wales)	8.4 m^3/km mains pipe/day 243 l/property/day

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

The amount of water that is needed to sustain an ecological value of an aquatic ecosystem is referred to as environmental water demand. The environment is increasingly being considered a legitimate water user in many European countries. Although the demand that is needed for the environment is a decision taken by the society, it has to be estimated as accurately as possible.

Besides the ecological demand there may be different requirements for instream water uses for different reasons such as navigation, prevention of saline intrusion, protection of the rights of abstractors, social and political reasons and others.

Methods for estimating instream flow requirements range from simple hydrological indices (e.g. flow duration curves, aquatic base flow method) to complex hydrological and habitat simulation models that provide information of how the habitat will change under given hydraulic conditions.

Forecasting water demand and use

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

The most influential factors affecting water demand and use are related to population, level of service, tariffs, demand management measures as well as climatic conditions.

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). In practice, the data available for water demand forecasts does not permit statistical analysis yielding to forecasts that is given with confidence limits around the forecasts. A common practice of dealing with uncertainties is therefore to model a number of scenarios to incorporate the sensitivity of given factors on a long-term perspective (e.g. low, medium and high growth rates for population).

Different methods for forecasting water demand and use are briefly described below.

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:

$$W = a_1 x_1 + a_2 x_2 + \ldots + a_n x_n + e$$

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 are a number of limitations; such approaches do not contain any allowance for price elasticity of demand and other factors.

Trend analysis

Trend based forecasts use historical data of water demand to fit mathematical functions that can be used to estimate future water use. The type of function that is fitted to the data depends on the data and on the choice of the user. Although this technique is very easy to implement and does not require extensive data on water demand it suffers from great uncertainties as it does not consider the driving factors for water demand and assumes that recorded water demand is representative of future water demand.

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 9.

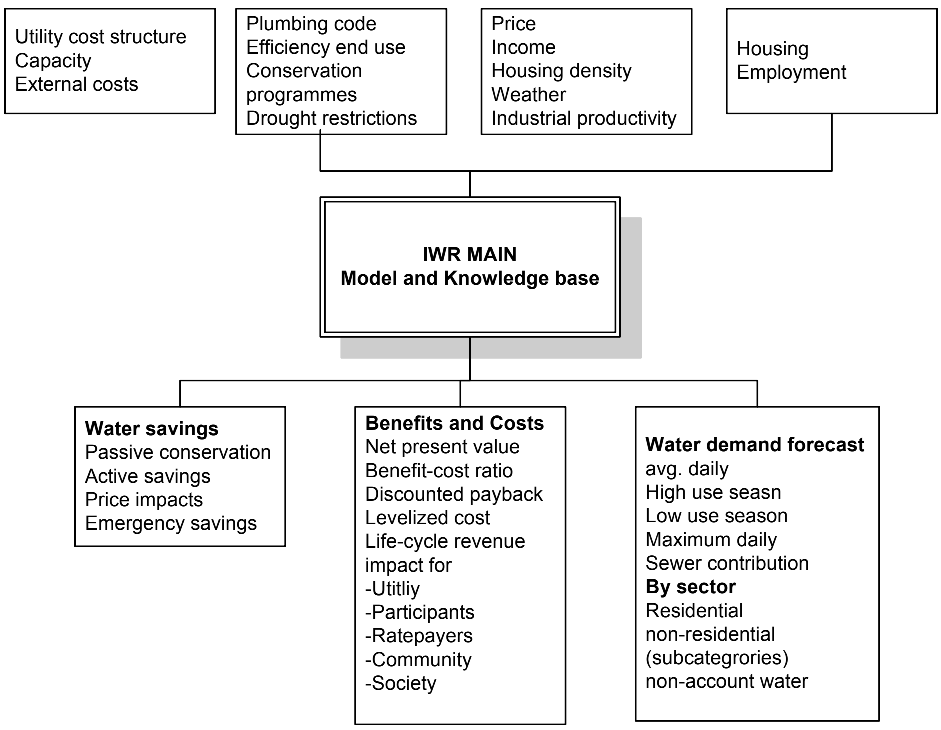

Figure 9: Structure of the IWR MAIN model

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 determines 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 if 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.

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	

River basin models

Without going into detail the following section briefly outlines some of the basic concepts for river basin models but is not intended at presenting readily available software packages or commonly used models, as these are described in detail in a different volume.

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.

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.

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 models, 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.

River basin quality simulation models

Although this section 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 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 two-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 strongly depends on the type of water body, amount of data, spatial and temporal resolution required, and many other factors.

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

$$minimise\left(\sum_i d_i - \sum_j f_j\right)$$

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

$$\underset{p\in P}{maximise}\, NB\left(X_P\right)$$

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.

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 to 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.

Daene et.al. (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)	

Water Resources Planning Under Uncertainties

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; it concentrates on methods that can be used in water resources planning.

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 10.

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 an extension of the deterministic case.

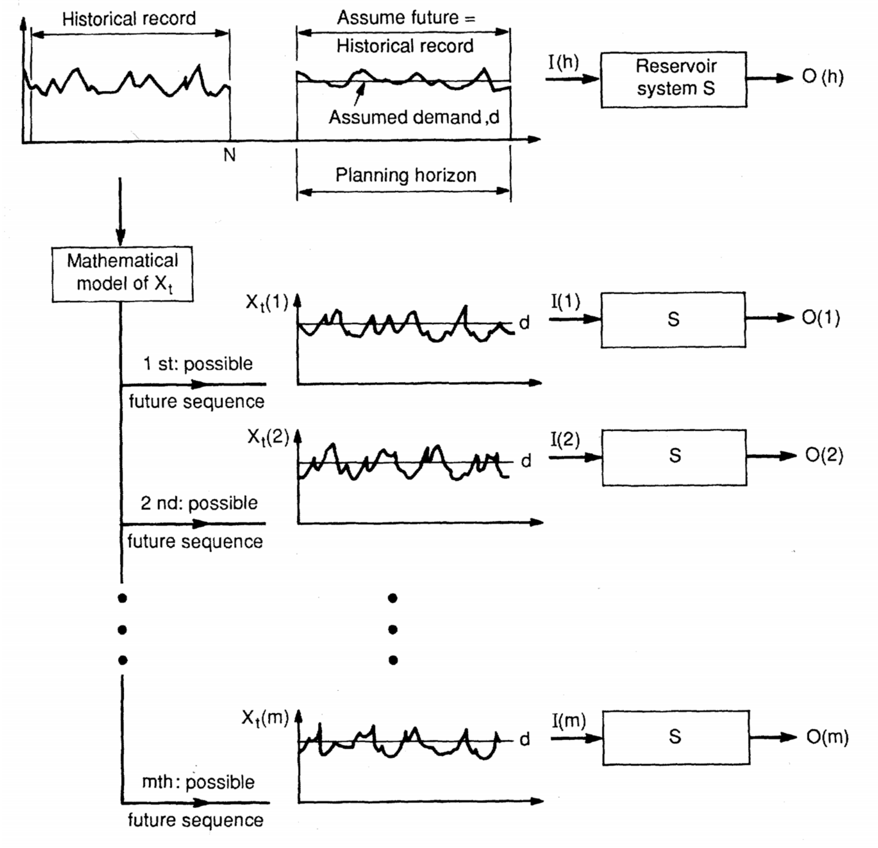

Figure 10: Schematic representation of a hydrologic system for Monte Carlo simulation

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 as "large," "medium," and "small" are each used to capture a range of numerical values.

Chapter 2 Economic Analysis of Water Resources Systems

Introduction

Water is a social good, irreplaceable for survival, human health and economic growth with important cultural or even religious value. Availability of high quality fresh water improves the individuals' welfare and benefits society as a whole. In this sense, water is not just a social but also a common good and access to clean water is a basic right of all. Social goods in some cases have also the characteristics of private goods. More water for someone may mean less water for others, which share the same water resources.

Increasing needs for water services in the 80's, led to the formulation of the *"Dublin Principles of Water"*. Among the 4 principles adopted with the Dublin declaration, the most controversial and confusing one was that *"water has an economic value in all its competing uses and should be recognised as an economic good"* (International Conference on Water and Environment, Dublin 1992). In fair interpretation, this statement does not mean that water is a commercial good but simply that it has a different value in competitive uses. Management of water as an economic good means that water should be allocated to competitive uses in such a way that the net social benefit is maximised. These arguments led to the opinion that *"integrated water resources management is based on the perception of water as an integral part of the ecosystem, a natural resource, and a social and economic good"* (Environment and Growth, Rio 1992).

The Water Framework Directive 2000/60 builds on these principles and constitutes a bold and forward-looking instrument for the future management of water and aquatic ecosystems throughout Europe, being the EU's first *"sustainable development"* Directive. It expresses a basic change in the priorities of water resources management, which has already taken place in the 90's. The Directive takes into account the value of water for the environment, human health and consumption in productive sectors and establishes a framework for the protection of inland surface waters, transitional waters, coastal waters and groundwater with the specific goals to:

- Prevent further deterioration, protect and enhance the status of aquatic ecosystems and, with regard to their water needs, terrestrial ecosystems and wetlands directly depending on the aquatic ecosystems;
- Promote sustainable water use based on a long-term protection of available water resources;
- Enhance protection and improve the aquatic environment through specific measures for the progressive reduction of discharges, emissions and losses of priority substances and the cessation or phasing-out of discharges, emissions and losses of the priority hazardous substances;

- Ensure the progressive reduction of pollution of groundwater and prevent its further pollution;
- Contribute to the mitigation of the effects of floods and droughts.

Article 9.1 of the Directive refers to the recovery of the costs of water services and clarifies the cost components that should be included in full cost.

Figure 11 presents the components of full water cost that include:

- The direct (financial) cost that represents the costs of investments, operation and maintenance, labour, administrative costs and other direct economic costs.
- The resource cost that represents the loss of profit because of the restriction of available water resources.
- The environmental cost that represents the cost from the damage on the environment and aquatic ecosystems incurred by water uses and services.

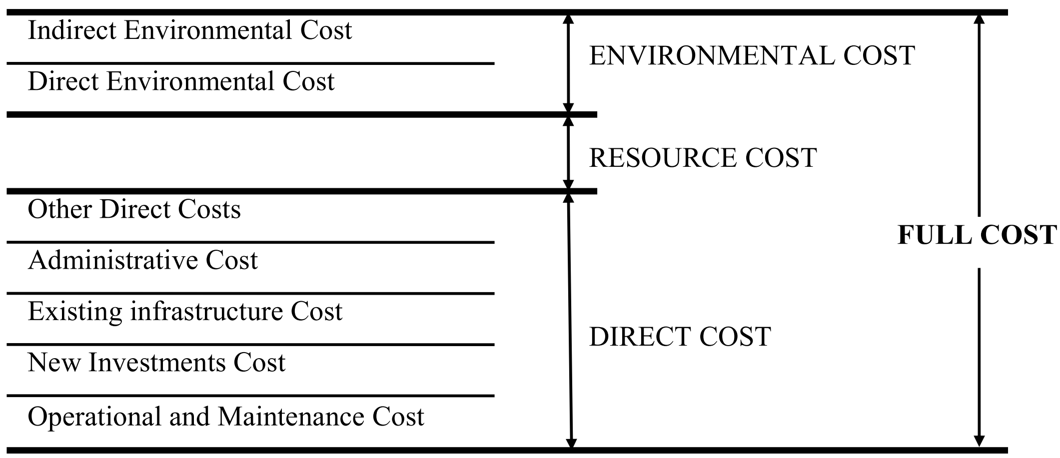

Figure 11: Components of the full cost of water services

The Water Framework Directive in fact **does not refer to the "full cost recovery"**. Member States must only report to the European Commission the **level of cost recovery** of water services. Of course, there is no obligation for the Member States to recover the full cost, but only to ensure that water uses contribute **adequately** to the costs of water services.

The WATECO Guidance Document aiming to guide experts and stakeholders in the implementation of the economic elements of the Water Framework Directive, proposes however an overall methodological approach. Due to the diversity of circumstances within the European Union and the different harmonisation procedures to be adopted by the member states, it does not offer specific guidance for dealing with specific issues that may arise in each river basin.

In this context, the following sections of this document analyse the theoretical background for the estimation of the different cost components and propose a simplified, easy-to-implement approach for their estimation.

Direct (or Financial) Costs

The estimation of direct costs is rather straightforward but involves the choice of suitable values for all the parameters as investment cost and lifetime, discount rates, value of existing infrastructure and of appropriate depreciation methods. General taxes and subsidies are not included, while the environmental taxes, which may represent internalised environmental costs, are included in the analysis.

Generally, direct (or financial) costs are the costs brought about by providing and administering water services. In this context, they can be broken down in a number of cost elements:

- Operational costs, defined as all costs incurred to keep a facility running
- Maintenance costs, defined as the costs for maintaining existing (or new) assets in good functioning order until the end of their useful life.
- Capital costs
    - New investment expenditures and associated costs
    - Depreciation of existing infrastructure, representing an annualised cost for replacing existing assets in the future
    - Cost of capital, representing the opportunity cost of capital
- Administrative costs, related to water resource management
- Other direct costs, such as those related to the loss of productivity due to restrictive measures.

The Total Direct Cost, excluding administrative costs is estimated as the sum of annual costs:

$$TDC = \sum_{i=1}^{n} AEC_i + OMCost_i + ECCost_i$$

where:
TDC is the total direct cost
AEC_i is the annual equivalent capital cost for infrastructure i
$OMCost_i$ are the annual operational costs (excluding energy consumption) for infrastructure i.
$ECCost_i$ are the annual energy costs for infrastructure i

Annual equivalent capital costs for a part of the infrastructure can be determined as:

$$AEC_i = \frac{CapitalCost_i \cdot DiscountRate}{1-(1+DiscountRate)^{-DepreciationPeriod_i}}$$

Energy costs are determined through the energy consumption for a part of the infrastructure.

The minimum data required for the estimation of direct costs include:

- Construction cost and depreciation period for the estimation of annualised capital costs,
- Operation and maintenance costs in terms of €/m^3 produced or treated and
- Energy consumption in kWh/m^3 produced or treated for the estimation of variable costs. For desalination plants, the net energy consumption (including possible energy recovery) should be entered.
- Energy prices in €/kWh consumed. Energy prices should be entered according to the total monthly energy consumption.

Costs, besides energy prices, should be defined for every single (or at least major) part of the infrastructure (dams, desalination, drinking water treatment, desalination, wastewater treatment, renewable groundwater, pipelines, sewer systems etc).

Direct costs are allocated to particular water uses proportionally to the volume of water distributed to each of them on an annual recording, based on the principle that each user should pay for the part of the infrastructure that he is using. For each use, the additional direct cost for allocating water is estimated, summing the direct costs from all infrastructure used for supplying the particular use. For domestic uses, an additional operation is to be performed in order to add the direct cost related to wastewater treatment.

In case of uses sharing the same supply sources or a part of the same distribution system, the total direct cost is distributed according to the supply allocated to each one. The operation is different according to the direct cost component:

- Annual equivalent capital costs are distributed according to the yearly share of the supply delivered vs. the total volume of water provided. This determines in the long run the part of the infrastructure that is used by each water use.
- Running costs (operation and energy) are distributed according to the monthly share of delivered supply.

Allocation of Limited Water Resources among Competing Users

An approximation for resource costs

Rational water users, whether they are urban consumers or agricultural producers, would buy an additional unit of water as long as its price does not exceed the benefit they can derive from it. Thus, the marginal value of water (*MVW*) to a user is the maximum utility (for urban consumers) or benefits (for producers) generated by the last water unit in use. Moreover, if the marginal value of water in one activity (e.g., agricultural use) is different than in another activity (e.g., industrial use), then transferring one (marginal) unit of water from the lower to the higher marginal value activity would increase the total benefits derived from the two activities without changing the total amount of water in use. Thus, water allocation that maximises total benefits derived from a given amount of water supply must equate the marginal values of water across all users. Such an allocation is called by economists an **efficient allocation**. It is convenient to clarify this concept via a simple illustrative example and graphical analysis.

Let's consider a single water source, with limited amount of available water of $\bar{Q}$ and two competing users: an urban centre (hereafter an urban user) and an agricultural plot (hereafter an agricultural user). The water demand curves for the urban and the agricultural users, which match the *MVW* for each of the users, are denoted by D_U and D_A, respectively (see Figure 12). Let's further assume that the specific direct marginal costs of extracting, treating and delivering the water from the water source to the urban and the agricultural users (including costs of labour and capital) are equal to MC_A and MC_U, respectively. To simplify the presentation it is additionally assumed that $MC_A = MC_U \equiv MC$ (see Figure 12). It should be noted that 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.

Figure 12 depicts that efficient water allocation is obtained when Q_U of water is allocated to the urban user and Q_A is allocated to the agricultural users ($Q_U + Q_A = \bar{Q}$). Under this allocation the *MVW* of water is evenly distributed across users.

Quota-based vs price-based allocation

In principal, there are two possible mechanisms (systems) to implement optimal allocation. The first one is administrative; a central public agency or a government will allocate quotas of Q_U and Q_A to the urban and the agricultural users respectively. The second one is price-based allocation; through setting the price of water at a level of

$$P = MVW^U(Q_U) = MVW^A(Q_A),$$

where $MVW^U(Q_U)$ and $MVW^A(Q_A)$ are the marginal values of water in the urban and the agricultural sectors, respectively. It should be noted that **the marginal values of water are the demand functions for water**, denoted by

$$D^U (= MVW^U(Q)) \text{ and } D^A (= MVW^A(Q)).$$

Optimal quota-based allocation requires a perfect knowledge on the demand curves of every user. Price-based allocation requires knowledge of the equilibrium price which balances demand with the (limited) supply. Such a price can be found by trial and error. If in one year the set price is too high, and the aggregate demand falls short of the fixed supply, then it can be reduced the next year. Moreover, in contrast to a quota system, prices have no preferences and therefore limit the ability of the relevant authorities to differentiate between users. Finally, it should be emphasised that in practice, water allocation is performed via **combination of quotas and prices.** Two examples are given in the paragraphs that follow.

Opportunity costs and scarcity rent

Note that if the urban centre will increase water use, the amount of water available for agriculture will be decreased by the same amount. The reduction in the value of agricultural production is the cost to society resulting from the increased water use in the urban centre.

Similarly, the reduction in the value of domestic water is the cost to society resulting from the increased water use on the agricultural fields. This specific cost, which is equal to P, is an **opportunity cost** – the benefits foregone when a scarce resource is used for one purpose instead of the next best alternative.

It should be emphasised that **all** costs in the economy are always *"opportunity costs":* energy, capital and labour used to extract and convey water to the farmers are not available to serve the urban centre and do not contribute to its welfare. In the next paragraphs the commonly used term of *"direct costs"* will be adopted to represent the costs of the relevant inputs (like energy, capital, labour, etc.) that can be bought in the markets and whose prices are known. *"Available water supply"* is the only input in our example that cannot be purchased in the markets.

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 equilibrium price P) and the per unit (marginal) direct costs (such as extraction, treatment and conveyance) of turning that natural resource into relevant products (agricultural crops for farmers and water services for the residence of the urban centre). 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 $\overline{Q}$.

To be more specific, the scarcity rent, denoted by λ (see Figure 12), is defined by $P - MC$, i.e., the equilibrium price of water (which is equal to $MVW^U(Q_U)$ as well as $MVW^A(Q_A)$) minus the marginal direct costs. If the quantity of water available for one of the users (either farmers or urban residents) is reduced, his marginal benefits will be reduced by $MVW^U(Q_U)$ (or $MVW^A(Q_A)$) but, at the same time, the cost of MC will be saved, implying a net loss of $\lambda = P - MC$. Or equivalently, increasing an increase in available supply of 1 m^3 will increase the net marginal benefits for the two-user economy by λ €. If the additional cubic meter is delivered to the farmers or to the urban centre or is split between the two users, its contribution to the total benefits of the economy is equal to the marginal value of water, $MVW^U(Q_U) = MVW^A(Q_A)$, minus the marginal direct costs associated with the supply of this cubic meter, MC.

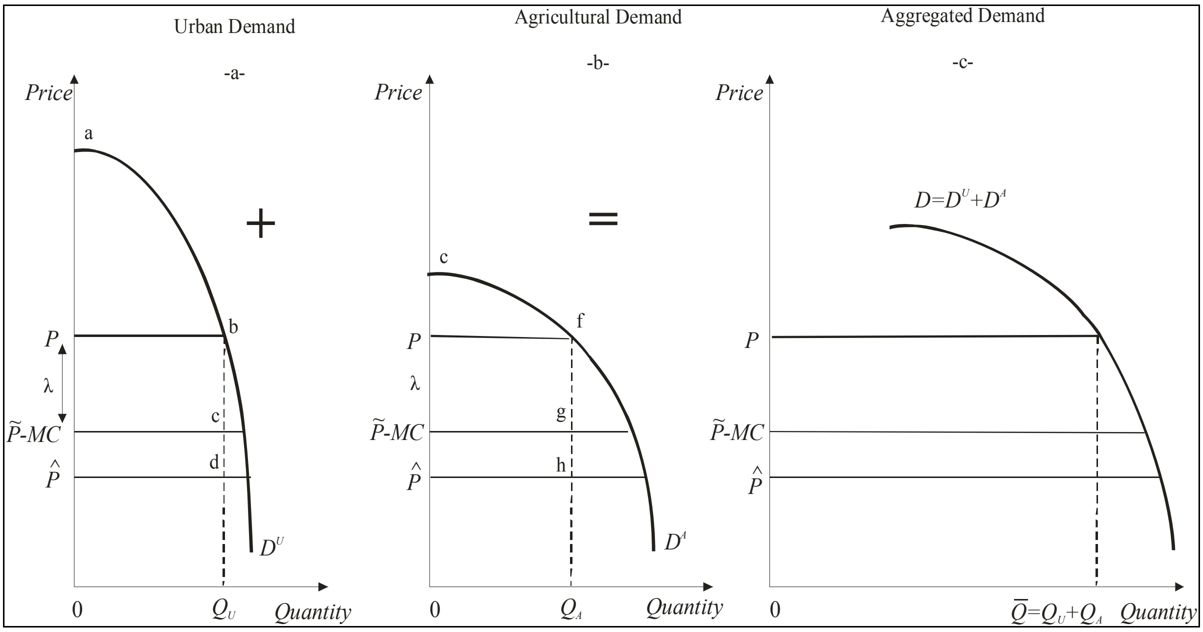

Figure 12: Optimal allocation of a limited water resource

Social vs. private water surplus, rate of cost recovery

The opportunity costs (P) of water in our example are equal to the sum of the marginal direct cost and the scarcity rent, i.e. $P = MC + \lambda$ €/m^3. Thus, the **total social costs (i.e., total water-associated costs accrue to the economy under consideration)** of supplying Q_U to the urban centre and Q_A for irrigation are $P \cdot Q_U$ (the area of the rectangular [PbQ_U0] in Figure 12a) and, $P \cdot Q_A$ (the area of the rectangular [PfQ_A0] in Figure 12b), respectively.

If the scarcity rent is ignored (as it is the case in the vast majority of the water economies of the developed countries), the **total direct costs** of supplying Q_U to the urban centre and Q_A for irrigation are $MC \cdot Q_U$ (the area of the rectangular [$MCcQ_UO$] in Figure 12a) and $MC \cdot Q_A$ (the area of the rectangular [$MCgQ_A0$] in Figure 12b), respectively.

The demand function for water gives the amount of water that will be demanded at each water price. But it can also be used to calculate the users' surpluses. To see this, let us consider the case under which water is allocated by prices: by setting the price of water at a level of $P (= MVW^U (Q_U) = MVW^A (Q_A))$. Urban residents and farmers will demand the quantities Q_U and Q_A, respectively. As mentioned above, the marginal value of water (MVW) to a user is the maximum utility (for urban consumers) or benefits (for agricultural producers) generated by the last water unit in use. Water input prior to this last unit has generated larger revenues (this is a consequence of the diminishing marginal value of water, i.e., downward sloping demand curves). Since the total revenue is the sum of revenues of all units of water input up to the level Q_U for the urban users and Q_A for the agricultural users, we see that after paying P for each unit of water input each user is left with some surplus. This surplus is the area between the demand for water and the water price. Specifically at water price P, the private water surplus for urban users is given by the area [abP] in Figure 12a, and the private water surplus for agricultural users is given by the area [efP] in Figure 12b.

As long as actual water price (P) is set at a level equal to the opportunity costs, $MC + \lambda$, the sum of the private water surpluses of the various users is equal to the social water surplus, which is given by the area between the aggregate demand curve, D, and the opportunity costs $P\ (= MC + \lambda)$, (see Figure 12c). Moreover, the total social costs of supplying the water, $P\ (Q_U + Q_A)$, is fully covered by the water charge. In other words, the rate of cost recovery is equal to 1 and the rate of recovery of the total direct costs,

$$\frac{P \cdot (Q_U + Q_A)}{MC \cdot (Q_U + Q_A)} = \frac{P}{MC}$$

is greater than 1.

Subsidised water prices – two examples

In reality water prices for consumers are subsidised and do not fully cover the opportunity costs of water. Below are two examples of interest, in which water is allocated via quotas and administrative consumer prices are used to recover some of the opportunity costs.

Example 1

Let's assume that consumer water price is set by the public agency at a level of $\tilde{P} = MC$ (see Figure 12) which is lower than the economically efficient price, P. In the absence of administrative quotas, urban demand will exceed Q_U, agricultural demand will exceed Q_A and aggregated demand will exceed the limited supply $\bar{Q}$. To obtain an efficient allocation of the limited capacity, the agency allocates water quotas of $Q_U\, m^3$ to the urban user and $Q_A\, m^3$ to the agricultural user. With water price lower than P the quotas will be fully utilised by the urban and agricultural consumers.

Inspection of Figure 12a and Figure 12b allows us to conclude as follows:

- **Private water surpluses** for urban consumers and for agricultural consumers are given by the areas $[abc\tilde{P}]$ (Figure 12a) and $[efg\tilde{P}]$ (Figure 12b), respectively. The surplus for the water producers is zero (water charges cover exactly the total direct costs).
- The total direct cost of water supply, $MC \cdot (Q_U + Q_A)$, is fully covered by the water charge. In other words, the rate of recovery of the direct cost is equal to 1. However, the rate of cost recovery of total social costs is lower than 1, i.e.,

$$\frac{\tilde{P} \cdot (Q_U + Q_A)}{P \cdot (Q_U + Q_A)} = \frac{\tilde{P}}{P} < 1 .$$

The gap between the actual price, $\tilde{P}$, and the economically efficient price, P, should be covered by governmental subsidy. The total subsidy would be equal to $(P - \tilde{P}) \cdot (Q_U + Q_A)$ €. Some portion of the subsidy can be transferred to water producers in order for them to have a positive surplus. The remainder can be used to recover capital expenditures for system expansion, upgrades, equipment replacement and more.

Example 2

In the second example we assume a consumer price of $\hat{P}$ €/m^3 which is lower than marginal costs MC (see Figure 12). Under this price, the water quotas Q_U and Q_A are fully utilised. Similarly to the previous example, inspection of Figure 12a and Figure 12b allows for the following conclusions:

- Water surpluses for urban consumers and for agricultural consumers are given by the areas $[abd\hat{P}]$ (Figure 12a) and $[efh\hat{P}]$ (Figure 12b), respectively. The surplus for the water producers is negative and equal to $-(MC-\hat{P})\cdot(Q_U+Q_A)$ €.
- In contrast to the case discussed in the previous example, the total direct cost of water supply, $MC\cdot(Q_U+Q_A)$, is not fully covered by the water charge. In other words, the rate of recovery of the direct cost is smaller than 1, and equal
$$\frac{\hat{P}\cdot(Q_U+Q_A)}{\tilde{P}\cdot(Q_U+Q_A)}=\frac{\hat{P}}{\tilde{P}}<1\,.$$
Obviously, the rate of cost recovery of total social costs is smaller,
$$\frac{\hat{P}\cdot(Q_U+Q_A)}{P\cdot(Q_U+Q_A)}=\frac{\hat{P}}{P}<\frac{\hat{P}}{\tilde{P}}<1\,.$$
- Total subsidy is $(P-\hat{P})\cdot(Q_U+Q_A)$ €. A part of the subsidy, larger than $(MC-\hat{P})\cdot(Q_U+Q_A)$ €, would probably be transferred to water producers in order for them to have a positive surplus. The remainder can be used to recover capital expenditures for system expansion, upgrades, equipment replacement, and more.

A comment about non-optimal allocation and its associated dead-weight loss

It should be clearly stated that opportunity costs, like any other concept of costs, have a meaning only when water allocation is optimal, like in Figure 12. If, for example, water allocation is arbitrary and the urban centre receives less water than Q_U, say $(Q_U - \Delta Q)$ and the farmers receive more water than Q_A, $(Q_A + \Delta Q)$, (see Figure 13a and Figure 13b), the opportunity costs can not be adequately quantified since one can increase the total value of production in the two-users economy without adding water above $\bar{Q}$, just by transferring water from the farmers to the urban centre.

The welfare loss associated with non optimal allocation of $\bar{Q}$ is illustrated in Figure 13a and Figure 13b. First, note that the total **social water surplus** under optimal allocation (i.e., Q_U for the urban centre and Q_A for the farmers) is given by the sum of the areas [abp] in Figure 13a, and [efP] in Figure 13b. Transferring ΔQ from the urban centre to the farmers reduces total social water surplus: (i) the surplus associated with the urban consumption reduces by the shaded area [cbd] in Figure 13a, and (ii) the surplus associated with agricultural consumption reduces by the shaded area [fgh] in Figure 13b. The first source of surplus reduction, resulting from a **cut** of ΔQ in the quota allocated to the urban centre, is intuitive and needs no further explanation. The second source of reduction, resulting from an

increase of ΔQ in the quota allocated to the farmers is less obvious. Let's recall that the demand curve D_A represents the marginal value of water (value of marginal product) allocated for farmers, which is the net revenue generated by the last (marginal) unit of water. Water input larger than this last unit generated smaller and smaller revenues (this is a consequence of the diminishing marginal productivity of water). We can note now from Figure 13b that each cubic meter above Q_A allocated to farmers generates revenues which are **lower** than its opportunity cost (P). The marginal loss associated with each unit of water input above Q_A is equal to P minus the marginal revenues generated by this unit. Thus, the total loss associated with the addition of ΔQ for the farmers is the shaded area in Figure 13b.

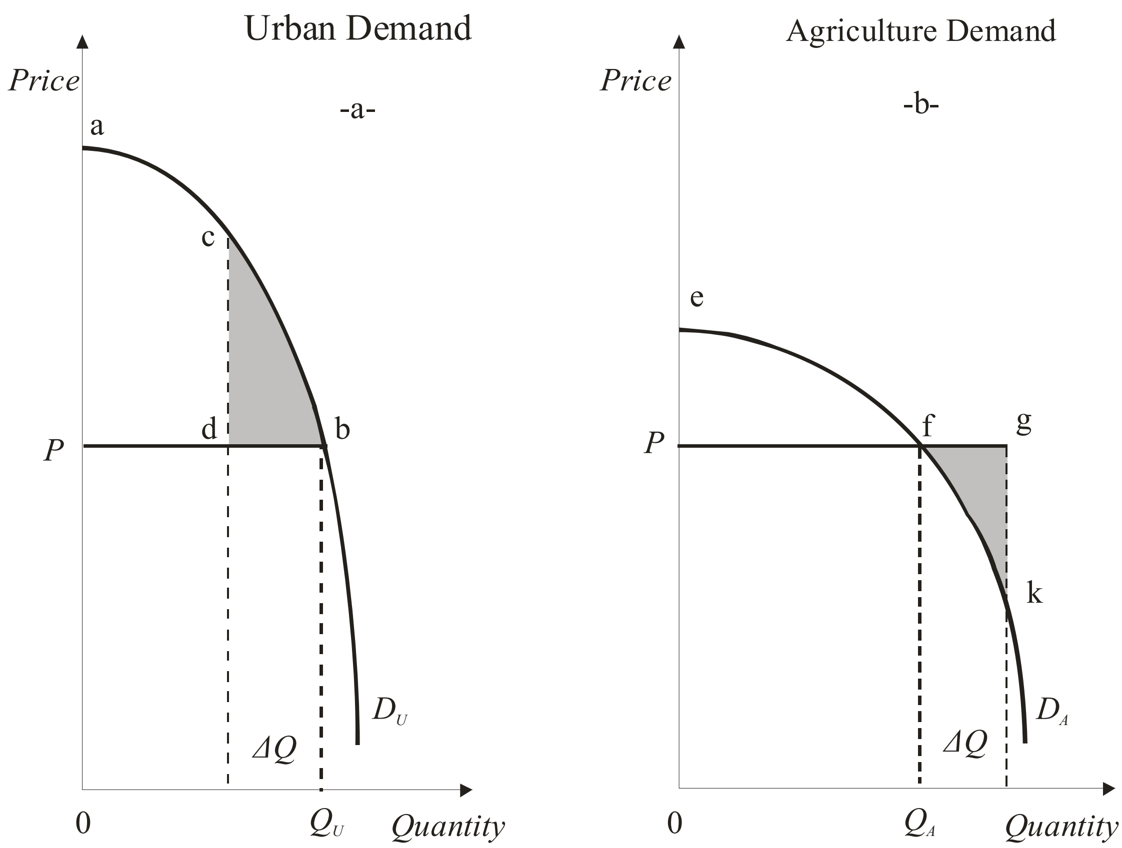

Figure 13: Non optimal allocation and dead-weight loss

The **total welfare reduction** in the (two-user) economy, resulting from non-optimal allocation is known in the economic literature as a **dead-weight loss** and is equal in our example to the sum of the areas [cbd] + [fgh] (Figure 13a,b). In practice, water allocation is commonly not optimal and generates dead-weight loss which can be quite significant. In principal, this loss (the social cost of an inefficient allocation) should be regarded as one of the cost components associated with the utilisation of scarce water resources. By ignoring the dead-weight loss we implicitly assume that the allocation of the limited water resource is efficient (i.e., equates the marginal values of water across all users).

We close this section by presenting the efficient allocation of $\bar{Q}$ when the urban centre and the agricultural fields are located at different distances from the water resource (the lake).

Water supplied at different locations

Let us assume now that the urban centre is very close to the water resource (as was implicitly assumed before) while the agricultural fields are located far away from it and the cost of transporting water from the abstraction point to the fields is t €/m^3. The direct marginal costs of extracting, treating and delivering the water from the lake to the urban centre and the agricultural users (including labour and capital) are equal to $MC_U = MC$ and $MC_A = MC + t$ €$/m^3$, respectively.

Price-based allocation of $\bar{Q}$ requires different water prices for the urban and for the agricultural users, denoted by P_U and P_A, respectively. It can be shown via straightforward mathematical analysis[1], that under optimal allocation:

(i) $P_U = MVW^U(\hat{Q}_U)$; $P_A = MVW^A(\hat{Q}_A)$; where $\bar{Q} = \hat{Q}_U + \hat{Q}_A$;
and the scarcity rent in this case, denoted by λ^* is given by:
(ii) $\lambda^* = P_U - MC = P_A - t - MC \Rightarrow P_A = P_U + t$.

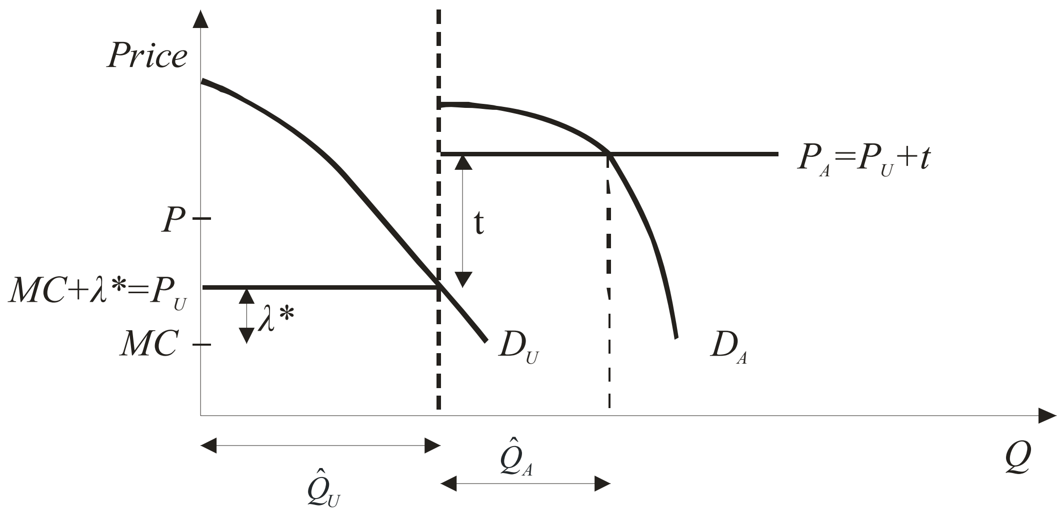

Figure 14: Water supplied at different locations

It is useful to present the intuition of this result. To see that $P_A = P_U + t$ must hold when water allocation is optimal, let's start by assuming that $P_A > P_U + t$ at the optimal solution. Then transferring one more cubic meter of water from the urban centre to the farmers would have two effects. Firstly, net benefits at the urban centre would decline by $P_U = MVW^U(\hat{Q}_U)$. Secondly, 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 to agricultural production would produce an increase in net benefits of $P_A = MVW^A(\hat{Q}_A)$. Since, by assumption, $P_A > P_U + t$, the proposed transfer would increase net benefits; hence, we cannot be at an optimum when this inequality holds. Similarly, let us assume that $P_A < P_U + t$. Then too much water has been transferred from the urban centre to agricultural use and transferring one less cubic meter would increase net benefits. Hence, again, we cannot be at an optimum.

[1] Available by request from the author.

Comparison of the market cleaning prices under this case with the one of the previous examples (P) (i.e., when both, the urban centre and the farmers are located at the same distance from the lake) and comparing λ^* with λ yields:

$$\text{(iii)}\ P_U < P \rightarrow \hat{Q}_U > Q_U;\ P_A > P \rightarrow \hat{Q}_A < Q_A;\ \lambda^* < \lambda.$$

⇒ Comment: in this example, the opportunity costs of water at the urban centre, $P_U = MC + \lambda^*$ and at the agricultural area $P_A = MC + \lambda^* + t$ differ. One may ask how come the opportunity costs associated with the same type of water are not identical 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 holds 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.

Estimation of Environmental Costs

Why society must take environmental costs into account?

An **Environmental Cost** can be defined as the cost that a "society" will have to pay in the future (soon or later) because of the impacts caused on the environment by economic activities, products or services. Usually, this type of cost is **external.** This means that the cost is equal to the monetary value attributed to the reduction of an advantage or to a damage undergone by society because of a deterioration of the environmental quality, and that it has **not** been taken into account in a market operation.

According to the neo-classical theory, it is essential to reintegrate (internalise) this monetary value in market operations. There are different justifications for this assumption in the case of water resource degradation:

- If this cost is underestimated or disregarded, then the future users of the resource are the ones that will have to pay for the measures needed for its restoration, while degradation has been caused by the current users;
- The polluter doesn't pay for the damage he caused;
- If this cost is underestimated or disregarded, the current users are not encouraged in protecting the resource.

For these reasons, the European Water Framework Directive underlines the following principle: "The use of economic instruments by Member States may be appropriate as part of a programme of measures. The principle of recovery of the costs of water services, **including environmental and resource costs** associated with damage or negative impact on the aquatic environment should be taken into account in accordance with, in particular, the polluter-pays principle".

Different methods are developed and applied to place monetary values on environmental services. Those are presented in the following section. The selection of the appropriate one depends on the nature of the environmental issue, the data and resources available for the analysis.

Available methods for estimating environmental costs

In the last few decades economists have devoted significant professional attention to developing and applying 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 monetary terms (Shabman and Stephenson, 2000). For this purpose different methods can be applied:

- *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 negatively the quality of shellfish and hence its price in the market.
- *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 and especially in those where mitigation costs are an underestimation of the true environmental cost. On the contrary, mitigation measures might not be cost-effective and can lead to 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 (internalised environmental 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 meeting the WFD objectives. These costs can be a good estimate of what society is willing to forego.
- *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 behaviour 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 behaviour*: This method derives from observations of how people change defensive behaviour – adapt coping mechanisms – in response to changes in environmental quality. Defensive behaviour 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, e.g. 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, patterns of travel to particular sites can be used to analyze how an individual values the site and, for example, the water quality at the site. Reductions in trips to a river due to deterioration of water quality and associated changes in expenditures reveal the cost of this deterioration.

- *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 markets. For a 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 values. Methods include contingent valuation and contingent ranking. It is also possible to construct an experimental market where money is exchanged, e.g. using simulated market models. In the questionnaire, it is also possible to ask respondents how much they would pay for avoiding an environmental cost or how much they value a given environmental benefit.

- *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 the respondent adequately understands the environmental change that is being valued in the survey.
- *Use of value transfer* (an alternative option to the 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 area under consideration. As a result, a set of data 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, amongst other problems, it is important to note that since benefits have been estimated in a different context they are unlikely to be as accurate as a primary research. A step-wise approach should be developed in order to ensure that the transfer of values derived in other contexts can minimise the potential for estimation errors.

A discussion on methods for the estimation of environmental costs

As mentioned before, economic valuation, as an empirical exercise, is based on the argument that choices individuals make in market exchanges provide the data that analysts can use to translate people's preference into money terms. The logic of the argument is straightforward. In market exchange, income is sacrificed (a price is paid) in order to secure a good or service. By arguing that preferences guide market choices, analysts conclude that the money value of a good or service is at least equal to the amount of income a person spends to obtain that good or service. Thus, market prices constitute the raw data for preference measurement. The often-explicit premises of this revealed choice framework are (Randal and Peterson, 1984):

- That individuals know their preference for goods and services (states of the environment) before being confronted with choice,
- That people are willing to pay to satisfy those preferences, and
- That whatever an individual chooses is in the best interest of that individual.

It is the benefit-cost analysts' responsibility to measure those preferences into monetary terms (Randal, 1999).

Not all economists support the expanded use of non-market valuation calculation in policy making. This criticism is supported by the concerns on non-market valuation expressed by psychologists, philosophers, and political scientists who are familiar with the valuation research program. Generally speaking, criticism focuses on 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 and Stephenson, 2000).

Free market environmentalist critique

Free Market Environmentalists have a particular understanding of the social purpose of market exchange. Under this concept 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 when making those 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 a 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 technologies and resources (Hayak, 1948). The function of market prices is to stimulate change, co-ordinate decentralised adaptation to that change, synthesise disperse and fragmented knowledge and promote individual incentives, discretion, and responsibilities.

In the Free Market Environmentalist economist's view, a fixed set of preferences cannot be found through examination of market prices, because preferences and other determinants of prices constantly fluctuate (Shabman and Stephenson, 2000).

The work of psychologists supports the Free Market Environmentalist view that preferences are dictated by choice circumstances and are context- and time-sensitive. Relevant research has found that in many choice situations, people do not retrieve preferences from previously formed preferences, but preferences are constructed at the time and in the context of the choice opportunity (Slovic et al., 1977; Tversky et al., 1988; Gregory et al., 1993; Schkade and Payne, 1994; Schkade 1995).

From this point of view, non-market valuation can be compared with the centralised economic planning of the former Soviet Block countries. For example, the inability of Soviet planners to calculate prices that would determine how much wheat or how many nails to produce, was not that different from a government analyst attempting to determine water quality standards, the allocation of water between municipalities and agriculture, and how many acres of timber to 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 estimating values.

Institutional economics critique

"To believe that markets determine value is to believe that milk comes from plastic bottles" (Bromley, 1985).

Institutional economists 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. However, for institutional economists the conditions establishing market exchange should themselves be subject to social debate, scrutiny, and policy change (Bromley, 1985). 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).

Under this perspective, institutional economists argue that non-market valuation inappropriately elevates the preferences of current individuals and particularly of 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 non-market 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 where preferences are a subject of investigation and social debate (Shabman and Stephenson, 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) suggests 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 unwarranted 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 not be resolved.".

Practices of the French Agences de l'Eau and proposal of a methodology

In the following chapter we will present the French Agences de l'Eau charge system. The different charges in use in France are proportional to the pollution or to the impact on the resource quantity. The aim of this chapter is to describe this system, explain advantages and disadvantages of the chosen options, and make some propositions aiming to adapt these different simple formulas used in this system for the calculation of the environmental cost of water resource degradation (in terms of quantity and quality).

Pollution charges

Table 4 outlines the general principle for the estimation of pollution charges. Pollution charges for each quality parameter depend on the quantities rejected by the different users during a normal day of the month when the maximal discharge occurs **(charge base).** Charge bases can be given either by monitoring measurements or estimated[2].

[2] In France estimation performed with the help of tables given by a Decree.

Table 4: General principles for pollution charge estimation

Parameters	Unit	Charge base[3]	Charge
Suspended Matter	kg /day	A	$A \times Coef_{Susp.\ Mat} \times R_{Susp.\ Mat}$
Oxidisable Matter[4]	kg /day	B	$B \times Coef_{Oxyd.\ Mat} \times R_{Oxyd.\ Mat}$
Dissolved salts	mho/cm	C	$C \times Coef_{Dis.\ Salt} \times R_{Dis.\ Sal}$
Inhibiting matter	K.equitox[5] / days	D	$D \times Coef_{Inhib.\ Mat} \times R_{Inhib.\ Mat}$
Reduced Nitrogen (organic + ammoniac)	kg/day	E	$E \times Coef_{Red.Nitr.} \times R_{Red.Nitr.}$
Oxidised Nitrogen (nitrate + nitrites)	kg /day	F	$F \times Coef_{Oxid.Nitr.} \times R_{Oxid.Nitr}$
Total Phosphorus (organic + mineral)	kg /day	G	$G \times Coef_{Tot.\ P} \times R_{Tot.\ P}$
Adsorbable Organic Halogens (AOX)	kg /day	H	$H \times Coef_{AOX} \times R_{AOX}$
METOX[6]	kg /day	I	$I \times Coef_{METOX} \times R_{METOX}$
Microbiologic elements [7]	Number	J	$J \times Coef_{Micr.\ Elements} \times R_{Micr.\ Elements}$
Total			$\sum_{Aa}^{Jj} (Y \times Coef_x \times R_x)$

The coefficient (Coef) used for the estimation of the charge for each quality parameter is a sum of different coefficients.

- Zone coefficients ($Coef_{zone}$), used to take into account the sensibility of the aquatic ecosystem. Table 5, Table 6 and Table 7 present sample coefficient values from the Rhin-Meuse, and Seine-Normandie and Loire-Bretagne basins.

[3] There is an abatement of charge base (Abt_{cb}) for bovine in pasture. This abatement should be calculated with the following formula: $Abt_{cb} = A \times \frac{N}{12}$ where:
A = Quantity rejected during a normal day of the month when occurs the maximal discharge (charge base)
N = time spent in pasture (in months)

[4] Oxidisable Matter: $Oxidisable\ Matter = \frac{COD + 2 \times BOD}{3}$

[5] 1 Equitox is the quantity of toxicity which, in 1 m^3 of water, immobilises, after 24 hours, 50 % of the daphnia's (fresh water microphone-shellfish).

[6] $METOX = (10 \times As) + (50 \times Cd) + (1 \times Cr) + (5 \times Cu) + (50 \times Hg) + (5 \times Ni) + (10 \times Pb) + (1 \times Zn)$, where each heavy metal load is expressed in kg/day.

[7] Microbiologic Elements = $ME = V \times \left(\frac{C}{106}\right) \times 0.3$ where:
C= number of Escherichia coli and Enterococci/m^3
V= volume of the effluent (m^3)
Until now charges for microbiologic elements had not applied for domestic use.

Table 5: Zone coefficients in Rhin-Meuse Basin

Level of sensibility	Parameter	$Coef_{zone}$
Level 0 (less fragile)	Suspended Matter; Oxidisable Matter; Dissolved salts; Inhibiting matter; Reduced Nitrogen (organic + ammoniac); Oxidised Nitrogen (nitrate + nitrites); Adsorbable Organic Halogens (AOX); METOX	1.00
Level 1	Suspended Matter; Oxidisable Matter; Dissolved salts; Inhibiting matter; Reduced Nitrogen (organic + ammoniac); Oxidised Nitrogen (nitrate + nitrites); Adsorbable Organic Halogens (AOX); METOX	1.70
Level 2 (most fragile)	Suspended Matter; Oxidisable Matter; Dissolved salts; Inhibiting matter; Reduced Nitrogen (organic + ammoniac); Oxidised Nitrogen (nitrate + nitrites); Adsorbable Organic Halogens (AOX); METOX	1.80

Table 6: Zone coefficients in Seine-Normandie Basin

Level of sensibility	Suspended Matter	Oxidisable Matter	Reduced Nitrogen; Total Phosphorus	Adsorbable Organic Halogens (AOX); METOX; Inhibiting matter
Level 1 (most fragile)	1.25	1.25	1.25	1.25
Level 2	1.25	1.15	1.15	1.15
Level 3 (less fragile)	1.25	1.00	1.00	1.00

Table 7: Zone coefficients in Loire-Bretagne Basin

Level of sensibility	Suspended Matter	Oxidisable Matter	Inhibiting matter	Reduced Nitrogen (organic + ammoniac)	Total Phosphorus (organic + mineral)	METOX
Areas with no particular action	1.00	1.00	1.00	1.00	1.00	1.00
Littoral areas with reinforced actions	1.15	1.15	1.00	1.00	1.00	1.00
Other areas with reinforced actions	1.00	1.15	1.00	1.15	1.00	1.00

- Phosphorus coefficients ($Coef_{Phos}$), used if pollution occurs in an area where discharges must be treated for phosphorus. The practice is applicable in some basins, as for example in n Rhin-Meuse basin where $Coef_{Phos}$= 4. Values are normally included in the overall zone coefficient.
- Groundwater coefficient ($Coef_{GW}$), used if there is a discharge to a groundwater body. Table 8 presents the relevant groundwater coefficients for the Rhin-Meuse basin.

Table 8: Rhin-Meuse basin groundwater coefficients

Parameter	$Coef_{GW}$
Suspended Matter	0
Oxidisable Matter; Dissolved salts; Reduced Nitrogen (organic + ammoniac); Oxidised Nitrogen (nitrate + nitrites); Total Phosphorus (organic + mineral)	3
Inhibiting matter; Adsorbable Organic Halogens (AOX); METOX	10

- Sewage coefficient ($Coef_{col}$), used only for domestic wastewater, in order to finance sewage collection works. The coefficient is not environmentally justified. There is
  - $Coef_{col}$ = 2.4 in the Rhin-Meuse basin in 2003
  - $Coef_{col}$ = 2.4 in the Seine-Normandie basin in 2003
  - $Coef_{col}$ = 2.8 in the Loire-Bretagne basin in 2003 for municipalities > 1000 inhabitants
  - $Coef_{col}$ = 1.4 in the Loire-Bretagne basin in 2003 for municipalities ≤ 1000 inhabitants

Table 9 summarises the charge rates R_x for three basins according to the quality parameter monitored by the system.

Table 9: Charge rates for Rhin-Meuse, Adour-Garonne and Seine-Normandie basins (2003)

Parameters	Unit	Charge rate (€/Unit)		
		Rhin-Meuse	Adour-Garonne	Seine Normandie
Suspended Matter	kg/day	23.03	30.66	26.76
Oxidisable Matter	kg /day	46.6	57.17	63.13
Dissolved salts	mho/cm X m^3	26.08	/	559.16
Inhibiting matter	K.equitox / day	748.32	1293.52	1524.50
Reduced Nitrogen (organic + ammoniac)	kg /day	31.60	56.52	66.81
Oxidised Nitrogen (nitrate + nitrites)	kg /day	15.75	/	/
Total Phosphorus (organic + mineral)	kg /day	48.90	75.9	57.06
Adsorbable Organic Halogens (AOX)	kg /day	476.84	160.51	410.98
METOX	kg /day	127.16	133.42	410.98

The estimation of charge bases (generated loads), which form the basis for the computation of pollution costs, is presented in the following paragraphs.

Domestic use

Domestic users pay for pollution charges through the water bill. There is a bonus system to take into account the existence and successful operation of wastewater treatment plants, which is paid back to the municipality.

In France charge estimation is based on the assumption that one inhabitant produces the following loads:

- Suspended Matter: 90 g/day;
- Oxidisable Matter: 57 g/day;
- Inhibiting matter: 0.2 Equitox/day;
- Reduced Nitrogen: 15 g/day;
- Phosphorus: 4 g/day;
- Adsorbable Organic Halogens (AOX): 0.05 g/day;
- METOX: 0.23 g/day.

The population taken into account to calculate the total pollution generated by a municipality is the permanent population + 0.4 × the seasonal population. For domestic uses there is another coefficient that is taken into account called Agglomeration Coefficient, which varies according to the size of the agglomeration (Table 10). This coefficient is not environmentally justified.

Table 10: Agglomeration Coefficient values

Number of agglomerated inhabitants	Agglomeration Coefficient
[0-500]	0.50
[500-2.000]	0.75
[2.000-10.000]	1.00
[10.000-50.000]	1.10
> 50.000	1.20
Paris + suburb	1.40
Municipality without water distribution network	0.00

Pollution generated by other economic activities (industries, agriculture…)

The estimation of loads produced by other economic activities is based on a Decree that specifies the amount of generated loads according to the type of the activity and a characteristic measure that provides its size and importance e.g. the number of employees or the production volume (Table 11).

Table 11: Estimation of pollution charges for economic activities

Economic activity	Characteristic measure of the pollutant activity	Generated Pollution									
		Suspended Matter (g)	Oxidisable Matter (g)	Inhibiting matter (equitox)	Dissolved salts (10^3 mho/cm X m^3)	Reduced Nitrogen (g)	Oxidised Nitrogen (g)	Total Phosphorus (g)	AOX (g)	METOX (g)	Microbiologic Elements (um)
Lime and cement production	1 employed person	400	100	/	/	7	/	2	/	/	/
…	…	…	…	…	…	…	…	…	…	…	…

A methodology proposal for the estimation of pollution costs

A methodology proposal for the estimation of environmental benefits from wastewater treatment

The methodology proposed is based on a cost-based valuation method that takes into account the costs of measures needed to prevent environmental damages up to a certain point, such as those that meet the Directives' Objectives. The assumption made is that these costs can be a good estimate of what society is willing to forego. The principle used to estimate pollution environmental costs is presented in Table 12.

Table 12: Proposed methodology for the estimation of pollution environmental costs

Quality Parameter	Unit	Quantity rejected during a normal day of the month base when occurs the maximal discharge. Given by monitoring or by estimation	Incurred environmental cost
M1	kg/day	A	$A \times Coef_{M1} \times R_{M1}$
M2	kg /day	B	$B \times Coef_{M2} \times R_{M2}$
M3	kg /day	C	$C \times Coef_{M3} \times R_{M3}$
M4	kg /day	D	$D \times Coef_{M4} \times R_{M4}$
M5	kg /day	E	$E \times Coef_{M5} \times R_{M5}$
M6	kg /day	F	$F \times Coef_{M6} \times R_{M6}$
M7	kg /day	G	$G \times Coef_{M7} \times R_{M7}$
…	…	…	…
MX		Z	$Z \times Coef_{MX} \times R_{MX}$
Total			$\sum_{Aa}^{Zz} (Y \times Coef_x \times R_x)$

The coefficient presented in Table 12 ($Coef_x$) should take into account the sensibility of the aquatic ecosystem only. Other coefficient, not relevant with environmental impact should be excluded from the analysis. The term $Z \times coef_{MX} \times R_{MX}$ should be equal to the cost (investment + maintenance + operation costs) of treatment for each quality parameter (waste water and water production) according to the principle of a cost-based valuation method.

According to this assumption, the following equation should be used:

Costs of preventive and / or mitigation measures ≅ Environmental costs

It should be noted that mitigation may not be possible in all cases, for example, in cases where those costs could be an underestimation of true environmental cost. On the other hand, a mitigation measure might not be cost-effective and these costs might result in an over-estimation of environmental costs.

Environmental benefit produced by a wastewater treatment plant

The general methodology applied by the French Agences de l' Eau for estimating the environmental benefit produced from a wastewater treatment plant is presented in

Table 13. Values for coefficients (*Coeffx*) and charges *Rx* are similar to those presented in Table 4.

Table 13: General methodology for the estimation of environmental benefits from wastewater treatment

Parameters	Unit	Charge base	Charge calculation method	Bonus calculation method
Suspended Matter	kg/day	A	$A \times Coef_{Susp.\ Mat} \times R_{Susp.\ Mat}$	$A \times Coef_{Susp.\ Mat} \times R_{Susp.\ Mat} \times BAC$ $for\ Suspended\ Matter$
Oxidisable Matter	kg /day	B	$B \times Coef_{Oxyd.\ Mat} \times R_{Oxyd.\ Mat}$	$B \times Coef_{Oxyd.\ Mat} \times R_{Oxyd.\ Mat} \times BAC$ $for\ Oxydable\ Matter$
Dissolved salts	Conductivity (mho/cm) => weight of rejected salt = mho/cm X m^3	C	$C \times Coef_{Dis.\ Salt} \times R_{Dis.\ Salt}$	$C \times Coef_{Dis.\ Salt} \times R_{Dis.\ Sal} \times BAC$ $for\ Dissolved\ Salt$
Inhibiting matter	K.equitox / days	D	$D \times Coef_{Inhib.\ Mat} \times R_{Inhib.\ Mat}$	$D \times Coef_{Inhib.\ Mat} \times R_{Inhib.\ Mat} \times BAC$ $for\ Inhibiting\ Matter$
Reduced Nitrogen (organic + ammoniac)	kg /day	E	$E \times Coef_{Red.Nitr.} \times R_{Red.Nitr.}$	$E \times Coef_{Red.Nitr.} \times R_{Red.Nitr.} \times BAC$ $for\ Reduced\ Nitrogen$
Oxidised Nitrogen (nitrate + nitrites)	kg /day	F	$F \times Coef_{Oxid.Nitr.} \times R_{Oxid.Nitr}$	$F \times Coef_{Oxid.Nitr.} \times R_{Oxid.Nitr} \times BAC$ $for\ Oxidised\ Nitrogen$
Total Phosphorus (organic + mineral)	kg /day	G	$G \times Coef_{Tot.\ P} \times R_{Tot.\ P}$	$G \times Coef_{Tot.\ P} \times R_{Tot.\ P} \times BAC$ $for\ Total\ Phosphor$
Adsorbable Organic Halogens (AOX)	kg /day	H	$H \times Coef_{AOX} \times R_{AOX}$	$H \times Coef_{AOX} \times R_{AOX} \times BAC$ $for\ AOX$
METOX	kg /day	I	$I \times Coef_{METOX} \times R_{METOX}$	$I \times Coef_{METOX} \times R_{METOX} \times BAC$ $for\ METOX$
Microbiologic Elements	Number	J	$J \times Coef_{Micr.\ El.} \times R_{Micr.\ El.}$	$J \times Coef_{Micr.\ El.} \times R_{Micr.\ El.} \times BAC$ $for\ Microbiologic\ Elements$
Total			$\sum_{Aa}^{Jj} (Y \times Coef_x \times R_x)$	$\sum_{Aa}^{Jj} (Y \times Coef_x \times R_x \times BAC\ for\ x)$

BAC = Bonus annual coefficient

Bonus annual coefficients

In most cases bonus annual coefficients for each quality parameter are set equal to the respective pollution abatement coefficients (Table 14). The latter reflect an overall appreciation of the effectiveness of the wastewater treatment process. Sewage treatment can also be considered in the calculation.

Table 14: Pollution abatement coefficients according to wastewater treatment processes

Wastewater treatment process type	Parameters	Pollution Abatement Coefficient				
		Bad	Mediocre	Average	Good	Very good
Settling basin.	Suspended Matter	0	0.3	0.5	0.7	0.90
	Oxidisable Matter	0	0	0	0	0
	Inhibiting matter	0	0	0	0	0
	Reduced Nitrogen	0	0	0	0	0.1
	Oxidised Nitrogen	0	0	0	0	0
	Total Phosphorus	0	0	0.1	0.2	0.3
	AOX	0	0	0	0	0
	METOX	0	0	0	0	0
Physico-chemical treatment (without detoxification).	Suspended Matter	0	0.4	0.7	0.9	0.95
	Oxidisable Matter	0	0.2	0.4	0.5	0.6
	Inhibiting matter	0	0.1	0.2	0.4	0.8
	Reduced Nitrogen	0	0	0	0.1	0.2
	Oxidised Nitrogen	0	0	0	0	0
	Total Phosphorus	0	0.4	0.8	0.9	0.95
	AOX	0	0	0	0	0
	METOX	0	0	0.5	0.6	0.7
Physical water treatment.	Suspended Matter	0	0.8	0.9	0.95	0.99
	Oxidisable Matter	0	0.3	0.6	0.8	0.9
	Inhibiting matter	0	0.1	0.2	0.4	0.8
	Reduced Nitrogen	0	0	0.3	0.6	0.8
	Oxidised Nitrogen	0	0.1	0.3	0.6	0.8
	Total Phosphorus	0	0.2	0.5	0.7	0.9
	AOX	0	0	0.3	0.6	0.8
	METOX	0	0.1	0.3	0.6	0.9
Incineration unit	Suspended Matter	Monitoring only				
	Oxidisable Matter					
	Inhibiting matter					
	Reduced Nitrogen					

Wastewater treatment process type	**Parameters**	**Pollution Abatement Coefficient**				
		Bad	**Mediocre**	**Average**	**Good**	**Very good**
	Oxidised Nitrogen					
	Total Phosphorus					
	AOX					
	METOX					
Biological wastewater treatment unit: **Biological wastewater treatment unit: common coefficients (for all types of units).**	Suspended Matter					
	Oxidisable Matter	0	0.4	0.7	0.9	0.95
	Inhibiting matter	0	0.3	0.7	0.8	0.9
	Reduced Nitrogen	0	0	0.3	0.5	0.6
	Oxidised Nitrogen	0	0	0.3	0.5	0.6
	Total Phosphorus	0	0	0.5	0.6	0.7
	AOX					
	METOX					
Biological wastewater treatment unit: specific coefficients:	Reduced Nitrogen	0	0.1	0.2	0.3	0.4
Biological wastewater treatment unit	Oxidised Nitrogen	0	0	0	0	0
without nitrate treatment and without phosphorus treatment	Total Phosphorus	0	0.1	0.2	0.3	0.4
Biological wastewater treatment unit with nitrification but no denitrification step.	Reduced Nitrogen	0	0.3	0.6	0.8	0.9
	Oxidised Nitrogen	0	0	0	0	0
	Total Phosphorus	0	0.1	0.2	0.3	0.4
Biological wastewater treatment unit with nitrification-denitrification step.	Reduced Nitrogen	0	0.3	0.6	0.8	0.9
	Oxidised Nitrogen	0	0.3	0.6	0.8	0.9
	Total Phosphorus	0	0.1	0.2	0.3	0.4
Biological wastewater treatment with biological phosphate removal step.	Reduced Nitrogen	0	0.1	0.2	0.3	0.4
	Oxidised Nitrogen	0	0	0	0	0
	Total Phosphorus	0	0.3	0.4	0.5	0.6
Biological wastewater treatment unit with physico-chemical phosphate removal step.	Reduced Nitrogen	0	0.1	0.2	0.3	0.4
	Oxidised Nitrogen	0	0	0	0	0
	Total Phosphorus	0	0.4	0.8	0.9	0.95
Biological wastewater treatment unit with nitrification but no denitrification step + biological phosphate removal step	Reduced Nitrogen	0	0.3	0.6	0.8	0.9
	Oxidised Nitrogen	0	0	0	0	0
	Total Phosphorus	0	0.3	0.4	0.5	0.6
Biological wastewater treatment unit with nitrification but no denitrification + physico-	Reduced Nitrogen	0	0.3	0.6	0.8	0.9
	Oxidised Nitrogen	0	0	0	0	0

Wastewater treatment process type	Parameters	Pollution Abatement Coefficient				
		Bad	Mediocre	Average	Good	Very good
chemical phosphate removal step	Total Phosphorus	0	0.4	0.8	0.9	0.95
Biological wastewater treatment unit with nitrification-denitrification + biological phosphate removal step.	Reduced Nitrogen	0	0.3	0.6	0.8	0.9
	Oxidised Nitrogen	0	0.3	0.6	0.8	0.9
	Total Phosphorus	0	0.3	0.4	0.5	0.6
Biological wastewater treatment unit with nitrification-denitrification + physico-chemical phosphate removal step	Reduced Nitrogen	0	0.3	0.6	0.8	0.9
	Oxidised Nitrogen	0	0.3	0.6	0.8	0.9
	Total Phosphorus	0	0.4	0.8	0.9	0.95
Detoxification unit for metal finishing factory **In situ treatment**	MES	0	0.5	0.7	0.9	0.95
	MO	0	0	0	0	0
	MI	0	0.5	0.7	0.9	0.95
	NR	0	0	0	0	0
	NO	0	0	0	0	0
	AOX	0	0	0	0	0
	METOX	0	0.5	0.7	0.9	0.95
Without physico-chemical phosphate removal step.	P	0	0.1	0.2	0.3	0.6
With physico-chemical phosphate removal step.	P	0	0.4	0.8	0.9	0.95
Spreading on soil used for vegetal production.	Suspended Matter	1	1	1	1	1
	Oxidisable Matter	0.4	0.6	0.8	0.9	0.95
	Inhibiting matter	0	0	0	0	1
	Reduced Nitrogen	0	0.4	0.6	0.8	0.9
	Oxidised Nitrogen	0	0.4	0.6	0.8	0.9
	Total Phosphorus	0	0.4	0.6	0.8	0.9
	AOX	0	0	0	0	1
	METOX	0	0	0	0	0

For the particular case of breading effluent spreading, bonus annual coefficients are presented in Table 15. Bonus annual coefficients in this case depend on the effectiveness of effluent recovery.

Table 15: Bonus annual coefficients for effluent spreading

	Parameter	Effectiveness of effluent recovery		
		Average (less than 80%)	**Good (more than 80%)**	**Very good (100%)**
Existence of planning document relating to spreading + Register relating to spreading + Pollution load / hectare is less than 3 UGBN[8] or if 3 UGBN < Pollution load / hectare < 5 UGBN and existence of a tool for fertilisation planning	Suspended Matter	1.00	1.00	1.00
	Oxidisable Matter	0.72	0.81	0.90
	Reduced Nitrogen	0.72	0.81	0.90
	Total Phosphorus	1.00	1.00	1.00
Existence of planning document relating to spreading + Register relating to spreading + 3 UGBN < Pollution load / hectare < 5 UGBN	Suspended Matter	1.00	1.00	1.00
	Oxidisable Matter	0.64	0.72	0.80
	Reduced Nitrogen	0.64	0.72	0.80
	Total Phosphorus	1.00	1.00	1.00
Existence of planning document relating to spreading Pollution load / hectare > 5 UGBN	Suspended Matter	1.00	1.00	1.00
	Oxidisable Matter	0.48	0.54	0.60
	Reduced Nitrogen	0.48	0.54	0.60
	Total Phosphorus	1.00	1.00	1.00

A methodology proposal for the estimation of environmental benefits from wastewater treatment

Environmental benefit estimation can be performed through the application of the following equation:

$$EB = \sum_{x=1}^{n} (Y_x' \times Coef_x \times R_x \times Bonus \quad annual \quad coefficient \quad for \quad x)$$

where:

Y_x' is the load for the relevant quality parameter that arrives at the entrance of the wastewater treatment plant each month; $Coeff_x$ and R_x are the coefficient and charge base respectively for the quality parameter as presented in paragraph 0, and $Bonus\,annual\,coefficient\;for\,x$ is the bonus annual coefficient for the wastewater treatment process referring to the x quality parameter.

[8] UGBN : Unité de Gros Bétail Azote : This is the common unit used in France for comparison of different kinds of breading. 1 UGBN is equal to 32 equivalent-inhabitants if we consider Oxidisable Matter and to 15 equivalent-inhabitant if we consider nitrogen.

Charges for water abstraction and consumption

Practices of the French Agences de l'Eau

Charges for abstraction and consumption of freshwater resources for the French Agences de l'Eau are estimated through the following equation:

$$\text{Charge}_{ic} = \left[\text{Abstraction during the reference period} \times \text{Abstraction charge base} \times (\text{Area coef} + \text{impact coef}) \times \text{use coef}\right] + \left[\text{Consumption during the reference period} \times \text{consumption charge base} \times (\text{Area coef} + \text{impact coef}) \times \text{use coef}\right]$$

The reference period depends on the type of the resource; normally for surface water it runs from 1st of May to the 30th of November and for groundwater from the 1st of April to the 31st of October. The volume consumed during the reference period is estimated on the basis of the abstraction during the same period, multiplied by a "net consumption coefficient" which takes, according to the type of water use, the values presented in Table 16.

Table 16: Net consumption coefficients (Agence de l'eau Loire-Bretagne)

Water use and restitution type	Net consumption coefficient
Direct restitution	0.07
Restitution after spreading	0.70
Steam production	1.00
Incorporation in a manufactured good	1.00
Supplement for closed circuit	1.00
Cooling in an open circuit	0.007

Area coefficients vary according to the localisation of the abstraction and the type of the resource. For example:

- Coef = 1 for surface-water abstraction in an area where there is no over exploitation of the resource;
- Coef = 1.8 for groundwater abstraction in an area where there is over exploitation of the resource.

Impact coefficients are applied in certain basins; when the following two conditions occur at the same time[9]:

- Abstraction > 100,000 m^3 during the reference period in no over exploited area
- $$\frac{\text{Average monthly flow at the abstraction point}}{\text{Natural flow of the driest month within five-yearly frequency at the abstraction point}} > 5\%$$

Use coefficients are not environmentally justified and depend on the type of consumptive use; for industrial uses they are equal to 1, for agricultural uses they are less than one while for domestic uses (drinking water), the use coefficient is approximately 3.

[9] Example taken from the Loire-Bretagne basin.

A methodology proposal for the estimation of abstraction environmental costs

The environmental cost for water abstraction and consumption could be estimated through the following equation:

$$\begin{aligned}Env\_cost_{ic} = &\left[\text{Abstraction during the reference period} \times \text{Abstracton charge base} \times \left(\text{Area coef} + \text{impact coef}\right)\right] + \\ &\left[\text{Consumption during the reference period} \times \text{consumption charge base} \times \left(\text{Area coef} + \text{impact coef}\right)\right]\end{aligned}$$

Reference periods are user-defined and vary according to the type of the resource. In principle, they should be chosen according to the local meteorological and hydrological conditions. Area coefficients vary according to the intake localization and the type of resource (chosen according to fragility of the area). An impact coefficient may be applied when the two following conditions occur at the same time:

- Abstraction > Y m^3 during the reference period in no over exploited area
- $\dfrac{\text{Average monthly flow at the abstraction point}}{\text{Natural flow of the driest month within five - yearly frequency at the abstraction point}} > X\%$

  where Y and X must be chosen according to local conditions.

Here again the following assumption should be used:

Costs of preventive and / or mitigation measures ≅ environmental costs

As proposed by Bernard Barraqué (2002), costs for sea or brackish-water desalination, depending on availability and transfer costs, could serve as the basis for the calculation of the costs of mitigation measures.

A Methodological Reflection

The most critical step in implementing the proposed methodology is the adaptation of different values used in France to other national and/or regional contexts.

For example, it is obvious that it is not appropriate to use without reflection the rule used in Loire-Bretagne basin for the estimation of environmental cost incurred from water abstraction and consumption:

"An impact coefficient is applied when the two following conditions occur at the same time:

- Abstraction > 100,000 m^3 during the reference period in no over exploited area
- $\dfrac{\text{Average monthly flow at the abstraction point}}{\text{Natural flow of the driest month within five - yearly frequency at the abstraction point}} > 5\%$"

Also, values provided for the estimation of pollution charges from the various economic activities (Table 11) should be adapted. The proposed methodology is effective in France, but can it be used for the estimation of pollution costs in other countries? Some industrial processes vary substantially between countries and regions and a value true in France, can be inappropriate for Greece, Portugal, Cyprus, Italy or Israel. Therefore, local experts should validate data presented in the different tables in this document and preferably use data obtained from monitoring.

Other Important Economic Indicators

Rate of costrRecovery

The total rate of recovery of costs is estimated as the sum of billing revenues from all water uses versus the total cost.

$$RCR = \frac{TotalRevenues}{TotalCost} \cdot 100\%$$

Including resource costs, RCR is defined as:

$$RCR_{Use} = \frac{TotalRevenues_{Use}}{DirectCost_{Use} + EnvironmentalCost_{Use} + OpportunityCost_{Use}} \cdot 100\%$$

Cost benefit balance

Cost-benefit balances are essential for the evaluation and the comparison of different options. A benefit-cost balance (BCBalance) is calculated using the following formula:

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

where the planning period begins in the current year, t = 0, and extends to some future planning horizon T (in years), B is the total benefit in the subscripted year (in monetary terms), C is the total cost in the subscripted year and *d* the discount rate in decimal form.

Chapter 3 MCDM in Water Resources

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.

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 15.

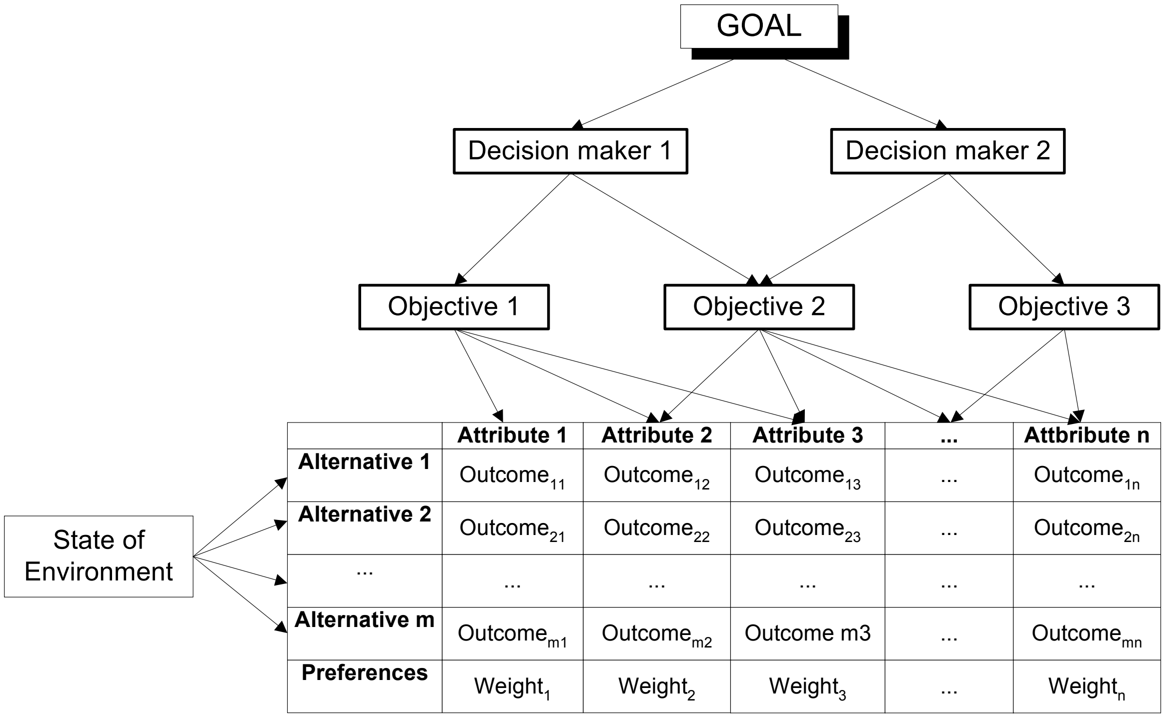

Figure 15: Framework for MCDM (Malczewski, 1999)

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 the 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

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

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

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

$$X = \left\{ x : x \in R^n, g_i\left(x\right) \le 0, x_j \ge 0 \ \right\}$$

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

$$z_q\left(x^{'}\right) > z_q\left(x\right)$$

for some $q \in \left\{1, 2, ..., p\right\}$ and $z_k\left(x^{'}\right) \ge z_k\left(x\right)$

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

$z\left(x^{'}\right) > z\left(x\right)$ 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

Weighting of Indicators

MCDM problems typically involve a number of criteria that are not equally important to the decision maker (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 DM. The weights typically sum up to one.

Formally, a set of weights is defined as follows:

$w = \left(w_1, w_2, ...w_j, ...w_n\right)$ 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 criterion
- 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 then 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.

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 16 (last diagram).

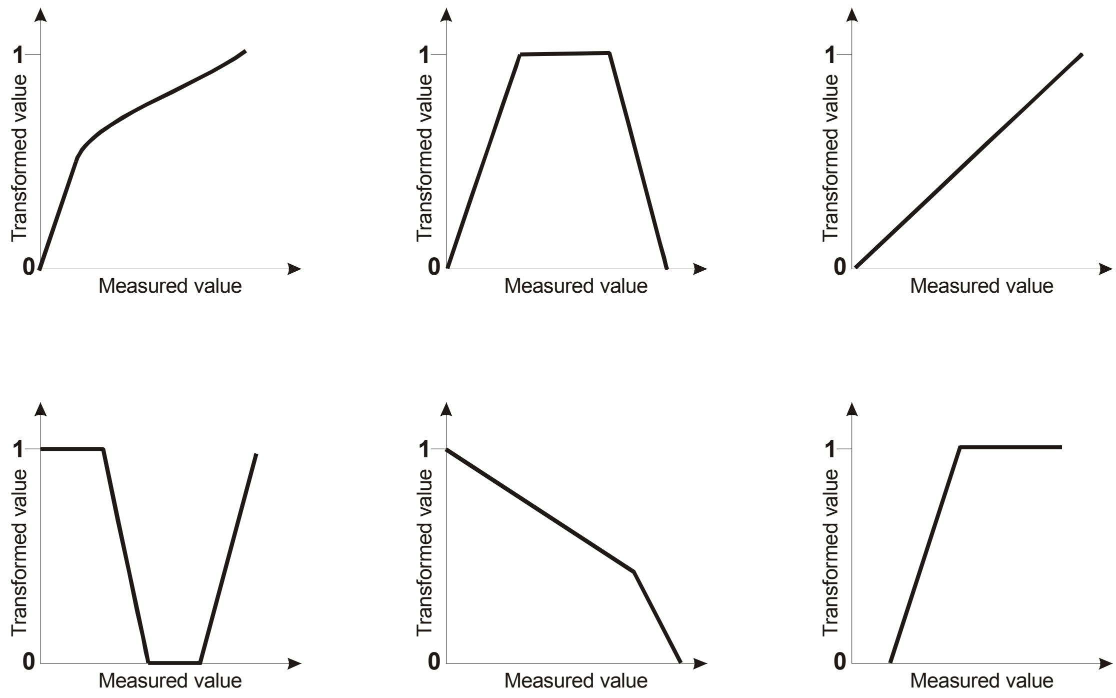

Figure 16: Typical transformation functions for indicators

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).

MCDM Techniques

Based on the prior articulation of preferences

Introduction

Methods in this category are 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 determining the preference structure may be difficult and is often time-consuming. Most of the approaches are particularly suitable for situations where the outcome is known 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, 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 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.

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 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", which 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.

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 a 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.

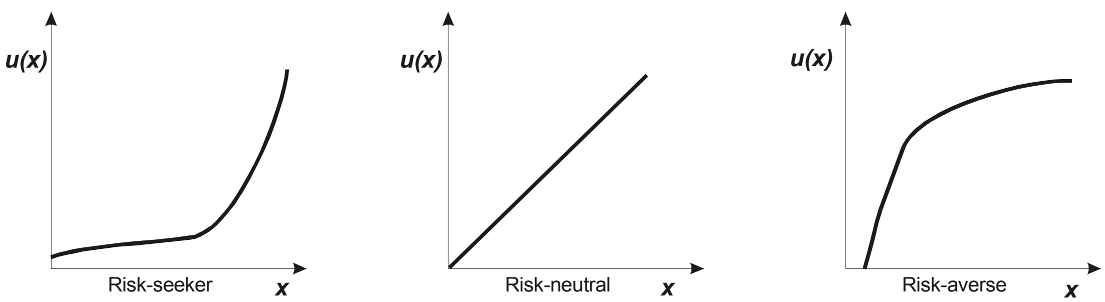

Figure 17:Basic shapes of utility functions

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 an 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:

$$c_{ij} \geq p \qquad \text{and}$$
$$d_{ij} \leq q$$

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

ELECTRE II

ELECTRE II is an extension of the ELECTRE I and has been developed by Roy in 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 hold:

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

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 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 ranks 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

$$x \in X$$

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

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

MCDM based on the progressive articulation of preferences

Introduction

Techniques based on the progressive articulation are characterised 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 co-ordinates (1,1) (Figure 18). 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:

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

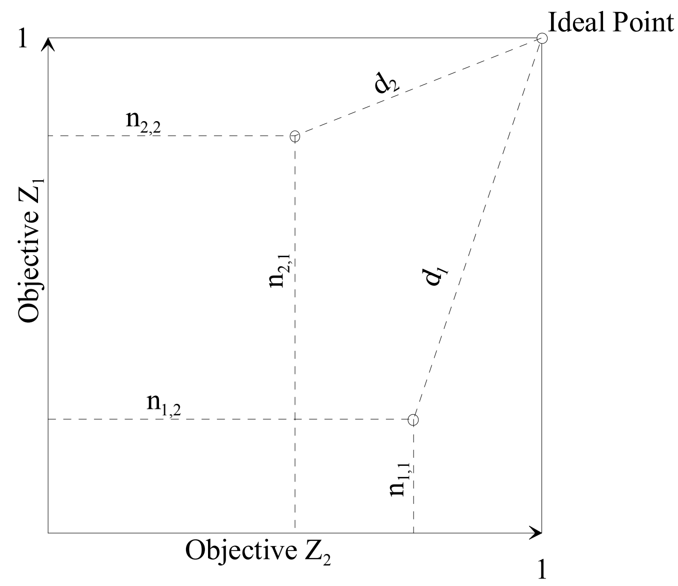

Figure 18: 2-D geometrical interpretation of distance-based methods

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=∞), 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 19.

The weight α_i reflects the DM`s 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.

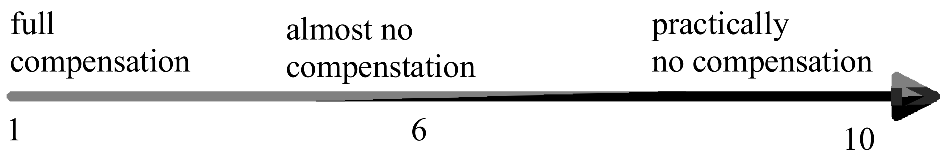

Figure 19: Sensitivity of the power factor p

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 Method (RGM) is particularly 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 a certain change in the alternatives has on the overall performance. The underlying theory however is not easy to explain to laymen.

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 Envelopment 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 DEA 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.

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 presented 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 refer 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 is 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, 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.

Fuzzy approaches have been proved to be very useful in water resources planning and have 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.

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.

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.

Chapter 4 Indicators for Water Resources Systems

Introduction

Indicators are a tool to describe the economic, environmental, social and/or institutional conditions of a system, i.e. a country, region, community, etc. Examples of indicators are the well-known and frequently used economic indicators gross domestic product (GDP) and gross national product (GNP).

Indicators are instruments of simplification as they summarise large amounts of measurements to a simple and understandable form in order to highlight the main characteristics of a system. Information is reduced to its elements, maintaining the crucial meaning for the questions under consideration. On the other hand, the aggregation causes a loss of information, but if the indicator is planned properly, the lost information will not gravely deform the result.

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.

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.

The following section 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.

Data requirements, scales of application and references for various water-related indicators and indices are at the end of this chapter.

Figure 20 shows the different levels of aggregation of data. The primary data, that are simple measurements, is analysed and combined to indicators, e.g. nitrate concentrations in a river reach or life expectancy at birth. These are formed to a subindex, also called transformed indicator, for each issue in order to convert them to a dimensionless range, typically from 0 to 1 or from 0 to 100, and then aggregated to an overall index consisting of a single number.

Indices include all aspects that are significant for the question under consideration, such as economic, social and environmental issues. Clearly, the application of indicators and indices is constrained by data availability.

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.

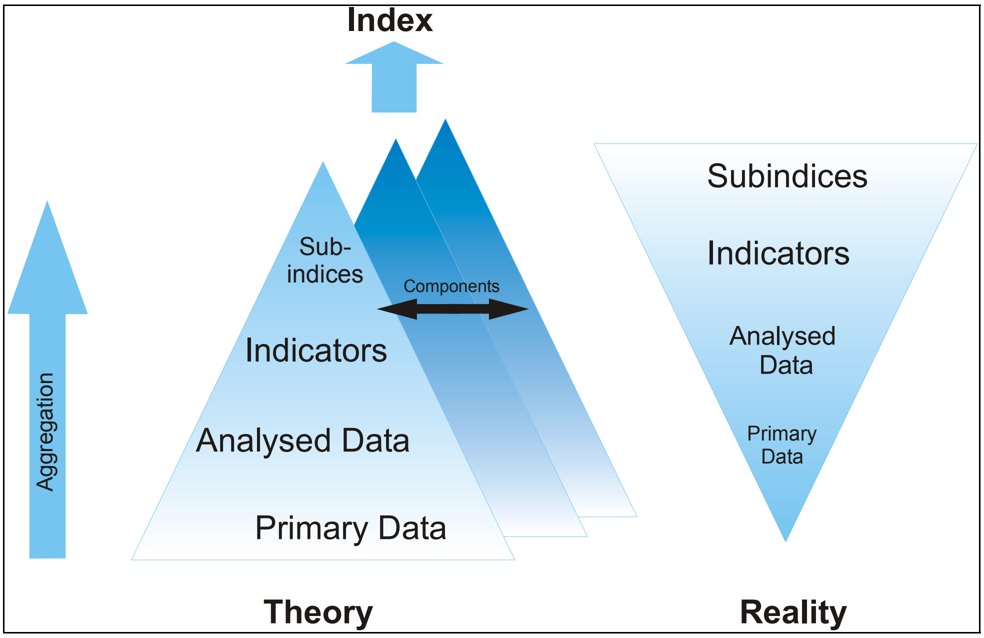

Figure 20: Information pyramid (Wamsley, 2002, modified)

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 assessed 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

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).

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 river basin scale which is the natural for water resources management. The best spatial scale for an indicator is one at which the indicator shows least stochastic variations and little variations if the spatial scale is slightly changed.

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.

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 measures 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" for example is widely used by international organisation, such FAO, World Bank, WHO etc.

However, as Gleick (2002) notes, 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.

Table 17: Definitions of "Access to Safe Drinking Water Source" (WHO, 1996)

Number of countries defining access as "Water source at a distance of less than…"									
	50 m	**100 m**	**250 m**	**500 m**	**1000 m**	**2000 m**	**5 minutes**	**15 minutes**	**30 minutes**
Urban	20	6	3	8	1	-	1	-	1
Rural	10	1	6	17	4	4	-	1	1

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.

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, causes 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.

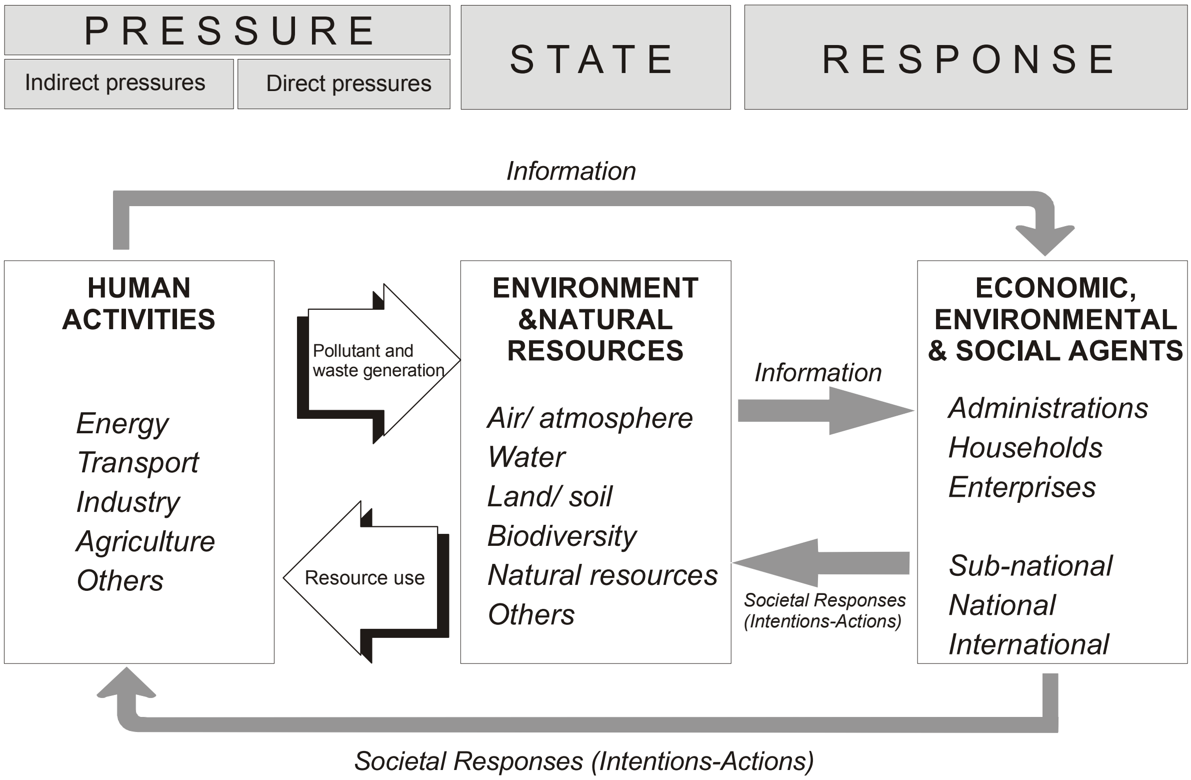

Figure 21: Pressure- State- Response (P-S-R) model (OECD)

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 sectoral level for analysing the pressures exerted by different economic sectors and distinguishing responses from government, private households and the business sector.

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.

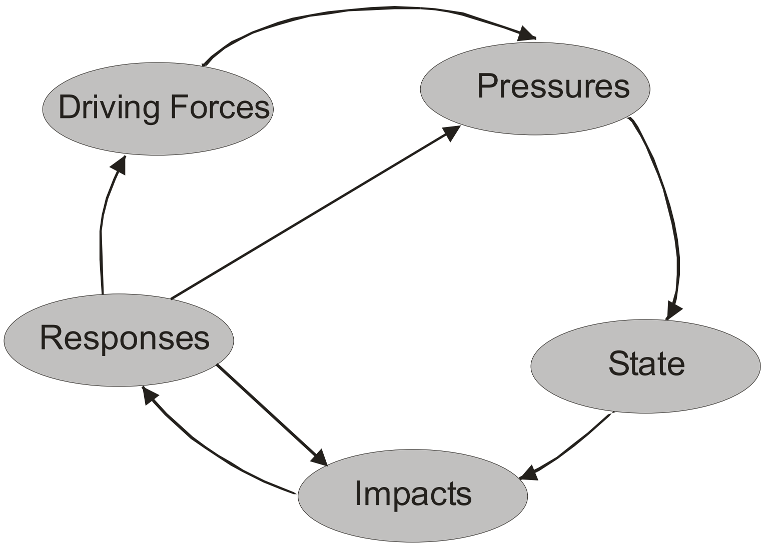

Figure 22: DPSIR model used by the European Environment Agency

The general framework for a DPSIR approach in water resource management is given in Figure 23.

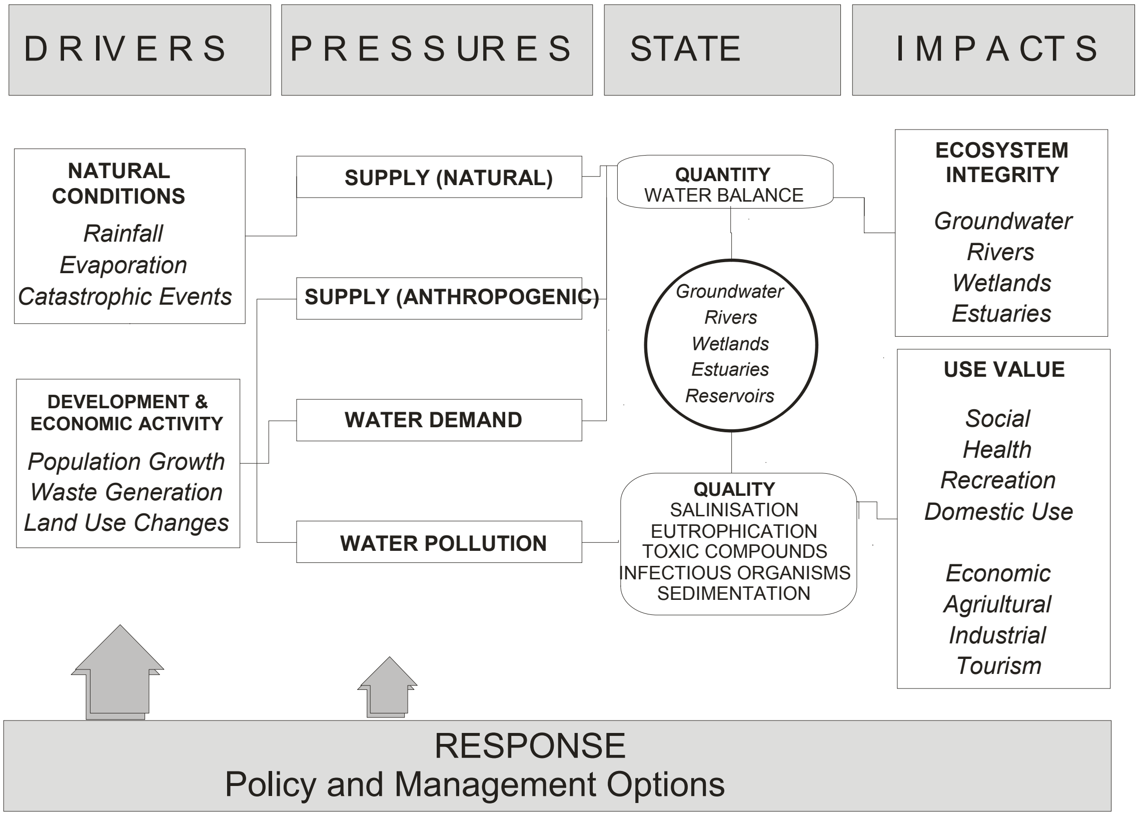

Figure 23: DPSIR Framework in water resources

Frequently Used Indicators and Indices

Agenda 21 calls upon countries and international organisations "to develop the concept of indicators of sustainable development in order to identify such indicators" (Agenda 21, 40.6). As a result, in addition to already existing indicator approaches, various international organisations developed new concepts to describe the conditions of water resources ranging from simple water indicators to composite environmental indicators that try to track many water-related issues. Some of them are briefly described below whilst the spatial applicability, data requirements and a glossary of terms is given later in this chapter.

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.

Falkenmark Water Stress Indicator

Probably the most frequently applied indicator is the Falkenmark Water Stress Indicator, that simply relates the available water resources in a given region (or country) per year to the number of inhabitants, regardless of the temporal and spatial distribution of the water resources.

Originally, the indicator based on the assumption that a flow unit of one million cubic metres of water can support 2,000 people in a society with a high level of development, using Israel as a reference by calculating the total annual renewable water resources per capita.

Water availability of more than 1,700 m^3/capita/year is defined as the threshold below which water shortage occurs only irregularly or locally. Below this level, water scarcity arises in different levels of severity. Below 1,700 m^3/capita/year water stress appears regularly, below 1,000 m^3/capita/year water scarcity is a limitation to economic development and human health and well-being, and below 500 m^3/capita/year water availability is a main constraint to life.

Despite its global acceptance, this indicator has numerous shortcomings. First of all, only the renewable surface and groundwater flows in a country are considered. Moreover, the water availability per person is calculated as an average with regard to both the temporal and the spatial scale and thereby neglecting water shortages in dry seasons or in certain regions within a country.

Furthermore, it does not take the water quality into account at all nor does it give information about a country's ability to use the resources. Even if a country has sufficient water according to the Falkenmark indicator, these water resources possibly cannot be used because of pollution or insufficient access to them.

Based on FAO-AQUASTAT data, countrywide values of water availability are plotted against water demand per capita for selected countries in Figure 24.

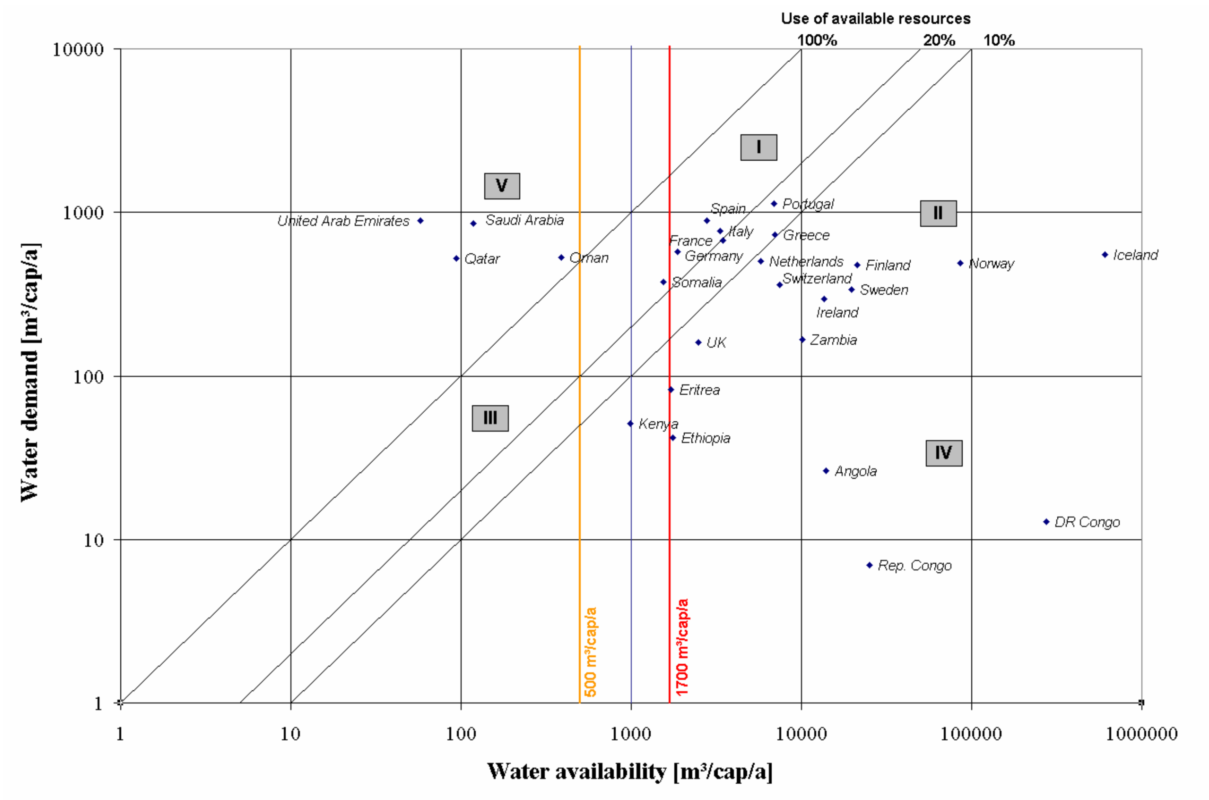

Figure 24: Falkenmark Water Stress Index for selected countries (based on FAO-AQUASTAT data)

Dry season flow index

This indicator was developed by the World Resources Institute (WRI) as part of the Pilot Analysis of Global Ecosystems (PAGE) (WRI, 2000) for the description of water conditions on a river basin level. It considers the temporal variability of water availability that is essential for example in regions with pronounced rainy seasons.

Watersheds with a dry season are those where less than 2 % of the surface runoff is available in the 4 driest months of the year. The indicator is calculated by relating the volume of runoff during the dry season, i.e. during the four consecutive months with the lowest cumulative runoff to the population. Based on the Falkenmark definition, a basin is water stressed if less than 1,700 m³/year/person is available, values between 1,700 m³/year/person and 4,000 m³/year/person indicate adequate supply of water.

Water availability index WAI

Meigh et al. (1999) took in their GWAVA (Global Water AVailability Assessment) model the temporal variability of water availability into account. The index includes surface water as well as groundwater resources, and compares the total amount to the demands of all sectors, i.e. domestic, industrial and agricultural demands. The month with the maximum deficit or minimum surplus respectively is decisive. The index is normalised to the range –1 to +1.

The index is zero if availability and demand are equal.

$$WAI = \frac{R + G - D}{R + G + D}$$

where
R = surface runoff
G = groundwater resources and
D = sum of demands of all sectors.

The surface water availability is calculated as the 90% reliable runoff. The groundwater availability is estimated either as the potential recharge that is calculated from the monthly surface water balance, or as the potential aquifer yield, whichever value is lower.

Basic Human Needs Index

This approach is based on the use of water instead of water availability. Gleick (1996) quantified the minimum amount of water that a person needs for basic water requirements (BWR), such as drinking, cooking, bathing, sanitation and hygiene, as 50 litres per person per day. According to this definition, estimates of the number of countries where the average domestic water use is below this threshold are made.

This indicator is only calculated on country-level so that regional water scarcity is not depicted. Again, water quality is not included into the concept. Furthermore, country data about the domestic water use are insufficient and unreliable, and the needs of other water users, such as the industry, agriculture or nature itself, are not included at all into the approach.

Index of water scarcity

An indicator that combines information about water abstractions and water availability is the index of water scarcity. It is defined by the intensity of use of water resources, i.e. the gross freshwater abstractions as percentage of the total renewable water resources or as percentage of internal water resources.

Heap et al. (1998) added the variable of desalinated water resources to this indicator. The share of desalinated water use is insignificant on the global scale, but it is crucial in some regions, as for example in the United Arab Emirates where desalinated water corresponds to 18 % of the annual abstractions. This indicator is defined by the ratio

$$R_{WS} = \frac{W - S}{Q}$$

where R_{WS} is the water scarcity index, W are the annual freshwater abstractions, S are the desalinated water resources and Q is the annual available water which is calculated by

$$Q = R + \alpha \cdot \sum D_{up}$$

where R are the internal water resources in the country, D_{up} is the amount of external water resources and α is the ratio of the external water resources that can be used. The factor α is influenced by the quality of the transboundary water, the real consumption of water resources in the upstream region, and the accessibility of water.

The severity of water stress is classified by

$R_{WS} < 0.1$	no water stress
$0.1 < R_{WS} < 0.2$	low water stress
$0.2 < R_{WS} < 0.4$	moderate water stress
$0.4 < R_{WS}$	high water stress

Again, this indicator neglects temporal and spatial variations as well as water quality data.

Vulnerability of water systems

Gleick (1990) developed this index for watersheds in the United States as part of an assessment of the potential impacts of climate change for water resources and water systems.

It describes the vulnerability of water resources systems based on the five criteria and corresponding thresholds that are briefly described below. These five indicators are not aggregated to an overall index but for each region the number of vulnerable sections is presented. This approach emphasises the sectors of watersheds that are threatened.

- **Storage volume relative to total renewable water resources**. A basin is defined as vulnerable if the storage capacity is less than 60 % of the total renewable water resources.
- **Consumptive use relative to total renewable water resources**. The threshold for vulnerability is a ratio of 0.2
- **Proportion of hydroelectricity relative to total electricity**. If the part of hydroelectricity is more than 25 %, the region is considered vulnerable.
- **Groundwater overdraft relative to total groundwater withdrawals**. Regions with a ratio above 0.25 are defined as vulnerable.
- **Variability of flow.** This sub-indicator is computed by dividing the surface runoff exceeded only 5 % of the time by the quantity exceeded 95 % of the time. A low ratio indicates a low variability of runoff and by that a low risk of both floods and droughts. A variability value above 3 indicates vulnerability in this aspect.

Water Resources Vulnerability Index (SEI)

The Water Resources Vulnerability Index (WRVI) has been proposed by the Stockholm Environmental Institute (SEI) in 1997 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 25). 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.

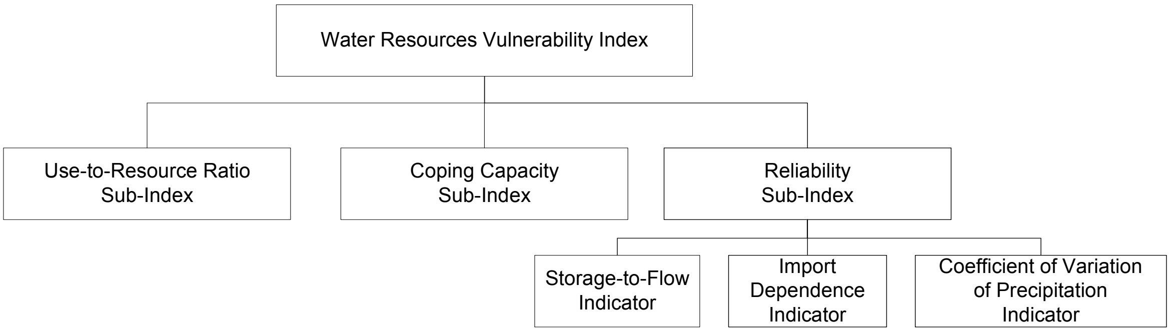

Figure 25: The SEI Water Resources Vulnerability Index (Gleick, 2002)

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 as 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.

Table 18: 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	

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.

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 has 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 been 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 four sub-criteria:

Table 19: 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 determining 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 26 illustrates a time series plot of 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.

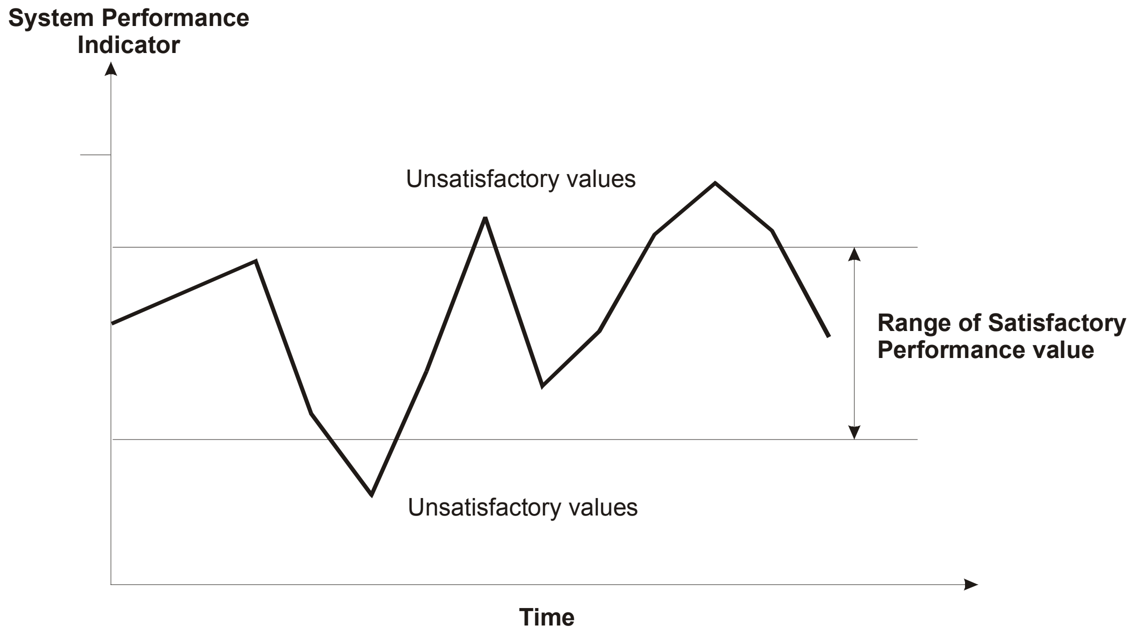

Figure 26: Measures of a system performance indicator (ASCE, 1998)

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 by 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 for 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.

Environmental Sustainability Index (ESI)

The Environmental Sustainability Index (ESI) measures overall progress towards environmental sustainability in five core components (World Economic Forum, 2002).

Table 20: Core components and indicators of the ESI

Category	Indicator
Resource Depletion	Water consumption
	Inputs of phosphate to agricultural land
Dispersion of Toxic Substances	Index of heavy metal emissions to water
	Emissions of persistent organic pollutants (POPs)
	Consumption of toxic chemicals
Water pollution	Emissions of nutrients by households
	Emissions of nutrients by industry
	Pesticides used per hectare of utilised agriculture area
	Nitrogen quantity used per hectare of utilised agriculture area
	Emissions of organic matter from households
	Emissions of organic matter from industry
	Non-treated urban waste water
Urban Environmental Problems	Non-treated urban wastewater
Marine Environment and Coastal Zones	Tourism intensity

OECD water related indicators

In 1994 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 21: Water-related indicators of OECD set of key indicators

Issue	Pressure	State	Response
Eutrophication	Emissions of N and P in water and soil	BOD/DO in inland waters	Population connected to secondary and /or tertiary sewage treatment plants
	N and P from fertiliser use and livestock	Concentration of N and P in inland waters	User charges for waste water treatment
			Market share of phosphate-free detergents
Toxic contamination	Emission of heavy metals	Concentrations of heavy metals and organic compounds in environmental media	
	Emission of organic compounds		
	Consumption of pesticides		
Acidification		Exceedance of critical loads of pH in water	
Water resources	Intensity of use of water resources (abstractions/ available resources)	Frequency, duration and extent of water shortages	Water prices and charges for sewage treatment
Biodiversity			Protected areas as % of national territory and by type of ecosystem

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

Water Poverty Index (WPI)

Recently, the Water Poverty Index (WPI) (Sullivan, 2002, Lawrence et al., 2002), developed by the Centre for Ecology and Hydrology (CEH) in Wallingford, has been intensively discussed. This index tries to show the connection between water scarcity issues and socio-economic aspects. It ranks countries according to the provision of water, combining five components which are:

- Resources
- Access
- Use
- Capacity and
- Environment

Each of these components is derived from two to five indicators which are normalised to a scale from 0 to 1.

In case of an equal weighting, the subindex and component values are then calculated as a simple average of the corresponding indicators, and this value is multiplied by 20. The overall index is generated as a sum of the component values so that the value is between 0 and 100. A value of 100 is only possible if a country ranks best in all of the five components.

Table 22: Components of the Water Poverty Index

Component	Subindex	Indicator	Unit
Resources		internal water resources	km³/cap/year
		external water resources	km³/cap/year
Access		access to safe water	%
		access to sanitation	%
		access to irrigation	--
Capacity		GDP per capita	US$
		under-5 mortality rate	per 1000 live births
		UNDP education index	--
		Gini coefficient	--
Use		domestic water use	l/cap/day
		industrial water use (as: proportion of GDP derived from industry/ proportion of water used by industry)	--
		agricultural water use (as: proportion of GDP derived from agriculture/ proportion of water used by agriculture)	--
Environment	water quality	dissolved oxygen concentration	mg/l
		phosphorus concentration	mg/l
		suspended solids	mg/l
		electrical conductivity	mS/cm
	water stress	fertiliser consumption	100 g
		pesticide use	kg
		industrial organic pollutants	metric tons/ km³
		% of countries territory under severe water stress (according to ESI-definition)	%
	regulation and management capacity	environmental regulatory stringency	--
		environmental regulatory innovation	--
		land under protected status	%
		number of sectoral EIA guidelines	--
	informational capacity	availability of sustainable development information at the national level, environmental strategies and action plans	--
		% of ESI variables missing from public global data sets	%
	biodiversity	% of threatened mammals	%
		% of threatened birds	%

Fairness, reversibility and risk

Bender and Simonic (1997) argue that a number of issues are 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.

Plan Bleu

The Mediterranean Commission on Sustainable Development (MCSD) has the target to provide a tool to measure progress to sustainable development in the Mediterranean countries. For that reason, a set of 130 indicators structured according to the PSR-approach was developed by its activity centre called “Plan Bleu pour l'environnement et le développement en Méditerranée” (Blue Plan for the Environment and Development in the Mediterranean), 40 among them were adopted from the UNCSD working list of indicators

The indicators provide information in the following categories:

- Population and society
- Lands and areas
- Economic activities and sustainability
- Environment
- The sustainable development: actors and policies
- Exchanges and co-operation in the Mediterranean

The included water-related indicators are summarised in the table below. The indicators that are indirectly connected to water include several that describe the importance of tourism in the country and thus the increasing water demand in the holiday season. Furthermore, two indicators of health that is influenced by the supply with safe water and two indicators of policies and strategies, representing the efforts to improve the situation concerning water resources, are shown.

Table 23: Directly water-related indicators of Plan Bleu

Chapter	Theme	Number	Indicator	Type[1)]
Population and society	Health, Public Health	13	Access to safe drinking water	R
Economic activities and sustainability	Agriculture	50	Use of agricultural pesticides	P
		51	Use of fertilisers per hectare of agricultural land	P
		52	Share of irrigated agricultural land	P
		53	Agriculture water demand per irrigated area	P
		57	Water use efficiency for irrigation	R
	Mines, Industry	63	Industrial releases into water	P
Environment	Freshwater and waste water	84	Exploitation index of renewable resources	P
		85	Non-sustainable water production index	P
		86	Share of distributed water not conform to quality standards	S
		87	Water global quality index	S
		88	Share of collected and treated wastewater by the public sewerage system	R
		89	Existence of economic tools to recover the water cost in various sector	R
		90	Drinking water use efficiency	R
		91	Share of industrial wastewater treated on site	R

[1)] P = Pressure indicator, S = State indicator, R = Response indicator

Index of Watershed Indicators (IWI)

The Index of Watershed Indicators (IWI) was developed by the United States Environmental Protection Agency (EPA, 2002). It includes 15 indicators of watershed condition and vulnerability, of which 7 are related to the condition and 8 to the vulnerability (see Table 24). The condition indicators show the present water quality in different watersheds, and the vulnerability indicators describe the human activities' pressure on the region.

Table 24: 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 above 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

Each condition indicator is given a score and the weighted sum of the scores is assigned, to one of the categories 'better water quality', 'water quality with less serious problems' or 'water quality with more serious problems'. Then, the indicators of vulnerability are scored and classified into high or low vulnerability. The scores vary between 0 and 2, apart from the first indicator which is scored between 0 and 3. The indicators are weighted equally except the first condition indicator which is weighted threefold.

These two groups of indicators are combined to generate the following scale:

Watersheds with:

- better water quality and lower vulnerability
- better water quality and higher vulnerability
- less serious water quality problems and lower vulnerability
- less serious water quality problems and higher vulnerability
- more serious water quality problems and lower vulnerability
- more serious water quality problems and higher vulnerability
- insufficient data

To ensure a solid data basis, information to at least 10 of the 15 indicators must be available. For the condition assessment, at least four of the seven indicators must be present, and at least six of the eight vulnerability indicators are necessary.

The indicators are taken from national databases of the United States. More information about the definitions of the indicators as well as the data sources can be taken from the report of the EPA (2002).

CSD working list of indicators of sustainable development

In 1996, the United Nations Commission on Sustainable Development (CSD) developed a working list of indicators of sustainable development of surface as well as groundwater. Directly water-related and indirectly water-related indicators are given below.

Table 25: Directly water-related indicators of the CSD working list of indicators of sustainable development

Category	Chapter	Driving Forces	State	Response
Social	Protecting and promoting human health		Percent of population with adequate excreta disposal facilities	
			access to safe drinking water	
Environmental	Protection of the quality and supply of freshwater resources	Annual withdrawals of ground and surface water	Groundwater reserves	Waste-water treatment coverage
		Domestic consumption of water per capita	Concentration of faecal coliform in freshwater	
			Biochemical oxygen demand in water bodies	
	Protection of the oceans, all kinds of seas and coastal areas	Discharges of oil into coastal waters	Algae index	
		Releases of nitrogen and phosphorus to coastal waters		
	Promoting sustainable agriculture and rural development	Use of agricultural pesticides	Area affected by salinization and waterlogging	
		Use of fertilizers		
		Irrigation percent of arable land		

Table 26: Indirectly water-related indicators of the CSD working list of indicators of sustainable development

Category	Chapter	Driving Forces	State	Response
Social	Protecting and promoting human health		Life expectancy at birth	
			Infant mortality rate	
Environ-mental	Protection of the quality and supply of freshwater resources			Density of hydrological networks
	Promoting sustainable agriculture and rural development			Agricultural education

European System of Environmental Pressure Indices (EPI)

The European Commission's Environmental Directorate financed an initiative to develop a set of environmental pressure indicators for the EU in order to describe human activities that have a negative impact on the environment. 48 indicators were defined structured according to the DPSIR-approach, including several connected to water (see table below).

Table 27: Directly water-related indicators of the European System of Environmental Pressure Indices

Category	Indicator
Resource Depletion	Water consumption
	Inputs of phosphate to agricultural land
Dispersion of Toxic Substances	Index of heavy metal emissions to water
	Emissions of persistent organic pollutants (POPs)
	Consumption of toxic chemicals
Water pollution	Emissions of nutrients by households
	Emissions of nutrients by industry
	Pesticides used per hectare of utilised agriculture area
	Nitrogen quantity used per hectare of utilised agriculture area
	Emissions of organic matter from households
	Emissions of organic matter from industry
	Non-treated urban waste water
Urban Environmental Problems	Non-treated urban wastewater
Marine Environment and Coastal Zones	Tourism intensity

Indicator Summery

Table 28 again summarises most of the frequently used indicators, their spatial scale and data requirements.

Table 28. Commonly used indicators, references, spatial scales and required data

Indicator/ Index	Reference	Spatial Scale	Required Data
Access to drinking water and sanitation services	WHO, 2000	country	❖ percentage of population with access to drinking water ❖ percentage of population with access to sanitation services
Falkenmark Water Stress Indicator	Falkenmark, 1989	country	❖ total annual renewable water resources ❖ population
Dry season flow by river basin	WRI, 2000	river basin	❖ time-series of surface runoff (monthly data) ❖ population
Basic Human Needs Index	Gleick, 1996	country	❖ domestic water use per capita
Indicator of water scarcity	OECD, 2001	country, region	❖ annual freshwater abstractions ❖ total renewable water resources
Indicator of water scarcity	Heap et al., 1998	country, region	❖ annual freshwater abstractions ❖ desalinated water resources ❖ internal renewable water resources ❖ external renewable water resources ❖ ratio of the ERWR that can be used
Water availability index WAI	Meigh et al., 1999	region	❖ time-series of surface runoff (monthly) ❖ time-series of groundwater resources (monthly) ❖ water demands of domestic, agricultural and industrial sector
Vulnerability of Water Systems	Gleick, 1990	watershed	❖ storage volume (of dams) ❖ total renewable water resources ❖ consumptive use ❖ proportion of hydroelectricity to total electricity ❖ groundwater withdrawals ❖ groundwater resources ❖ time-series of surface runoff
Water Resources Vulnerability Index (WRVI)	Raskin, 1997	country	❖ annual water withdrawals ❖ total renewable water resources ❖ GDP per capita ❖ national reservoir storage volume ❖ time-series of precipitation ❖ percentage of external water resources
Indicator of Relative Water Scarcity	Seckler et al., 1998	country	❖ water withdrawals in 1990 ❖ water withdrawals in 2025
Index of Watershed Indicators (IWI)	EPA, 2002	watershed	❖ 15 condition and vulnerability indicators
Water Poverty Index (WPI)	Sullivan, 2002	country, region	❖ internal renewable water resources ❖ external renewable water resources ❖ access to safe water, access to sanitation ❖ irrigated land, total arable land, total area ❖ GDP per capita ❖ under-5 mortality rate ❖ UNDP education index ❖ Gini coefficient ❖ domestic water use per capita ❖ GDP per sector ❖ Water quality variables, use of pesticides ❖ Environmental data (ESI)

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Agence de l’Eau Rhin-Meuse: www.eau-rhin-meuse.fr

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Appendices

Glossary of Terms and Definitions

Actual external renewable water resources

Part of the external water resources that is available, taking into consideration the quantity of flows reserved to upstream and downstream countries through formal or informal agreements or treaties.

Annual water withdrawals

Amount of water that is abstracted from surface or groundwater resources by water companies (public water supply) or directly by water consumers.

Consumption Index

Consumptive use/Total water production

Consumptive use

Water that is abstracted and not longer available for use because it has evaporated, transpired, been incorporated into products and crops, consumed by man or livestock, ejected directly to the sea, or otherwise removed from freshwater resources.

Demand coverage

Describes the relative coverage of water demand for a given sector

Dependency Ratio

Measures the dependence of a region on external water being computed as the total volume of external water flows (importing and inflows) over the total volume of water produced on a yearly basis.

Exploitation Index

Measures the relative pressure of annual production on groundwater resources; Sum of the volumes of annual conventional renewable natural freshwater production for all uses including all loses over the volume of average annual flows of renewable groundwater resources (recharge).

External renewable water resources ERWR

Part of the renewable water resources coming from outside the country or shared with neighbouring countries.

Global renewable water resources GRWR

Long-term average precipitation minus long-term average evapotranspiration plus long-term average incoming flow originating outside the country/region/basin

Internal renewable water resources IRWR

Average annual flow of rivers and recharge of groundwater generated from endogenous precipitation.

Non-sustainable Water production index

Measures the amount of water that is abstracted in excess of the sources' recharge on a yearly basis as a fraction of the total water abstractions. For groundwater it is based on the concept of sustainable yield which is defined as the quantity that can be extracted from an aquifer on a sustainable basis.

Surface runoff

Average annual flow of rivers.

Sustainable yield of aquifers

Quantity that can be extracted from an aquifer on a sustainable basis. Theoretically, the sustainable yield is equal to recharge but it is in most cases considered less than recharge as it must also allow for adequate provision of water to sustain streams, springs, wetlands and groundwater dependent ecosystems. Abstractions from renewable groundwater are therefore considered to be unsustainable if the yearly amount abstracted exceeds the amount of recharge multiplied by a factor that allows for such needs

Transboundary water

see: external renewable water resources

Unconventional water resources

The sum of desalinated water resources and reused treated wastewater.