SABINE LATTEMANN

DEVELOPMENT OF AN ENVIRONMENTAL IMPACT ASSESSMENT AND DECISION SUPPORT SYSTEM FOR SEAWATER DESALINATION PLANTS

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DISSERTATION

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Παντα ρει

“All is flux, nothing stays still”

Attributed to the Greek philosopher Heracleitus.

“Plato had taken over from his predecessor Heracleitus [...] the doctrine that the world of sensible things is a world of things in constant flux; as he put it [...] ‘nothing is in this world because everything is in a state of becoming something else’.”

In addition, Heracleitus “noted that a single substance may be perceived in varied ways−seawater is both harmful (for human beings) and beneficial (for fishes).”

From the Encyclopædia Britannica

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Preamble

The large scale production of drinking water from the sea started with the advancement of desalination technologies in the mid-20th century. Today, many countries of the Middle East depend heavily on desalinated water, and many other countries, which so far relied on conventional water supplies, turn to desalination in order to develop and diversify their water supply options in the face of economic and demographic growth, urbanization and climate change. As many of the world's freshwater reserves are being depleted [1], seawater is often proclaimed as the only truly *unlimited* water resource.

This is true in so far as 97% of the world's water is stored in the oceans, and only an infinitesimal small amount of water is removed by desalination processes compared to natural evaporation. A major constraint rather lies in the waste water discharges of desalination plants and their local ecosystem impacts. Many coastal ecosystems and regional seas are already under stress from other anthropogenic activities including land reclamation and habitat degradation, fisheries and maritime shipping, eutrophication and land-based pollution. Desalination is listed among the main sources of land-based pollution in the Gulf and Red Sea areas according to the UNEP regional seas reports [2, 3] and is on the verge of becoming a new coastal-based industry in other parts of the world.

As the need for desalination accelerates and spreads to new markets, it is realized in many parts of the world that a balance between water supply through desalination development and environmental protection must be maintained. While some people portray desalination as the panacea for much of the world's water woes, others perceive it very negatively, and it remains even more necessary to gain an objective understanding of the real stakes in desalination within the context of integrated water resource management [4]. Environmental research needs to be emphasized up front [5] and should lead to sustainable development, otherwise the desalination boom is in danger of shifting the problem from water to energy [6], and from the freshwater to the marine ecosystems [7].

Problem setting

The increases in desalination activity in many sea regions and the growing number of industrial-sized facilities raise concerns over potential negative impacts of the technology on the environment. Environmental impact assessment (EIA) studies are widely recognized and accepted as a suitable approach for identifying, evaluating and mitigating the wide range of potential impacts of new development project on the environment. However, an "internationally agreed environmental assessment methodology for desalination plants does not exist so far and its development would be desirable" [Manuel Schiffler, The World Bank, 4]. Another problem is that there is still "a considerable amount of uncertainty about the environmental impacts of desalination and, consequently, concern over its potential effects" [U.S. National Research Council, 5].

Both, a structured EIA methodology and a basic understanding of the environmental impacts is necessary for a successful EIA process. A range of manuals are available that offer guidance on the EIA process *in general*, but it is beyond the scope of these documents to cover all the necessary details that are required to carry out an EIA for a specific project *in practice*. Moreover, difficulties may be experienced in handling the large amounts of complex information that are typically generated by EIA studies in a consistent way for decision making, especially when different project alternatives need to be compared. The development of a systematic EIA methodology and decision support system (DSS) is desirable in order to facilitate the process of impact assessment and decision making for desalination projects in the future.

The first step towards a better understanding of the environmental effects is to systematically document the existing knowledge, followed by further research in the field and laboratory, and a meta-analysis of the effects. Although the number of publications discussing the *potential* for negative environmental impacts of effluents from desalination facilities has been steadily increasing over the last years, a surprising paucity of useful *experimental data*, either from laboratory tests or from field monitoring still exists [5]. To facilitate studies on potential biological or ecological effects at the project level, and thereby improve understanding at the meta level, guidance on field and laboratory studies is needed, such as monitoring and assessment protocols.

Research objectives

The overarching goal of this PhD study is to elaborate and validate a systematic EIA and DSS for large seawater desalination projects. The emphasis is on seawater reverse osmosis (SWRO) as the predominant process in the newly emerging desalination markets outside the Middle East, and even in the Middle East, a trend towards SWRO can be observed. The main objectives of the PhD study are to:

- conduct a systematic analysis and evaluation of potentially significant impacts, following the approach of an 'ecological risk assessment', and develop strategies and identify measures for impact mitigation,
- develop and validate an environmental impact assessment and decision making framework for large SWRO projects including:
  - a methodology for environmental impact assessment studies including specifications for accompanying laboratory, modeling and field monitoring studies,
  - a DSS based on multi-criteria analysis.

Research context

The first paper which noted that the brine and chemical discharges of desalination plants may pose a risk to the marine environment appeared in 1979. It called for a thorough investigation of both the physical and biological components of the environment prior to construction and on a regular basis during operation [8]. However, it took until the 1990's before the scientific interest in the marine environmental concerns of desalination plants became more apparent and the number of publications on the topic increased [e.g. 9–14]. The first comprehensive review of the existing literature sources on desalination plant effluents and their potential impacts on the marine environment followed in 2001. Lattemann and Höpner [15] concluded that more experimental data is needed, including field investigations, laboratory toxicity tests and modeling studies. No more than a dozen of these experimental studies have been published since. A likely explanation is that project developers are statutorily not required in many countries to conduct extensive experimental studies, or to provide access to the results to a wider public.

In 2004, the World Health Organization (WHO) identified a clear public health and environmental protection argument to develop a publication on "Desalination for Safe Water Supply". A project was initiated through the WHO Eastern Mediterranean Regional Office (EMRO), and five technical working groups were established which were assigned the task to investigate and evaluate the different topic areas of the publication, i.e., technology, health, microbiology, monitoring and environment. The results and recommendations from the *environmental* working group were partly included in the WHO publication [16] and reproduced in full by the United Nations Environment Programme (UNEP) as a stand-alone guidance document in 2008 [17].

Although international organizations like WHO, UNEP, and recently also the World Wildlife Fund [18] have taken up the subject and published reports on desalination, the environmental impacts are still the subject of considerable debate, which is often based on hypothesis rather than scientific evidence. A report released by the U.S. National Research Council (NRC) in 2008 concluded that there is still "a surprising paucity of useful experimental data, either from laboratory tests or from field monitoring, to assess these impacts" [5]. Two long-term goals for further research identified in the NRC report are to develop monitoring and assessment protocols for evaluating the potential ecological impacts of surface water concentrate discharge, and to carry out site-specific assessments of the impacts of source water withdrawals and concentrate management.

Independent from the NRC conclusions, the European Community has decided to foster the sustainable use of desalination processes in the EU by financing the research project "Membrane-Based Desalination: An Integrated Approach" (MEDINA) within the Sixth Research Framework from 2006 to 2009. The project's overall objective is to improve the performance of membrane-based water desalination processes by developing advanced analytical methods for feedwater characterization, identifying optimal pre-treatment and cleaning strategies, reducing the environmental impacts of brine disposal and energy consumption, and by developing strategies for environmental impact assessment (EIA) studies and monitoring activities. This PhD work was partly carried out within the WHO project on "Desalination for Safe Water Supply" and partly within the EU research project MEDINA. The above mentioned UNEP guidance document on desalination combines results from both projects.

Thesis outline

Part I of this thesis comprises six chapters, which revolve around the present state of the technology (chapters 1 and 2), its potential environmental impacts (chapters 3 to 5) and approaches for mitigating these impacts (chapter 6):

Chapter 1 provides an overview on the worldwide desalination capacity and discusses regional and future trends. The figures show that desalination is developing into a coastal-based industry with potentially harmful effects on the environment. The impacts will generally depend on the location, size and process of a desalination project.

Chapter 2 therefore describes the main desalination processes and modes of operations from intakes to outfalls in further detail. The intention of this introductory chapter on desalination technologies was not to reproduce textbook materials but to highlight aspects of resource consumption and resulting emissions into water, soils and air.

Chapter 3 discusses the potential impacts on the marine environment caused by the construction of intake and outfalls, the impingement and entrainment of marine organisms with the intake water, and the concentrate and chemical discharges into the sea.

Chapter 4 deals with the second main environmental concern, that is, energy use. This chapter puts energy demand of desalination into perspective, provides estimates of air pollutant emissions, and discusses the environmental implications of energy use.

Chapter 5 provides a synthesis and evaluation of the potential environmental impacts of desalination plants following the approach of an 'ecological risk assessment'. This comprises the key issues discussed in chapters 3 and 4 as well as a long list of other potentially relevant issues related to the construction and operation of a desalination project, as would have been the case in a full-fledged environmental impact assessment.

Chapter 6 discusses if desalination can be considered as a 'sustainable' and 'green' technology, as claimed for some projects, and proposes a concept for 'best available techniques' (BAT) of seawater desalination projects. BAT aims at the identification of state of the art technologies which indicate the practical suitability for preventing or reducing pollution and therefore minimizing impacts on the environment as a whole.

Part II deals with the EIA of desalination projects in three chapters:

Chapter 7 discusses the concept and methodology of EIAs in general and specifically for desalination plants. A 10-step approach is proposed, ranging from screening and scoping of a project to decision making and post-EIA environmental management.

Chapter 8 then deals with the environmental monitoring of desalination projects, which is an intrinsic element of EIAs. The chapter discusses problems in designing adequate monitoring programmes that can adequately distinguish impacts from natural processes. The scopes of the studies are outlined, including baseline and operational monitoring, compliance monitoring, toxicity testing and hydrodynamic modeling, and criteria for assessing the sensitivity of species and habitats are proposed.

Chapter 9 explores the usefulness of multi-criteria analysis (MCA) as a decision support tool for EIAs of desalination projects. The process of decision making can be described as a conflict analysis characterized by environmental, social, economic and political value judgments, which is essentially a search for an acceptable compromise solution. The process can be facilitated by a formalized decision support tool, such as MCA, which allows for a comparison of alternatives under multiple quantitative and qualitative criteria and under different stakeholder perspectives. As a practical example, MCA is used to evaluate different intake and pretreatment options for SWRO.

Part III contains the general conclusions of this thesis.

Part I

Potential environmental impacts of seawater desalination

Global desalination situation

1

1.1 Introduction

Sea- and brackish ground water are the single most important sources of drinking water in a few water-scarce but oil-rich countries of the Middle East which depend heavily on desalination, such as Kuwait or the United Arab Emirates. Many industrialized and developing regions, however, have recently also started to use desalination as a way to supplement and diversify their water supply options. Desalinated water has become a commodity for these countries in order to satisfy their growing demand for water.

For the 'pioneering' countries, the driving factors were often a lack of surface waters and groundwater coupled with sufficient natural or financial resources to engage in energy-intensive and costly desalination projects. For the newly emerging desalination markets, driving factors are more diverse and include economic and demographic growth, urbanization, droughts and climate change, or declining conventional water resources in terms of quality and quantity due to overuse, pollution or salinisation. Moreover, as conventional water production costs have been rising in many parts of the world and the costs of desalination − particularly seawater desalination − have been declining over the years, desalination has also become economically more competitive.

The selection of the desalination process is typically based on different operational parameters such as the availability of a raw water or energy source (e.g., seawater vs. brackish water or low-cost heat vs. electricity), the product water demand, intended use and product water quality specifications (industrial vs. municipal use), or the technical know-how, capacity and costs to build, maintain and operate the plant [19].

This chapter gives a short account of the historical development of desalination technologies, an overview on the presently installed worldwide desalination capacity, distinguishing between different raw water sources, processes and use types. It furthermore discusses regional and future trends, and driving factors such as cost and energy demand. The main objective of this chapter is to set the stage for the following chapters on environmental impacts, by illustrating that desalination is at the brink of becoming a global coastal industry.

Parts of this chapter were based on:

S. Lattemann, M.D. Kennedy, J.C. Schippers and G. Amy. Global desalination situation. In I.C. Escobar and A.I. Schäfer, editors. Sustainability science and engineering, vol. 2, Sustainable water for the future, pages 7−39, Elsevier, The Netherlands, 2009.

1.2 Historical development

The extraction of salt from salty water by means of natural evaporation has been practiced for a long time, dating from the time when salt, not water, was the precious commodity [20]. Advanced technologies that mimic natural processes such as evaporation or osmosis in order to extract the water have only been developed in modern times. Basic desalting processes were first used on naval ships in the 17th to 19th century. The island of Curaçao in the Netherlands Antilles was the first location to make a major commitment to desalination in 1928, followed by a major seawater desalination plant built in what is now Saudi Arabia in 1938 [5, 20].

Many of the early projects focused on thermal processes. Significant work was completed on construction materials, heat transfer surfaces and corrosion, which was instrumental in assisting the design and construction of the first large distillation systems in the Middle East [5]. The multi-effect distillation (MED) process has been used in industry for a long time, traditionally for the production of sugar and salt. Some of the early distillation plants also used the MED process, however, the multi-stage flash (MSF) process that was developed in the 1950s continually displaced the MED process due to a higher resistance against scaling. A revived interest in MED can be observed since the 1980s due to a lower operating temperature and energy demand of the process [21].

During the late 1950s, the first asymmetric membrane for desalination was developed by Loeb and Sourirajan, which consisted of cellulose acetate polymer [22]. The electrodialysis (ED) process, which was commercially introduced in the early 1960s, moves salts selectively through a membrane driven by an electrical potential. It was the first cost-effective way to desalt brackish water and spurred a considerable interest in using desalting technologies for municipal water supply, especially in the United States. ED is exclusively applied to low brackish and fresh water desalination, since the energy consumption for seawater treatment would be far too high. Other milestones included the commercialization of reverse osmosis (RO), a pressure-driven membrane process, in the early 1970s [21], followed by the development of a more robust composite aromatic polyamide spiral wound membrane in the 1980s [22].

A wide variety of membrane materials and module configurations have been developed over the years, including hollow fine fibres from cellulosic or non-cellulosic materials, but spiral wound composite polyamide membranes are almost exclusively used today. While cellulose acetate membranes had a specific permeate flux of 0.5 l $m^2/h/bar$ and a salt rejection of 98.8% in the 1970s, the latest polyamide seawater membranes have a specific flux of more than 1.2 l $m^2/h/bar$ and a salt rejection of 99.8%. The improvement in specific flux translates into a significant reduction of the specific energy demand of the RO process [22]. Another significant power and cost reduction stems from the development of energy recovery devices, which reduced the *total* energy demand of seawater RO to 3-4 kWh per m^3 of permeate water using state of the art technology.

To conclude, it took about 50 years from the first land-based distillation plants to a fully developed industry in the 1980s. By the 1990s, the use of desalting technologies for municipal water supplies had become commonplace [21]. Today, municipalities are the main end users of desalinated water and the market continues to grow exponentially, with a doubling of the installed capacity expected from 2006 to 2015. RO has emerged as the most important desalination process today. In 1969, the world's largest RO system in operation was a 380 m^3/d brackish water plant in Dallas, Texas [23]. Today, the largest seawater RO plant, located in Ashkelon, Israel, produces 330,000 m^3 of water per day, and plants with capacities of 500,000 m^3/d are under development.

1.3 Globally installed desalination capacity

The worldwide installed desalination capacity is increasing at a rapid pace. The 20th IDA Worldwide Desalting Plant Inventory [24] indicates that the production capacity of all desalination plants worldwide was around 44.1 million cubic meters per day (Mm^3/d) by the year 2007. The inventory lists facilities that treat seawater, brine, brackish, river, waste or pure water, which are either in construction, online or presumed online. The data of the inventory has been analyzed and interpreted with the following results.

▸ Projected growth of the desalination market

The worldwide installed desalination capacity grew at a compound average rate of 12% a year over the past five years, and the rate of capacity growth is expected to increase even further. Based upon country-by-country analyses involving desalination projects and official data on water supply and demand from agencies around the world, it is projected that the installed capacity may more than double to 98 Mm^3/d by 2015 [25].

▸ Global capacity by source water type

Much of the expected growth of the desalination market will take place in the seawater sector, although brackish water and wastewater desalination processes will presumably also become more important in the future. In some regions, such as California and Israel, waste water exploitation even preceded seawater desalination. 5% of the present capacity of 44.1 Mm^3/d is produced from waste water, 19% from brackish water and 63% from seawater sources (Figure 1, top). Desalination of seawater is hence the dominant desalination process and accounts for a worldwide water production of almost 27.9 Mm^3/d, which is comparable to the average discharge of the Seine River at Paris (28.3 Mm^3/d).

A limited number of plants are being located in estuarine sites, such as the Thames Gateway desalination plant in East London with a capacity of 150,000 m^3/d. The plant withdraws brackish water with a maximum salt content of 11,000 mg/l during low tide and therefore requires only about half the energy (1.7 kWh/m^3) of seawater RO plants. However, the tidal and seasonal variability of the raw water with regard to dissolved and particulate organic matter requires a complex pretreatment consisting of coagulation, flocculation, clarification, media filtration and ultrafiltration [26, 27]. The lower energy demand is a main benefit of estuarine sites, however, the pretreatment challenge may be the reason why only a limited number of projects have been implemented to date.

▸ Global capacity by process

All source water types included, reverse osmosis (RO) is the prevalent desalination process. It accounts for slightly more than half (51% or 22.4 Mm^3/d) of the global capacity (Figure 1, 2nd row). 40% or 17.7 Mm^3/d of the global capacity is produced by distillation plants, either multi-stage flash (MSF) or multi-effect distillation (MED) plants, with relative market shares of 32% (14 Mm^3/d) and 8% (3.7 Mm^3/d), respectively. Minor desalination processes include the membrane-based nanofiltration (NF) and electrodialysis (ED) processes with about 4% market share each (2 Mm^3/d and 1.6 Mm^3/d, respectively).

The picture changes considerably if one distinguishes between the different source water types: Thermal processes account for 61% (17.2 Mm^3/d) of the production of all seawater desalination plants, of which 50% is produced in MSF plants, and seawater RO (SWRO) accounts for 35%. On the contrary, RO accounts for 84% (6.9 Mm^3/d) of the production in brackish water (BWRO) and 79% (1.7 Mm^3/d) of the production in waste water plants, whereas distillation plays a negligible role for brackish water (<2%, 0.1 Mm^3/d) and a minor role (13%, 0.3 Mm^3/d) for waste water desalination.

▸ Global capacity by use type

All source water types included, desalinated water is mainly used for municipal and industrial purposes: 70% (31 Mm^3/d) of the globally desalinated water is used by municipalities and 21% (9 Mm^3/d) by industries (Figure 1, 3rd row). Other end users include the power generation industry (4%), irrigation (2%), military (1%) and tourism (1%).

Again, the picture is different if one distinguishes between the different source water types. Municipalities are also the main end users of desalinated sea and brackish water and account for 83% (23.2 Mm^3/d) and 61% (5 Mm^3/d) of the production, respectively, and 20% (0.4 Mm^3/d) of the production of repurified waste water. As one moves from seawater to brackish water and waste water, the share of municipal use decreases, while the share of industrial use increases. The latter accounts for 12% of the production from seawater, for 23% of the production from brackish water sources, and it is the primary use of repurified waste water: 39% (0.8 Mm^3/d) is used for industrial purposes plus an additional 12% used by the power industry. Irrigation is only the second most important use of repurified waste water with a share of 27% (0.6 Mm^3/d).

▸ Global capacity by plant size

49% of the desalinated water is produced by very large facilities with production capacities of 50,000 m^3/d or more ('XL-sized' plants, Figure 1, last row). The share of production in very large facilities is even higher in the seawater sector, where 66% (18.2 Mm^3/d) of the water is produced in only 122 industrial-sized plants. On the other end of the scale, about 1660 small seawater desalination facilities with production capacities of less than 1000 m^3/d account for only 2% (0.6 Mm^3/d) of the production. The plant size distribution is a bit more homogeneous in the brackish and waste water sectors, where 24% and 27% of the water is produced in XL-sized plants, where large plants account for 34% and 36% of the production, medium plants for 33% and 32% of the production, and small plants for 9% and 5% of the production, respectively.

1.4 Regionally installed desalination capacities

48% (21.0 Mm^3/d) of the global desalination production takes place in the Middle East, mainly in the Gulf countries (19.3 Mm^3/d). 19% (8.2 Mm^3/d) of the desalinated water is produced in the Americas, 14% (6.2 Mm^3/d) in Asia, 14% (6.0 Mm^3/d) in Europe and 6% (2.8 Mm^3/d) in Africa [Figure 2, primary data from 24].

Seawater desalination is the prevalent process in most regions. 61% (17.1 Mm^3/d) of the global seawater desalination capacity is located in the six GCC states, i.e., in Saudi Arabia, the United Arab Emirates, Kuwait, Bahrain, Qatar and Oman. 11% (2.9 Mm^3/d) of the global capacity is located in Southern Europe and 7% (2.0 Mm^3/d) in North Africa. The three enclosed sea areas of the Gulf, the Red Sea and the Mediterranean therefore account for about three quarters of the global seawater desalination capacity.

North America is the only region where brackish water desalination is the dominating process with a capacity of 3.0 Mm^3/d, which represents more than one third (36%) of the global brackish water desalination capacity. 21% of the capacity is located in the GCC states (1.7 Mm^3/d) and 13% (1.1 Mm^3/d) in Southern Europe.

Waste water purification is also primarily practiced in North America (22% of the global waste water desalination capacity, or a total of 0.49 Mm^3/d), closely followed by East Asia (21% or 0.46 Mm^3/d) and the GCC states (19% or 0.42 Mm^3/d). Each of these

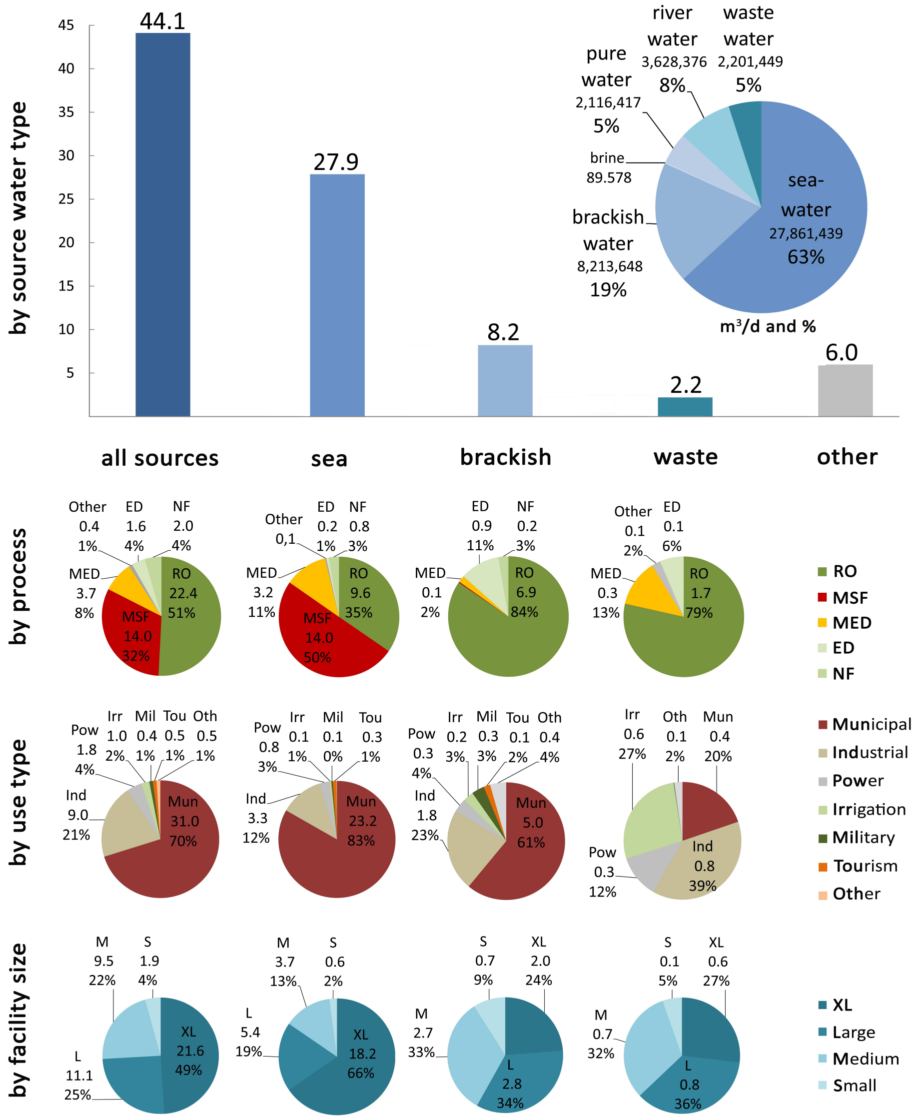

Figure 1: Global desalination capacity (in Mm^3/d and %) by source water type (top row), by process and source water type (2nd row), by use type and source water type (3rd row) and by plant size and source water type (last row). Abbreviations: reverse osmosis (RO), multi-stage flash distillation (MSF), multi-effect distillation (MED), nanofiltration (NF), electrodialysis (ED), XL $\geq$ 50,000 m^3/d > L $\geq$ 10,000 m^3/d > M $\geq$ 1,000 m^3/d > S [primary data from 24].

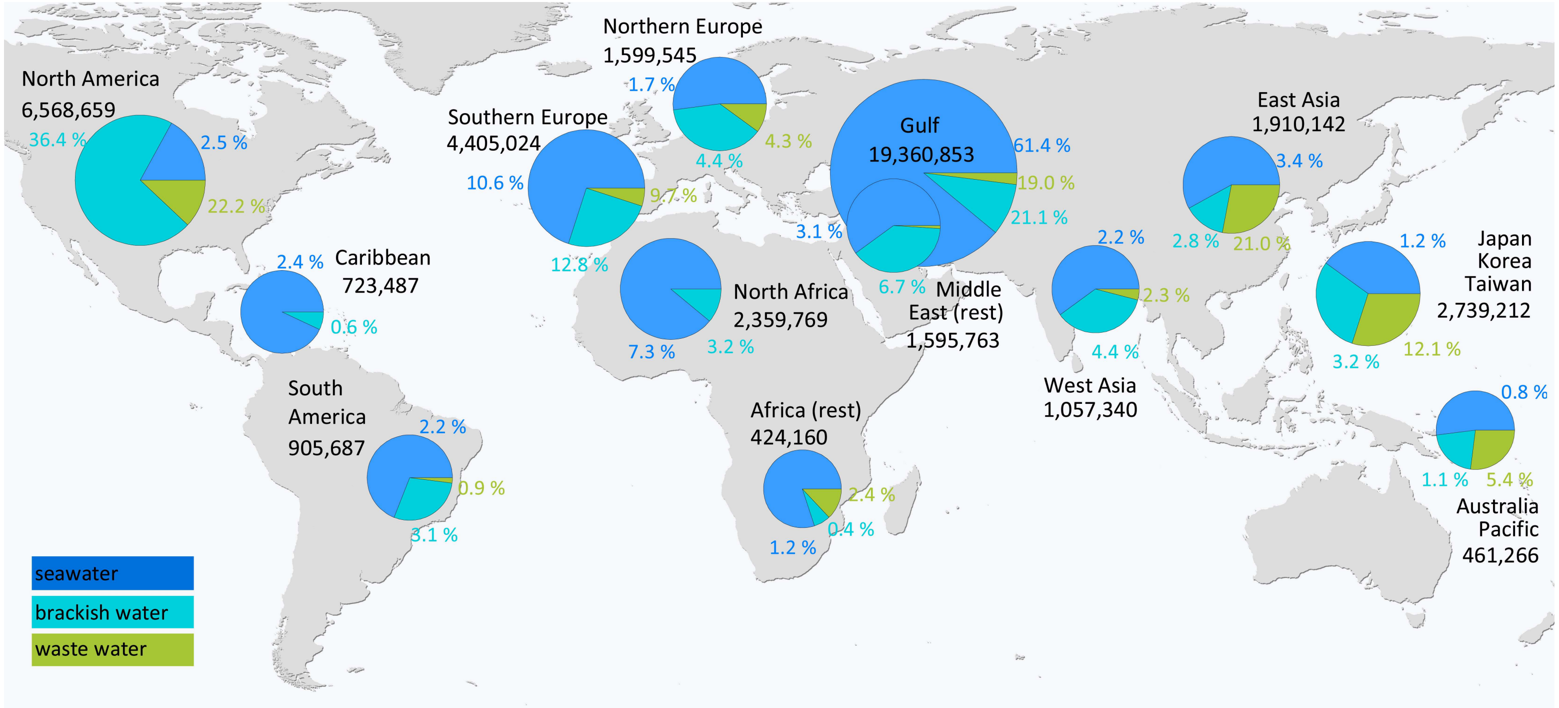

Figure 2: Global desalination capacity in m^3/d and %. For example, the installed capacity in Southern Europe is 4,405,024 m^3/d. This figure includes all source water types. Seawater desalination accounts for most of the production in Southern Europe, brackish water for about 1/4 of the production and waste water desalination plays a relatively minor role (pie diagram). The figures next to the pie diagram give the contribution to the global production, i.e., the seawater desalination capacity in Southern Europe represents 10.6% of the global seawater desalination capacity. The brackish water capacity – though it is less than half the seawater desalination capacity in Southern Europe – represents 12.8% of the global brackish water desalination capacity [primary data from 24].

three regions accounts for roughly one fifth of the global waste water treatment capacity, followed by Japan, Korea and Taiwan (12%) and Southern Europe (10%).

As seawater desalination accounts for most of the production and is in the focus of this thesis, the term desalination is used as a synonym for seawater desalination in the following. In the context of this thesis, it is also of greater interest to consider the installed capacities by *sea area* rather than by world region, due to potential cumulative impacts of desalination activity on the marine environment. The response of the affected ecosystem depends on the magnitude of the impact and the sensitivity of the system to pollution and disturbance. Small semi-enclosed seas may be understood as self-contained ecosystems [28], which are particularly sensitive to pollution. Cumulative effects of desalination plants should therefore be considered in addition to local effects on certain biotopes, such as seagrass meadows or coral reefs. In contrast, the large number of desalination plants on the Canary Islands will unlikely produce measurable effects on the Atlantic Ocean as a whole, although local coastal habitats may as well be affected by the discharges.

1.4.1 The Gulf

In terms of sea areas, the largest number of desalination plants can be found in the Gulf with a total seawater desalination capacity of approximately 12.1 Mm^3/d – or about 44% of the worldwide daily production (Figure 3)[a]. The largest producers of desalinated water in the Gulf (and worldwide) are Saudi Arabia (25% of the worldwide seawater desalination capacity, of which 11% are located in the Gulf region, 12% in the Red Sea region, and 2% with unknown locations), the United Arab Emirates (23% of the worldwide seawater desalination capacity), and Kuwait (6%).

Thermal desalination processes dominate in the Gulf region, as water and electricity are typically generated by large co-generation plants, which use low value steam as a heat source and electricity from power plants for desalination. About 81% of the desalinated water in the Gulf region is produced by MSF and 13% by MED plants, and only a minor amount by SWRO (6%) [primary data from 24].

1.4.2 The Red Sea

In the Red Sea area, desalination plants have a total production capacity of 3.6 Mm^3/d (13% of the worldwide desalination capacity, Figure 4). Similar to the Gulf, most of the water is produced in large co-generation plants (72%), mainly on the Saudi Arabian coast in the locations of Yanbu, Rabigh, Jeddah, Shoaiba and Assir. The world's largest desalination complex with a total water production of 1.6 Mm^3/d is located in Shoaiba. Saudi Arabia accounts for 92% of the desalinated water production in the Red Sea region, with 78% (2.6 Mm^3/d) of the national production coming from thermal plants, whereas Egypt accounts for only 7% of the desalinated water production from the Red Sea, of which 90% (0.2 Mm^3/d) is produced by smaller RO plants, mainly on the Sinai Peninsula and in the tourist resorts along the Red Sea coast [primary data from 24].

► **Figures 3 to 5** show all sites in the Gulf, in the Red Sea and in the Mediterranean Sea with a cumulative MSF, MED or RO capacity ≥ 1,000 m^3/d, and specifically identify all sites ≥ 100,000 m^3/d (Gulf and Red Sea) and ≥ 50,000 m^3/d (Mediterranean Sea) by name and capacity. The total capacity (triangles) of each riparian state and the installed capacity in the sea region is given [first published in 15, updated after 24].

[a] The figure of 44% includes only those plants located on the shores of the Gulf. In contrast to the figure of 61%, which is given for the GCC states on page 10, the figure of 44% does not include plants in Oman and on the Red Sea coast of Saudi Arabia, but it does include plants in Iran.

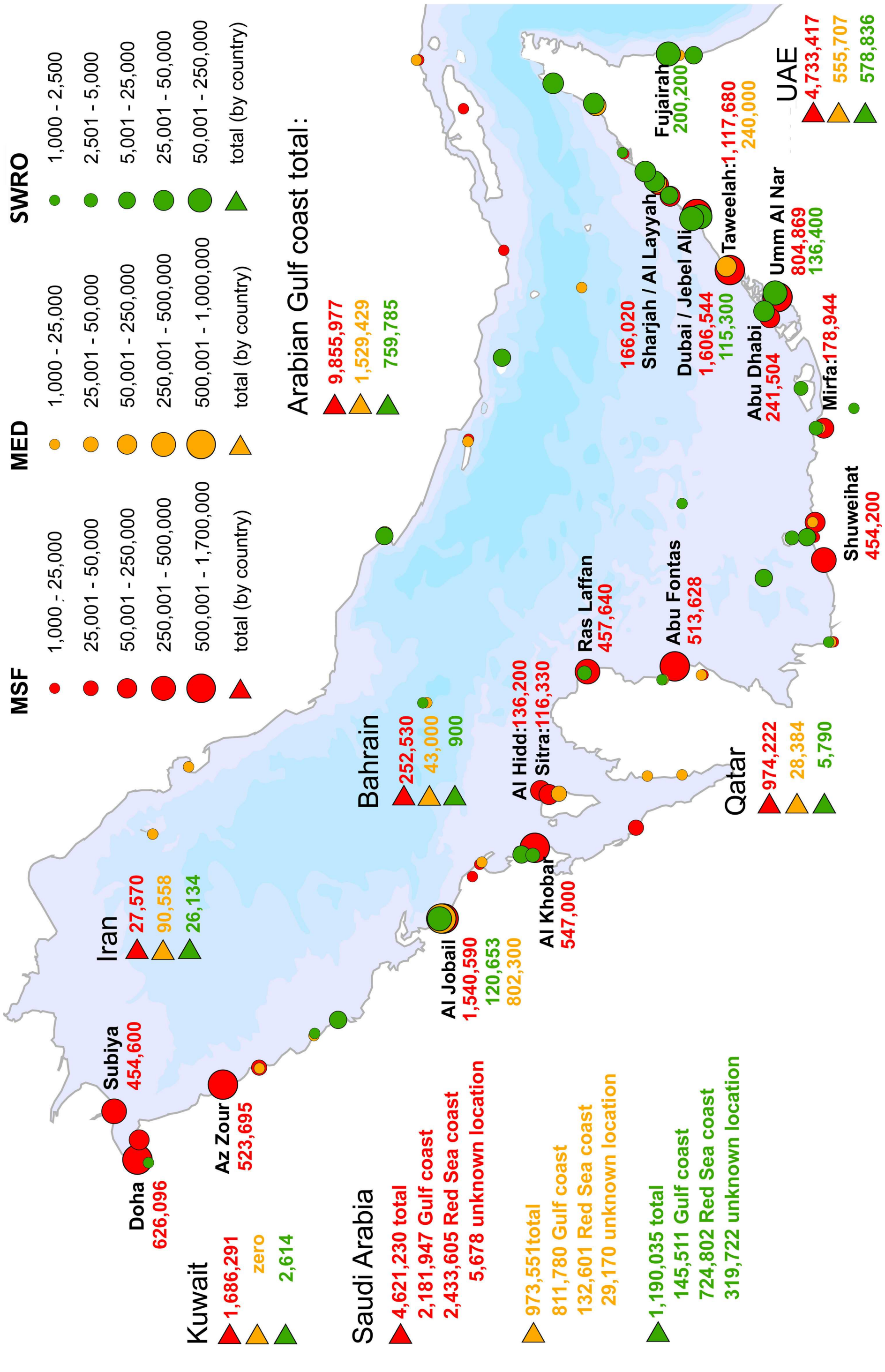

Figure 3: Cumulative MSF, MED and SWRO capacities in the Gulf in m^3/d.

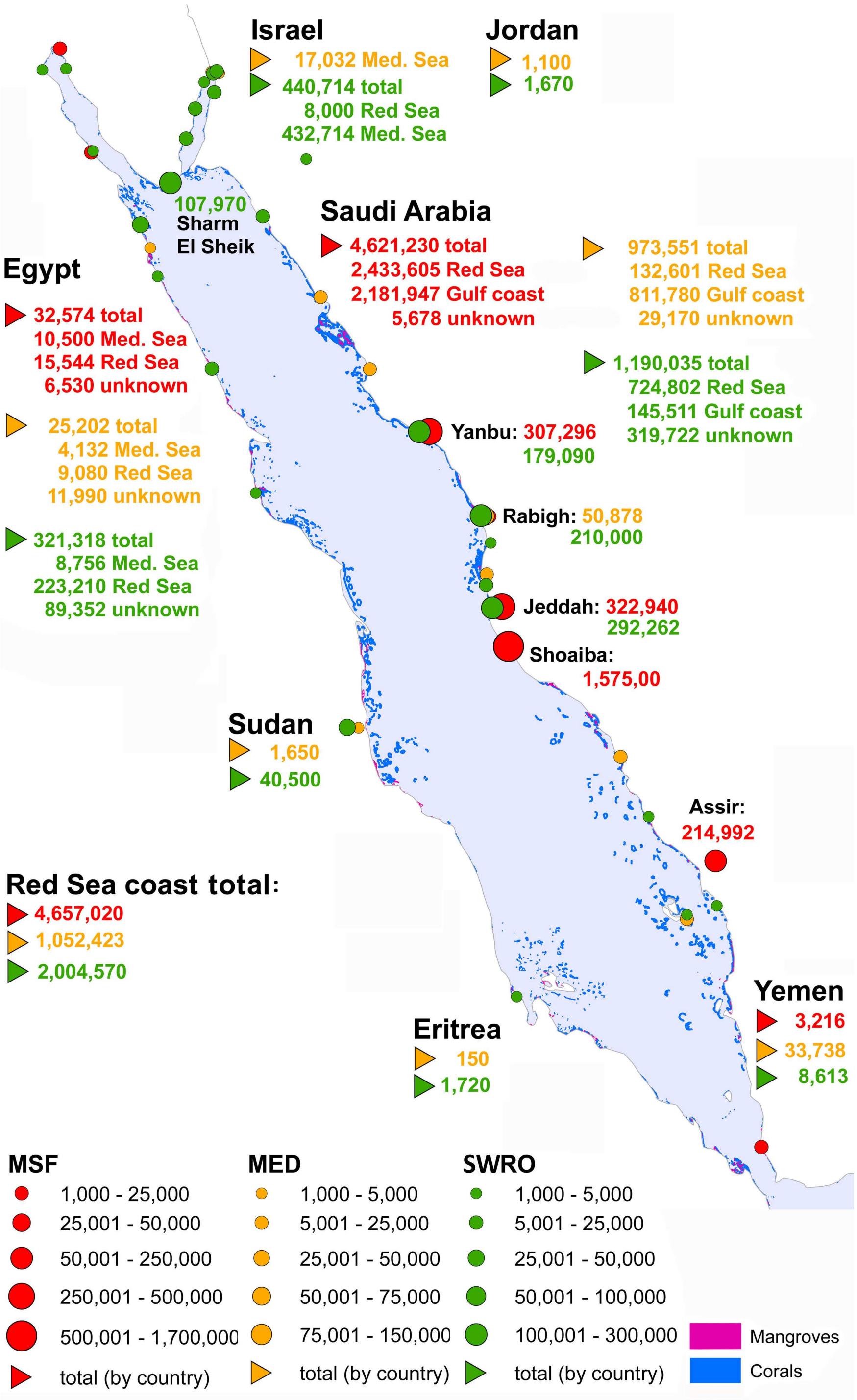

Figure 4: Cumulative MSF, MED and SWRO capacities in the Red Sea in m^3/d.

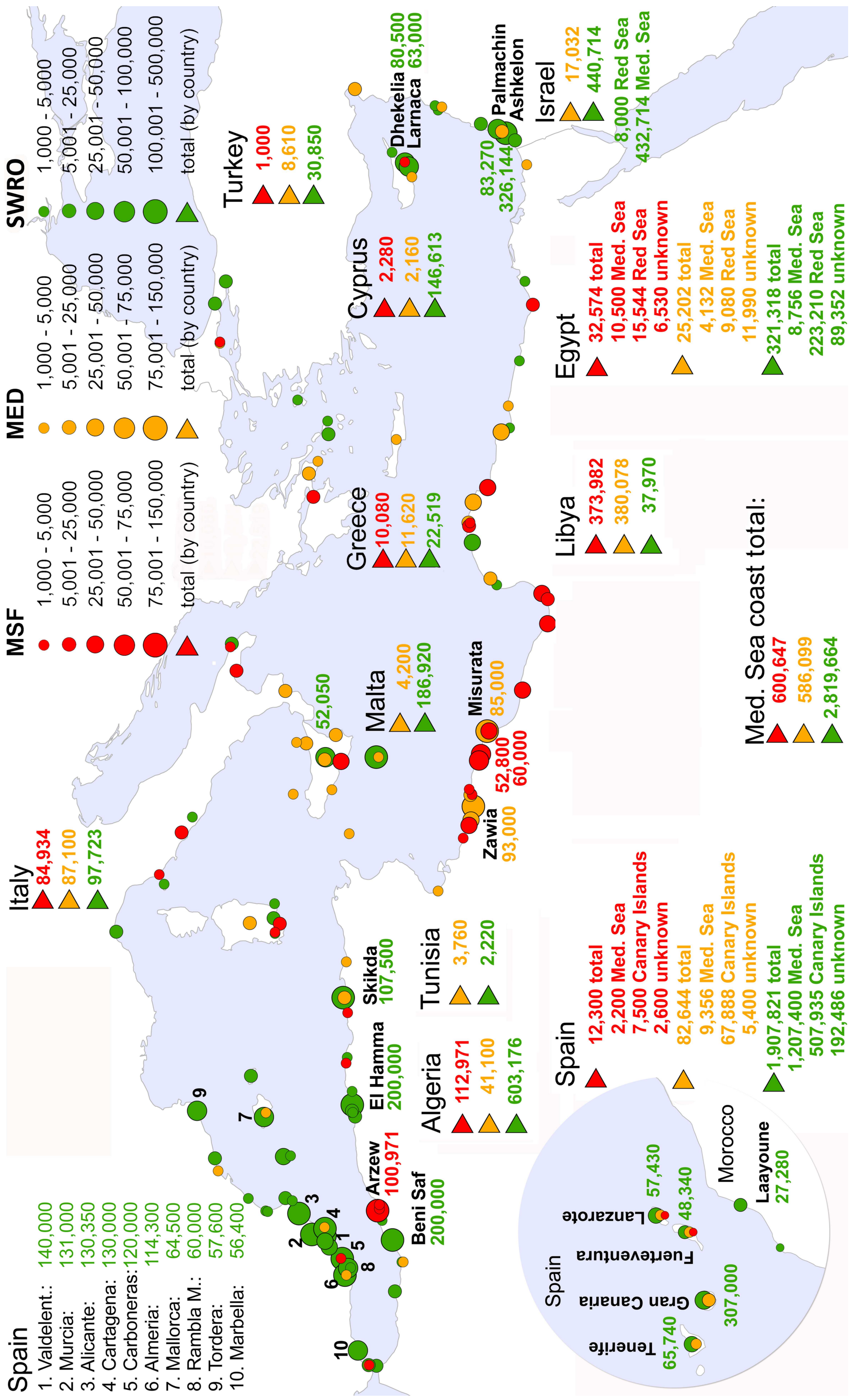

Figure 5: Cumulative MSF, MED and SWRO capacities in the Mediterranean Sea in m^3/d.

1.4.3 The Mediterranean Sea

In the Mediterranean, the total water production from seawater is about 4.0 Mm^3/d (14% of the worldwide desalination capacity, Figure 5). Spain, with about 8% of the world's total production (2.2 Mm^3/d), is the third largest producer of desalinated water globally and the largest in the region. However, about 25% of the Spanish capacity is located on the Canary Islands in the Atlantic Ocean, and 'only' about 65% in the Mediterranean [primary data from 24]. The Spanish AGUA programme will further augment water supply on the Mediterranean coast by desalination, increasing the capacity from 1.4 Mm^3/d (2005) to over 2.7 Mm^3/d (1,000 Hm^3/a) until 2010. The government programme, which also includes water use efficiency and reuse measures, was introduced to avert another main water supply project, that is, diversion of the Ebro river to southern Spain.

While thermal processes (MSF and MED) dominate in the Gulf and Red Sea, the main process in the Mediterranean is seawater RO (SWRO). In 2002, both SWRO and distillation plants still had about equal market shares in the Mediterranean [29]. Today, SWRO accounts for 70% of the production in the Mediterranean and for 99% of the Spanish production in the Mediterranean. Distillation plants are still found in Libya, Algeria and Italy (in decreasing order of priority), but new plants in these countries are also often SWRO plants. For instance, a tremendous expansion of capacities is currently taking place in Algeria, North Africa's fastest growing desalination market, where also the first large SWRO plant (200,000 m^3/d) was opened in February 2008. It is the first in a series of other large projects which will increase the country's desalination capacity to 2 Mm^3/d by 2008 and to 4 Mm^3/d by 2020. Nine projects are currently proposed with capacities between 50,000 m^3/d and 500,000 m^3/d (see also Table 5 on page 30, [30]).

On the Mediterranean coast of Israel, two large SWRO plants are currently in operation – the Ashkelon plant with a capacity of 330,000 m^3/d – which is also the world's largest SWRO project to date, and the Palmachin plant (83,270 m^3/d). Both account for approximately 8% of Israel's water supply [31]. In 2008, the Israeli government approved an emergency programme to address the country's growing water shortage, which will raise the target for desalinated water production to 2.1-2.7 Mm^3/d by 2020 depending on water demand and other alternatives [32]. Several large SWRO desalination plants with capacities up to 274,000 m^3/d are planned along Israel's Mediterranean coast [30]. Furthermore, it is planned to sharply increase the use of the country's brackish water resources, from presently around 16,500 m^3/d to 220,000-274,000 m^3/d [33]. Other measures include more water efficient practices, fixed water quotas, greater enforcement of water restrictions, and upgrading of waste water treatment capacities in order to increase recycling of wastewater from presently 75% to 95% in five years time [30].

1.4.4 Other sea regions

While seawater desalination is already a well-established technology in the above mentioned sea regions, the era of large-scale desalination projects is about to start in other parts of the world. In California, a potential for 15-20 seawater desalination projects with a combined capacity of 1.7 Mm^3/d is expected for 2030 (Figure 6). These would increase the share of desalination to 6% of California's 2000 urban water use. The two largest and most advanced projects are located in the cities of Carlsbad and Huntington Beach with a proposed capacity of 200,000 m^3/d each [34].

In Australia, the first large SWRO plant with a capacity of 144,000 m^3/d became operational in Perth in 2006, followed by a second project of similar size at the Gold Coast in 2008, and a third project in Sydney currently under construction that will have an initial capacity of 250,000 m^3/d and can be expanded to 500,000 m^3/d if necessary

(Figure 6). Further projects are under development in Adelaide, the Upper Spencer Gulf, near Perth and in Karratha with capacities between 120,000 and 140,000 m^3/d, and in Victoria and Queensland with capacities of 400,000-450,000 m^3/d [30, 35].

A third impressive example is China. Desalination capacity is currently estimated at around 366,000 m^3/d, which may increase by the factor of one hundred to 36 Mm^3/d until 2020. It is expected that most of the investment will go into the four north-eastern coastal provinces of Tianjin, Hebei, Liaoning and Shandong, where total water shortage is expected to reach between 16.6-25.5 billion m^3/a in 2010. Besides desalination of seawater, wastewater reclamation is a serious option under consideration [30].

1.5 Summary and conclusions

In a nutshell, 63% or about 28 Mm^3/d of the worldwide desalination capacity is produced from seawater sources. Of this water, 61% is produced by thermal processes. The MSF distillation process is almost exclusively used for the desalination of seawater, mainly in the Gulf countries. The RO process is the second most important process for treating seawater on a global scale, but the first choice in many countries outside the Middle East. 83% of the treated seawater is for municipal use.

Most of the desalinated water today is produced in industrial-sized facilities. These include the huge thermal distillation plants in the Middle East, with production capacities in single locations up to 1.6 Mm^3/d. The world's largest SWRO plant currently produces 330,000 m^3/d and a few SWRO projects up to 500,000 m^3/d are under development. The 50 largest SWRO plants account for almost half the worldwide production by SWRO.

79% of the global seawater desalination capacity is located in the Middle East, North Africa and Southern Europe, with 71% being located in the sea areas of the Gulf, the Red Sea and the Mediterranean Sea. Due to their enclosed nature, these sea areas are susceptible to any form of pollution, and desalination plants have already been classified as a main contributor to land-based pollution sources in the Gulf and Red Sea [2, 3, see also sections 3.4.1 to 3.4.3].

19% of the global desalination capacity is presently produced from brackish water sources and 5% from waste water sources, with 98% of the brackish water and 85% of the waste water being treated by membrane-based processes. Brackish water desalination is mainly used in North America – the only region where brackish water plays a more important role than seawater desalination – followed by the GCC states and Southern Europe. Waste water purification is mainly applied in North America as well, closely followed by East Asia and the GCC states. While the primary use of brackish water (similar to seawater) is community water supply, the main use for purified waste water is industrial use (including the power industry), followed by irrigation.

Although brackish water and waste water treatment offer a great future potential, desalination of seawater will remain the dominant process for some time. This is mainly because Saudi Arabia and the United Arab Emirates will continue to be the largest desalination markets in the foreseeable future, where seawater desalination plays a prominent role. MSF distillation will therefore continue to be the main desalination process, but will presumably lose further market shares to MED and SWRO. While thermal cogeneration plants, which produce electricity and water, predominate in the oil-rich countries of the Middle East, SWRO is usually the preferred process where cheap fossil energy or low value heat is not available. Consequently, most countries outside the Middle East that are now starting to consider seawater desalination choose the SWRO process.

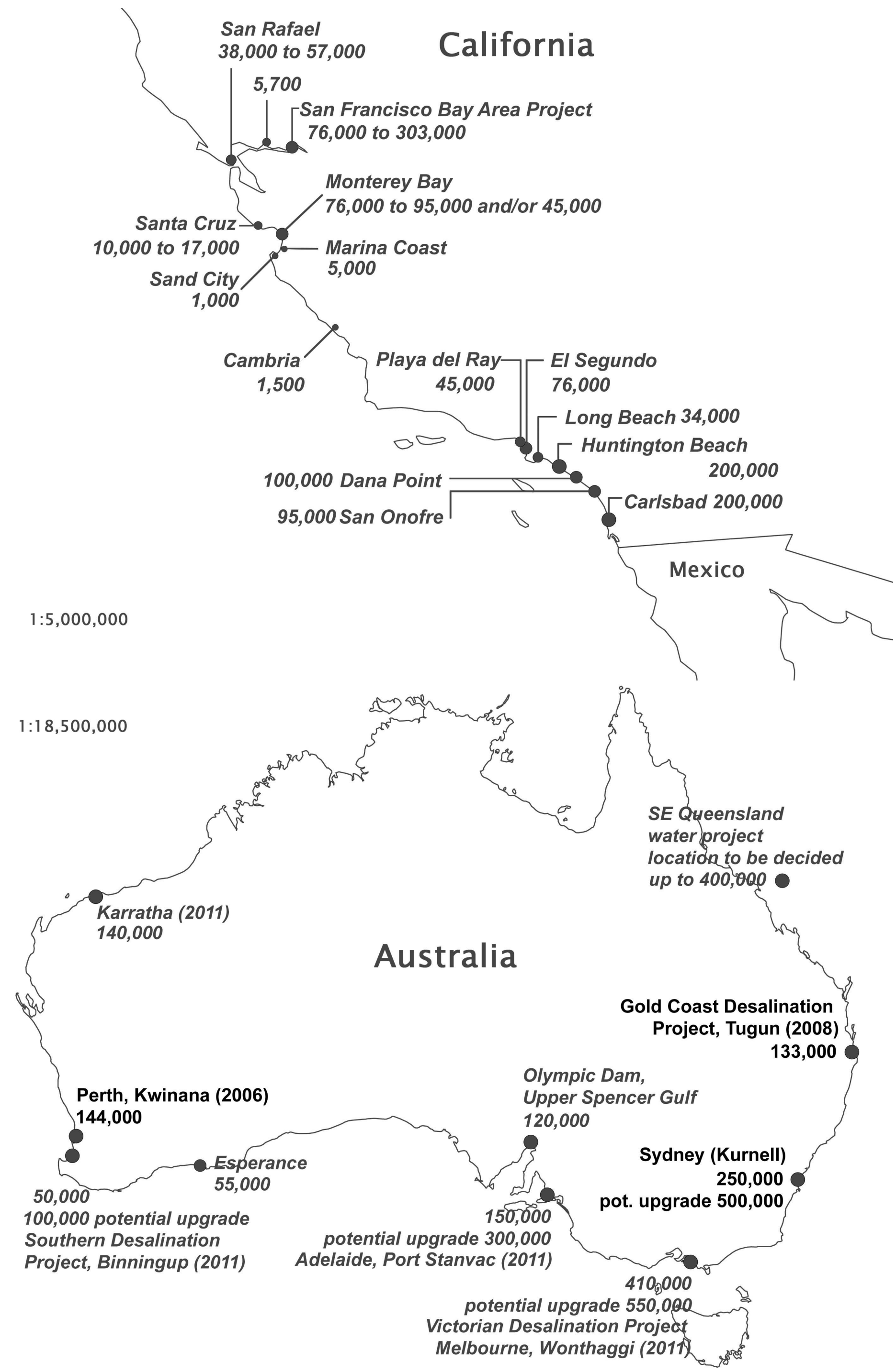

Figure 6: SWRO capacities in Australia and California in m^3/d (bold/black: in operation or construction, italic/grey: in planning or anticipation) [30, 34, 35].

With coastal population densities on the increase in many parts of the world − half of the world's population already lives within 200 kilometers of the ocean, and 70% of the world's mega cities are located near the coast [36] − the development potential for seawater desalination facilities is huge. However, as the need for desalination accelerates in many parts of the world, concerns are raised over potential negative impacts of desalination on the environment. The key concerns are summarized in chapters 3 to 5.

Seawater desalination processes | 2

2.1 Introduction

The environmental impacts of a desalination plant depend on its size and location as well as on the desalination process and its modus operandi. The latter has been extensively described in the literature [e.g., 5, 16, 17, 21, 22, 37]. The intention of this chapter is not to reproduce these textbook materials but to describe the different desalination processes with regard to environmental considerations, including the use of resources such as water, energy, land and materials, and the resulting emissions into water, soils and air. While the design of the intake, the pretreatment, the desalination process, and the outfall largely determine the impacts on the marine environment (chapter 3, page 57ff.), energy demand and air quality impacts (chapter 4, page 91ff.) mainly depend on the process type.

2.2 Seawater intakes

Seawater desalination plants can receive feedwater either through a *surface* water intake or a *subsurface* intake embedded in the seafloor or beach sediments. Surface intakes include the nearshore intakes of most distillation plants, which are often located directly at the shoreline, and the submerged intakes which are more common for large SWRO projects and which are typically located further offshore and in greater water depths. For SWRO plants, different types of subsurface intakes with either vertical or horizontal collectors have been tested or used successfully [5, 38–42, Figure 7], including:

- Vertical wells with a vertical caisson, usually drilled to a depth of 30 to 50 m into the permeable *onshore* sediments.
- Horizontal radial wells with a vertical caisson and horizontal radial collectors embedded in the permeable *onshore* sediments.
- Horizontal drains, drilled horizontally from a central point on land into the *offshore* sediments (horizontally drilled/directed drains, HDD).
- Infiltration galleries where the natural sediments are not sufficiently permeable, which consist of perforated pipes arranged in a radial pattern in the *onshore* sediments, and which are constructed by excavation of sediments, refilling with coarse, more permeable material, and covering by natural sediments.
- Seabed drains, similar to infiltration galleries but placed into the *offshore* sediments.

Parts of this chapter were based on:

S. Lattemann, M.D. Kennedy, J.C. Schippers and G. Amy. Seawater reverse osmosis: A sustainable and green solution for water supply in coastal areas? Submitted to Balaban Desalination Publications for a book on seawater desalination to be published in memory of Sydney Loeb.

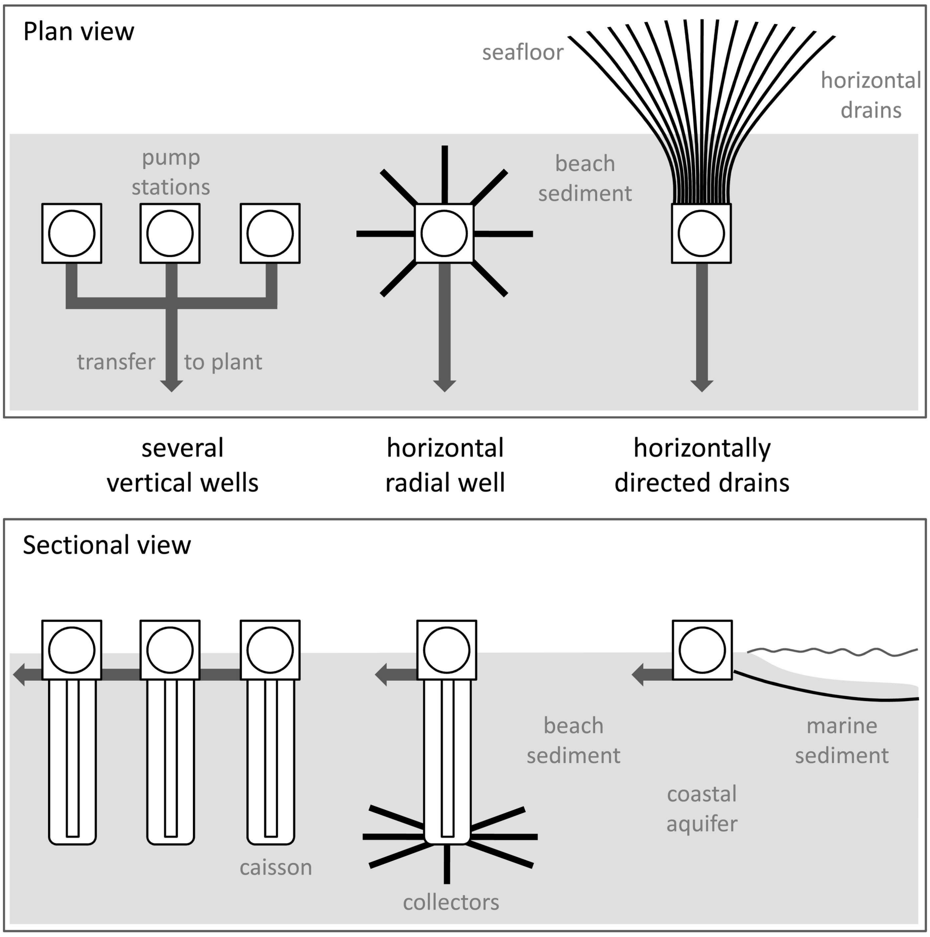

Figure 7: Subsurface intakes: vertical wells, horizontal radial well and horizontally drilled drains.

Surface intakes

Offshore submerged intakes, which are located at some distance from the coast and in greater water depth where marine life is less abundant, often produce a better feed water quality with lower contents of suspended solids and microorganisms than near shore intakes. They are the prevailing intake type for SWRO plants and are typically placed in 10-15 m water depth and 2-5 m above the seafloor (see Table 5 for examples). Depending on the seafloor bathymetry, this may require a distance of several hundred meters from the shore [22]. The seawater transmission pipeline from an offshore submerged intake to the shore can either be placed on or below the seabed, using open-trench or tunneling techniques. Intakes which are located directly at the shoreline, which is common for distillation plants but also seen in some SWRO plants, are often protected by a jetty or breakwater basin in order to reduce wave action and to allow suspended material to settle.

Surface intakes are usually equipped with a combination of screens to reduce the amount of debris and the number of organisms that are taken into the plant with the feedwater. Much of the advances in screen design stem from the power industry. State-of-the-art intake systems can effectively reduce the impingement of aquatic organisms against screens and the entrainment of organisms into the plant. The following screen systems are available [5, Table 1]. To prevent biogrowth on the intake structures, chlorination or antifouling paints with biocidal properties are commonly used [43].

Table 1: Intake screen systems [5].

Screen type	effective against
▸ Passive screens, which have no moving parts, operate with a very low velocity to mitigate impingement, and can be back-flushed with compressed air.	▸ impingement
▸ Modified traveling screens with water-filled lifting buckets that collect organisms and transport them to a bypass or trough (Ristroph screen), or traveling screens with attached baskets consisting of framed screen panels.	▸ impingement
▸ Louvers consisting of a series of vertical panels placed perpendicular to the intake, thereby creating a new velocity field that carries organism toward a bypass.	▸ impingement
▸ Fine-mesh screens of 5 mm or less that exclude larger eggs, larvae, and juvenile fish from the intakes, or cylindrical wedge wire screens with a mesh size up to 10 mm that dissipate the velocity so that organisms can escape the flow field.	▸ impingement, partially also prevents entrainment
▸ Marine life exclusion systems (see Figure 19, page 128) consisting of a water-permeable 'curtain' that completely surrounds the intake structure and is sealed against the seafloor and shoreline structures, preventing any organisms small or large from entering the system; due to a large surface area the water velocity through the curtain is up to 98% less than the velocity near the intake structure (currently tested in marine settings to examine its durability, susceptibility to fouling, and cleaning requirements).	▸ impingement, entrainment

For instance, the Thames Gateway desalination plant in East London with a capacity of 150,000 m^3/d will have an intake located at 150 m from the shore at a minimum submergence of 0.5 m. Water is withdrawn during low tide using seven submersible pumps, each fitted with a pair of 1.1 m diameter cylindrical copper-nickel screens having 3 mm openings, which are designed with a 0.15 m/s velocity and oriented parallel to the current. Ten to fifteen minutes before starting the intake pumps, an acoustic device is activated to produce a sound level of 160 dB and a frequency of 25-400 Hz, which is intended as a deterrent to fish entrainment, followed by a short air backwash to dislodge debris from the screen face [44].

Subsurface intakes

Because of the limited output capacity of beachwells and because of the lower recovery of SWRO systems compared to BWRO systems, a large number of single wells would be required for a large SWRO plant, which is difficult to realize [22]. Beachwells are therefore usually only considered for small SWRO plants with capacities ≤ 20,000 m^3/d. A feasible alternative to open intakes for large seawater desalination plants are horizontal drains, such as the Neodren system, which is installed from an onshore site by horizontal directional drilling into the seafloor sediments. Sufficient flow rates can be realized for all plant sizes depending on the number of drains installed. The technology is for example used in the SWRO plant in San Pedro del Pinatar in Spain [45, capacity of 65,000 m^3/d, Table 4], and was also considered in the planning studies for the Barcelona plant [46, 200,000 m^3/d] and other large plants in Algeria [47, capacities up to 500,000 m^3/d, Table 5], but not selected in the final design. For a 200,000 m^3/d plant, a maximum batch of 25 drains would be required, spaced at a distance of 2-3 m onshore (50-75 m total) and spreading out approximately 300 m offshore in a fan-like arrangement.

In Long Beach, California, an ocean floor demonstration system is currently being tested which combines seabed drains for the intake and a discharge gallery for the outfall (see also page 53). Both consist of perforated laterals placed under the ocean floor to collect or to discharge the water. The infiltration rates are 2 and 4 liters per m^2 and minute and the discharge rates are 5.3 and 6.9 liters per m^2 and minute. The intake in combination with a 100 µm and 5 µm cartridge filter were found to achieve sufficient DOC and turbidity removal to be used as feed water to a SWRO plant. The performance was comparable to effluent produced by a microfiltration system [41].

As subsurface intakes use natural sediments for pre-filtration, they can be described as a 'natural treatment system' or 'biofilter'. Favorable conditions for subsurface intakes are geologic formations with a high transmissivity and a certain sediment thickness, whereas unfavorable conditions include sediments with high volumes of mud and a low degree of 'flushing' [48]. Biofilters often produce a better feedwater quality than open intakes as the water is typically characterized by lower and less variable amounts of organic carbon (DOC and AOC), suspended solids, nutrients and microorganisms, and hence by a lower fouling potential [38]. This considerably reduces the pretreatment requirements in the 'engineered pretreatment system' following the biofilter. Further pretreatment is usually limited to acid and/or antiscalant addition followed by cartridge filtration. Extensive field experience shows that SWRO systems treating well water, with cartridge filtration as the only filtration step, operated successfully over the years [22].

Shallow beachwells sometimes contain significant amounts of suspended particles [22]. Moreover, water from beachwells is often anaerobic or anoxic and may contain hydrogen sulfide, as well as iron (II) and manganese (II) depending on the geology. Aeration may lead to the precipitation of ferric hydroxide, which causes turbidity, and the formation of manganese dioxide deposits on the membranes over time. In some SWRO plants operating on well water, iron (II) and manganese (II) were initially absent but increased over time, e.g., in Malta [49]. This appearance of insoluble salts in the feed water is a risk of beachwell intakes, which may necessitate the installation of granular media filters in case that feedwater conditions should deteriorate over the life-time of the project. Moreover, the water composition of well water with regard to sparingly soluble salts (barium and strontium sulphates and silicates) may differ substantially from surface seawater, which may necessitate the use of an antiscalant.

2.3 Desalination processes

2.3.1 Reverse osmosis

The RO process uses semi-permeable membranes and a driving force of hydraulic pressure to separate water from dissolved solids. Most membranes have a spiral wound configuration, in which several flat sheet membranes are formed into an envelope and wrapped around a central collecting tube (spirals in cross-sectional view). A pressure vessel typically contains up to eight of these membrane elements in series. As the feedwater flows through each subsequent element, part of the water is removed as permeate, and the salt content of the remaining feedwater increases along the pressure vessel. Pressure vessels are grouped in parallel in so called stages. The number of membrane elements per pressure vessels and the number of stages determines the recovery rate of the system, which is typically limited to 50% in SWRO plants with a single RO stage.

The main operational concerns in SWRO plants which need to be resolved by the pretreatment (Figure 8) are particulate fouling by suspended particles, biofouling by microorganisms caused by nutrients in the feedwater, organic fouling by dissolved organic matter, scaling by sparingly soluble inorganic compounds, and oxidation and halogenation by residual chlorine which is added during the pretreatment. The type and amount of pretreatment depends on the intake water quality and the desalination process. As surface intakes have to cope with more variable water quality due to seasonal weather conditions and algae blooms, pretreatment is generally more complex and extensive than for sub-surface intakes, which can achieve a silt density index (SDI) < 3 with single stage sand filters (without coagulant) or with cartridge filters only [50].

An overview on the main pretreatment alternatives for SWRO plants is given in Table 2 on page 28. The conventional pretreatment for SWRO plants with surface intakes includes shock chlorination to control marine growth in the intake system, followed by coagulation-flocculation and filtration to remove suspended solids and colloids, and dechlorination prior to the RO units. Sometimes additional screening, sedimentation or flotation is included as an initial pretreatment step. The two main alternatives to conventional pretreatment systems are a natural intake (page 24), which also compensates for some pretreatment steps in the conventional design, and UF or MF pretreatment, which is discussed in section 2.3.2 on integrated membrane systems (IMS) on page 39.

In most desalination plants, chlorine is added to control biogrowth on the intake screens, inside the intake pipe, and in the pretreatment line, with one or several dosing points along that line. The practice of maintaining a *continuous* concentration of 0.5-1 mg/l free residual chlorine inside the plant [15] has been replaced by *intermittent* chlorination in doses up to 10 mg/l in most plants today. Chlorination can be carried out daily, weekly or biweekly, depending on the site. Some plants also operate more successfully without chlorination, which breaks natural organic matter into biodegradable compounds that may increase biofouling in SWRO elements (Table 5).

Coagulation-flocculation is a combined process. Coagulation is the destabilization of the particle surface charge of small and colloid particles, which is followed by flocculation, i.e., the formation of larger flocs. In SWRO plants, mostly ferric chloride ($FeCl_3$) or ferrous sulfate ($FeSO_4$) are used as primary coagulants. Effective coagulation-flocculation requires intensive mixing to bring the coagulant in contact with the colloid particles. This is achieved downstream of the injection point by static mixers or flocculation chambers. The process can be enhanced by adding coagulant aids (long chain organic polymers). For example, the SWRO plants in Australia, Fujairah (UAE), Point

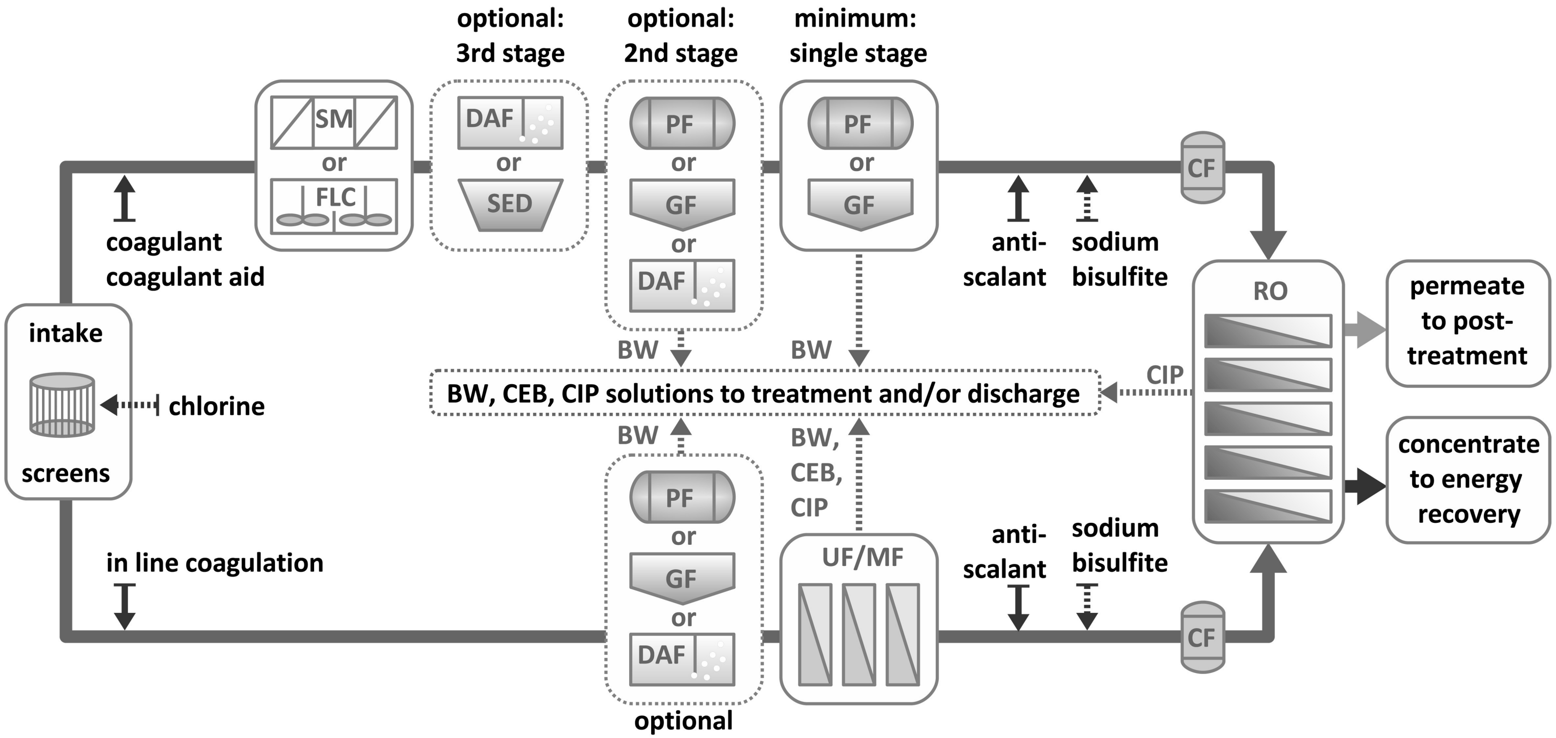

Figure 8: Simplified flow-scheme of a SWRO plant with conventional pretreatment (top) or UF/MF pretreatment (bottom). Abbreviations: SM: static mixer, FLC: flocculation chambers, DAF: dissolved air flotation, SED: sedimentation, PF: pressurized media filter, GF: gravity media filter, CF: cartridge filter, BW: backwash, CEB: chemically enhanced backwash, CIP: cleaning in place. Dotted lines show intermittent flows/doses [adapted from 22, 51–53].

Lisas and Barcelona reported the use of a coagulant aid in addition to $FeCl_3$ or $FeSO_4$ (Table 5). Cationic polymers can also be used as primary coagulants [22].

Filtration is either performed in pressurized vessels or gravity concrete chambers, which contain a single or dual medium, usually anthracite and sand. Filters are backwashed with either filtrate or concentrate. The frequency depends on the quality of the raw water and is usually at least once a day [50]. The backwash is either discharged into the sea or dewatered, and the sludge sent to a landfill (cf. section 3.3.3, page 71). Filters are typically arranged in a single or two stage configuration. Two stage media filtration is usually effective in producing a consistent feed quality, also during seasonal feedwater fluctuations [54]. In locations with high turbidity, natural organic matter, algae blooms, and hydrocarbon pollution, dissolved air flotation (DAF) can precede single or dual stage filtration or ultrafiltration to handle poor seawater quality [51]. Many SWRO plants have only a single stage pressurized (e.g., Perth) or gravity filter (e.g., Sydney, Tugun, Fujairah, Ashkelon, Hamma, Larnaca). Some combine a gravity with a pressurized filter (e.g., Chennai) or two pressurized filters (e.g., Algerian projects). In Tampa, a sand filter is followed by a diatomaceous earth filter, whereas a sedimentation step precedes the gravity filter in Point Lisas. In Singapore and Barcelona, a three stage pretreatment using DAF, gravity and pressurized filters has been implemented (Table 5). In the Gulf, some SWRO plants (Fujairah II, Hamriyah, Layyah) have recently incorporated DAF followed by UF or media filters to deal with the problem of extended periods of red tides [55].

The formation of inorganic scales, metal oxides and hydroxides, sometimes encountered in the tail elements of BWRO plants, does not present a problem with most seawater feeds, where the precipitation of sparingly soluble salts is less likely to occur due to the relatively low recovery rate of SWRO plants, which is usually limited to 50%, the high ionic strength of seawater, and the low concentration of bicarbonate ions in seawater [54]. Nevertheless, most SWRO plants use antiscalant, acid or a combination of both to avoid the risk of scale formation (Table 5). Some plants are also known to operate without any antiscalant, e.g., on the Cayman Islands or in the Mediterranean (Table 3).

The question is if polymer antiscalants are actually needed in SWRO systems. Laboratory studies indicate that the induction time of calcium carbonate, the main scalant in SWRO, is about 100 minutes, which suggests that scaling will not occur in SWRO systems with a residence time in the membrane units of only a few minutes [56]. This result needs to be verified by pilot studies and in full scale SWRO plants. If a second stage is necessary for boron removal, which requires that the pH is raised to about 10, antiscalants are needed to prevent the formation of magnesium hydroxide ($Mg(OH)_2$). SWRO plants with a single stage often use acid only, while plants with a second RO stage often use polymer antiscalant to maintain a higher pH for boron removal.

Chapter 3 (page 57) on marine environmental impacts gives a short description of each conventional pretreatment step as well as typical chemical dosing concentrations, which are evaluated with regard to their potential marine environmental impacts. Integrated membrane systems, which are an emerging alternative to conventional pretreatment systems in SWRO applications, are discussed in the next section on page 39.

► **Table 2** summarizes the pretreatment characteristics of the main pretreatment alternatives for SWRO, followed by plant-specific details for beachwells (**Tables 3, 4**), conventional pretreatment (**Table 5**), and UF pretreatment (**Tables 6, 7**).

Table 2: Pretreatment options prior to SWRO [50, 57].

Minimal conventional pretreatment after well intakes (see Tables 3 and 4 for specific plant details)	Full conventional pretreatment after surface intakes (see Table 5 for specific plant details)	UF/MF pretreatment after surface intakes (see Tables 6 and 7 for specific plant details)
▸ Chlorination-dechlorination typically not necessary ▸ Sand filters (if necessary, often without coagulant) ▸ Pretreatment against scaling (if necessary) ▸ Selective pretreatment to remove certain constituents such as dissolved iron, manganese, and sulfides, which, if oxidized, create particulates that can foul RO membranes (depending on well water properties). ▸ Cartridge filters	▸ Control of marine growth on intakes (see also page 64): continuously or intermittently, chlorine residual of 0.5-1.0 mg/l, dechlorination with sodium bisulfite ▸ Removal of suspended matter (page 71): coagulation-flocculation, granular media filtration use of coagulants, typically ferric salts, and coagulant aids, typically polyelectrolytes, pH adjustment to pH 6-7 ▸ Control of scaling (page 75): acid addition and/or antiscalant chemicals ▸ Cartridge filters as a final protection barrier against suspended particles and microorganisms	▸ Chlorination-dechlorination (often not necessary) ▸ Pretreatment against scaling (if necessary) ▸ Granular media filtration (if necessary)
Backwash	Backwash	Backwash
▸ generally the same procedure as for conventional pretreatment, but backwash frequency may be lower	▸ of media filters with filtered water from the media filters or with RO concentrate, may be enhanced with air, usually once a day (depending on raw water quality)	▸ of UF/MF membranes with UF/MF filtrate, optionally air enhanced backwash (AEB) or air scour, typically every 20-60 min. for 1-2 min. ▸ chemically enhanced backwash (CEB) of the UF/MF, usually with high chlorine levels and occasionally with acid or caustic solutions ▸ the regular backwash may simultaneously be a CEB with chlorine, or CEBs are carried out intermittently every 6 h, 24 h, every week etc.)
Cleaning in place (CIP)	Cleaning in place (CIP)	Cleaning in place (CIP)
▸ CIP of SWRO membranes generally the same procedure as for conventional, but cleaning frequency may be lower (1-2 times per year)	▸ CIP of SWRO membranes with alkaline and acidic solutions containing chemical additives such as oxidants or detergents, 4-12 times per year	▸ CIP of SWRO membranes generally the same procedure as for conventional, but cleaning frequency may be lower (1-2 times per year) ▸ CIP of UF/MF membranes, usually with chlorine, acid, caustic solutions

Table 3: SWRO plants with a beachwell intake.

Plant characteristics	Intake & pretreatment	Chlorine & SBS	Coagulant & sludge	Antiscalant & acid	Outfall design	Energy demand
Cayman Islands, North Sound [58] 6,000 m^3/d (data representative of all plants on the Cayman Islands)	deep wells spaced approx. 30 m apart, very good water quality (SDI<1), CF	none used	none used	none used	concentrate disposal into deep wells, S_{CC}=47.5	60 kW intake/pretreat. ≡0.2 kWh/m^3 RO permeate ≡0.1 kWh/m^3 feedwater at 50% recovery
Javea, Spain, Mediterranean Sea [59] 26,000 m^3/d, commissioned in 2002, 45% recovery	intake wells in riverbed (=seawater salinity), sand filters, CF	none used	none used, BW with Cc every 10-14 days, filter BW to clarifier, solids to landfill	only HCl and NaOH for cleaning once per year, w/out additional cleaning chemicals, neutralization before discharge with Cc		3 kWh/m^3 (incl. pretreatment and RO)
Sinai, Egypt, 11,000 m^3/d, commissioned in 2004, recovery: 35% (1st pass), 90 % (2nd pass), 33% (total)	wells 60-150 m deep, DMF, SDI<2.8, BW 1x/3 days (160 m^3), solids: landfill or mixed with Cc	none used	none used	8 mg/l SMHP, no acid	Cc: 20,400 m^3/d, S_{CC}=63, nearshore pipe (50 m at sea, -1 m below surface)	energy recovery turbine, 8.5 kWh/m^3 total energy, 1.3 kWh/m^3 intake/pretreat. (due to oversized motors) 10 kWh for cleaning

Table 4: SWRO 'twin plants' with beachwell (top row) and conventional (bottom row) pretreatment.

Plant characteristics	Intake & pretreatment	Chlorine & SBS	Coagulant & sludge	Antiscalant & acid	Outfall design	Energy demand
San Pedro del Pinatar I, Murcia, Spain [60] 65,000 m^3/d, recovery: 45%	Neodren wells, sand filters (silica), CF, SDI 0.4 TOC 0.2-0.3 mg/l	none used	none used, BW with Cc (1000 m^3), tank for sedimentation, sludge treated	0.9-1.2 mg/l, occasionally some acid	S_{CC}=54	4.22 kWh/m^3 total including: >3 kWh/m^3 specific demand 0.40 kWh/m^3 intake/pretreat.
San Pedro del Pinatar II, Murcia, Spain [60] 65,000 m^3/d, recovery: 45%	open intake, two stage DMF, CF, SDI 4.5, NTU 0.5-6 TOC 1 mg/l	none used	4 mg/l $FeCl_3$, no coag. aid, BW with Cc (750-1700 m^3 every 8 h), tank for sedimentation, sludge treated	1 mg/l, pH 6.5	S_{CC}=54	3.90 kWh/m^3 total including: 3.10 kWh/m^3 specific demand 0.35 kWh/m^3 intake/pretreat. (≡0.175 kWh per m^3 feedwater at 50% recovery) 0.30 kWh/m^3 post-treatment 0.10 kWh/m^3 auxiliary equip.

Abbreviations: BW: backwash, CF: cartridge filters, DMF: dual media filter, SBS: sodium bisulfite ($NaHSO_3$), Cc: concentrate, S: salinity, S_{CC} concentrate salinity, SDI: silt density index, TOC: total organic carbon, NTU: nephelometric turbidity unit, SHMP: sodium hexametaphosphate

Table 5: SWRO plants with conventional pretreatment (completed after year 2000, capacities 20,000-330,000 m^3/d).

Plant characteristics	Intake & pretreatment	Chlorine & SBS	Coagulant & sludge	Antiscalant & acid	Outfall design	Energy demand
Sydney, Australia [61–63] commissioning in 2009, (design parameters given) 250,000 m^3/d, provides 15% of Sydney's water, two pass SWRO	4 risers 200-300 m offshore, tunnel to plant, intake velocity <0.1 m/s single stage DMF (gravity filters were used in pilot tests; flocculation and second stage filtration was not found to be necessary)	intermittently 10 mg/l NaOCl at intake 3 mg/l at plant intake 3-30 mg/l SBS	3 mg/l $FeCl_3$ and 0.2 mg/l polyDADMAC or 6 mg/l $FeCl_3$ without polyDADMAC, sludge thickened in lamella settlers, dewatered by centrifuge, special landfill due to salt content, supernatant discharged with Cc	antiscalant not anticipated (if needed: 1.5 mg/l), 55 mg/l H_2SO_4 (pH 6.9), or pH 6.5 with more H_2SO_4	4 risers 200-300 m offshore, tunnel from plant, 2 nozzles per riser, S_{CC}=65, S<1 ppt above ambient at the edge of the near field mixing zone in 50-75 m distance	DWEER (97% recovery) 4.2 kWh/m^3 (max. S) 3.6 kWh/m^3 (average S) compensated by 67 wind turbines (132 MW) vs. 42 MW as required by the plant
Tugun, Gold Coast, Australia [64] commissioned in 2009, (design parameters given) 125,000 m^3/d, two pass RO, 41% total recovery	twin risers 1 km offshore, 20 m water depth, intake tunnel to plant, intake velocity <0.1 m/s gravity DMF	intermittently 10 mg/l NaOCl at intake screen, dechlorination with 3 mg/l SBS prior to RO	18 mg/l $FeSO_4$ (4-20 mg/l) 0-0.4 mg/l polyelectrolyte, lamella-plate thickeners, dewatered by centrifuge, sludge (15-20% solids) to landfill, supernatant discharged with Cc, system also treats CIP wastes	2 mg/l antiscalant (1st pass) 35 mg/l H_2SO_4	diffuser 1 km offshore 287 m long seabed pipeline with 14 nozzles, outfall tunnel from plant to diffuser	DWEER (on 1st pass), variable speed motor drives at pumps throughout the plant
Perth, Australia [65–72] commissioned in 2006, 144,000 m^3/d, provides 17% of Perth's water, 45% recovery (1st pass), 95% recovery (2nd pass)	364,000 m^3/d intake, one 8 m diameter inlet, 10 m water depth, 200 m offshore, buried glass reinforced pipe (diameter 2.3 m), one pressurized DMF	intermittently in intervals of 1-2 weeks, NaOCl at intake, dechlorination with SBS	$FeCl_3$ and polyDADMAC (organic coagulant), sludge thickened/centrifuged, supernatant, neutralized CIP chemicals, and plant flushing water discharged along with Cc	antiscalant (1st pass) H_2SO_4	Cc: 216,000 m^3/d diffuser 470 m offshore, 200 m length, 1.2 m diameter, 40 nozzles, buried pipeline, S<1.2 units above ambient within 50 m and S<0.8 within 1 km distance	ERI-PX (97%, 1st pass), variable frequency pumps 2.3 kWh/m^3 (specific) and 3.2–3.5 kWh/m^3 (total plant), compensated by 48 wind turbines (82 MW) vs. 24 MW required by plant
Tuas, Singapore [52, 73] commissioned in 2005 136,000 m^3/d, 38.5% recovery	open offshore intake, DAF, gravity DMF, pressurized DMF	no information	$FeCl_3$, BW blended with Cc, occasionally visible plume	H_2SO_4 to pH 6-6.5	Cc: 217,000 m^3/d, S_{CC}=57, discharge offshore (120 m / - 3 m), pipe laid on seabed, discharge angled for better mixing	4.3 kWh/m^3
Fujairah, UAE [73, 74] Gulf of Oman, commissioned 2003, 170,500 m^3/d, recovery: 43% (1st pass) 90% (2nd pass), 41% (total)	380 m offshore, 10 m below sea surface, gravity DMF	intermittently electro-chlorination at the intake (max. 4 mg/l) and in the coagulation chambers (max. 10 mg/l) for max. 4 hours	up to 5 mg/l $FeCl_3$, up to 1.5 mg/l coagulant aid BW blended with Cc, plume not visible	up to 25 mg/l H_2SO_4	Cc: 225,000 m^3/d, S_{CC}=70.2, diluted with PPCW to 44.2 before discharge from channel at shoreline	Pelton turbine (89% recovery) 3 kWh/m^3 (1st pass) 3.8 kWh/m^3 (total RO) 4.5 kWh/m^3 (total plant) 0.4 kWh per m^3 filtrate for pretreatment

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Plant data	Intake/ pretreatment	Chlorine	Coagulant	Antiscalant	Outfall	Energy
Tampa, Florida [75, 76] 95,000 m^3/d, providing 10 % of Tampa Bay water supply	166,000 m^3/d intake from PPCW, coagulation, flocculation, single stage sand filtration, followed by DE filters	chlorine dioxide, dechlorination with SBS	$FeSO_4$	adjustment to pH 6.5-6.8	Cc mixed with PPCW (ratio 70:1), reducing S to 1-1.5% above ambient, discharge within natural variability of S=16-32	Pelton turbines
Point Lisas, Trinidad and Tobago [52, 73, 77] commissioned in 2002, 119,000 m^3/d, two pass, recovery: 47% (1st pass), 90% (2nd pass), 41% (total)	open intake with high turbidity, flocculation, sedimentation, single stage gravity DMF	5-10 mg/l chlorine added at plant inlet to obtain a residual after the flocculation basins, chlorine soakings of deep bed filters	0.14-0.17 mg/l $FeCl_3$, cationic polymer, filter BW settled, sludge to landfill, supernatant blended with Cc	no information	Cc: 119,000 m^3/d, S_{CC}=58, diluted with industrial CW, discharge at shoreline	3.8 kWh/m^3
Barcelona, Spain [46, 78, 79], commissioned in 2009 200,000 m^3/d, provides 20% of Barcelona's water, recovery: 45% (1st pass), 85% (2nd pass), 44% (total)	offshore submerged intake (2.2 km, -25 m), 3 stages: 1. flocculation, DAF 2. gravity DMF 3. pressurized DMF	shock chlorination at 3 points (offshore, pumping station, before first pretreatment stage), dechlorination with SBS	$FeCl_3$ and coagulant aid, sludge thickening and dewatering, <1 mg/l suspended solids in the final Cc	antiscalant H_2SO_4	dilution with wastewater (1:1 ratio), discharge 3 km offshore	ERI-PX 2.88 kWh/m^3 (1st pass)
Carboneras, Spain [60] 120,000 m^3/d, operated at 10% of nominal capacity recovery: 43%	open intake, DMF	not used	1 mg/l $FeCl_3$, no coag. aid, BW with Cc, sludge treated	1 mg/l antiscalant 22 mg/l H_2SO_4		3 kWh/m^3 specific demand 4.1 kWh/m^3 total demand
Alicante II, Spain [60] 65,000 m^3/d, recovery: 43%	mixed open and well intake (Neodren), each providing 50% of feed DMF, CF; SDI 2.2, NTU 0.01, TOC 1 mg/l	none used	none used, BW with Cc, tank for sedimentation, sludge treated	1.1 mg/l	S_{CC}=57, dilution 6:1 before discharge via 1 km pipe	3.7 kWh/m^3 total including: 2.4-2.5 kWh/m^3 specific d. 0.5 kWh/m^3 intake/pretreat.
Ashkelon, Israel [80–82] Mediterranean Sea, commissioned in 2006, 320,000 m^3/d	3 offshore submerged intakes (1 km, -7 m), buried pipelines (1.6 m diameter, 13 m sediment depth), gravity DMF	no chlorine	dose: 0.5-3.0 mg/l as Fe discharge: 40-50 mg/l in peaks for 10-20 min./hour, equal to average of 1.8 mg/l, discharge standard is 2.0 mg/l reduction goal is 0.3 mg/l, no coagulant aid	phosphonate antiscalant, HCl	S_{CC}=73.5, reduced to 1-5% above ambient by PPCW, monitoring has shown a 3% salinity increase above ambient in 500 m distance, and 1% increase for a few km	3.5 kWh/m^3 on average
Hamma, Algeria [52, 83] commissioned in 2006, 200,000 m^3/d, provides 25% of Algier's water 44.5% recovery	1. flocculation, sedimentation 2. gravity DMF	no information	no information	no information	no information	no information

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Plant data	Intake/ pretreatment	Chlorine	Coagulant	Antiscalant	Outfall	Energy
Skikda, Cap Djinet, Beni Saf, Hounaine and Mostaganem, Algeria [52] 100,000-200,000 m^3/d, all commissioning 2008/09	two pressurized DMF	no information	no information	no information	no information	no information
Dhekelia, Cyprus [84] 20,000 m^3/d, two pass	open intake, 600 m long intake pipe embedded in or anchored to the sea bed, DMF	intermittent use	$FeCl_3$	H_2SO_4 to pH 7	no information	pressure exchangers (96% recovery), frequency converters, high pressure pumps (86% efficiency), 3.6-4.6 kWh/m^3
Larnaca, Cyprus [85] 54,000 m^3/d, two pass, 50% recovery	open intake, 1 km long pipeline, rotating screen, flocculation, gravity DMF	no chlorine use	no information	no information	no information	4.4 kWh/m^3 on average
Southern Europe I (2002) 120,000 m^3/d 45 % recovery [73]	open sea, pressurized DMF	no information	$FeCl_3$, BW and Cc blended with PPCW	no information	Cc: 146,667 m^3/d, S_{CC}=71, diluted with PPCW, discharge offshore open sea (150 m / -13m)	4.08 kWh/m^3
Southern Europe II (2006) 65,000 m^3/d 53 % recovery [73]	open intake, two stage pressure filter	no information	$FeCl_3$, BW blended with Cc	no information	Cc: 57,642 m^3/d, S_{CC}=83, discharge offshore open sea (5,100 m), submerged diffuser	4.3 kWh/m^3
Middle East I (2001) 90,900 m^3/d 35 % recovery [73]	open sea, gravity DMF	no information	$FeCl_3$, BW blended with Cc	no information	Cc: 168,800 m^3/d, S_{CC}=69.2	7.5 kWh/m^3
Chennai, India 100,000 m^3/d [52] commissioning in 2009	one gravity DMF, one pressurized DMF	no information	no information	no information	no information	no information
Literature review (plants before 2000) [15]		0.5-1 mg/l continuously, 3 mg/l SBS	0.8-4.5 $FeCl_3$ or $FeSO_4$ 0.2-0.5 coagulant aid	1.5-3 mg/l polymer antiscalant (e.g., PAA) or 2-5 mg/l phosphate antiscalant (e.g., SHMP) or 30-100 mg/l H_2SO_4 (pH 6-7)		

Abbreviations: BW: backwash, DE: diatomaceous earth, SBS: sodium bisulfite ($NaHSO_3$), CIP: cleaning in place, Cc: concentrate, S: salinity, S_{CC} concentrate salinity, DMF: dual media filtration, DAF: dissolved air flotation, PPCW: power plant cooling water, SHMP: sodium hexametaphosphate, PAA: polyacrylic acid, DADMAC: diallyldimethylammonium chloride, $FeCl_3$/$FeSO_4$: ferric chloride, ferrous sulphate, NaOCl: sodium hypochlorite, Cl_2: chlorine, ClO_2: chlorine dioxide, H_2SO_4: sulfuric acid, DWEER: dual work exchanger energy recovery, PX: pressure exchanger

Table 6: Operational SWRO plants with ultrafiltration (UF) or microfiltration (MF) pretreatment.

Plant characteristics	Chlorine & SBS	Coagulant	Antiscalant & acid	CEB/CIP	Remarks
Outside feed – in housing and pressurized, mostly PVDF (Pall/Asahi, Dow/Omexell, Memcor), some PS and PAN					
Yueqing, Zhejiang, China Yueqing power station [52] 21,600 m^3/d RO	not specified	0.3-1.5 mg/l $FeCl_3$, flocculation, sedimentation	not specified	not specified	driver: reliability, difficult feed water MF: Asahi
Penghu, Taiwan [52] 13,000 m^3/d RO	no chlorine	no coagulant	not specified	not specified	open intake, driver: difficult feed water UF: Dow
Wang Tan Power Station, China [52, 86] 9,600 m^3/d UF 3,120 m^3/d RO	2 mg/l Cl_2 dose resulting in 0.5 mg/l Cl_2 residual, SBS	no coagulant	antiscalant NaOH (prior to 2nd pass)	15 mg/l NaOCl every backwash (1x/hour) (equivalent to 0.5 mg/l cont.) CIP: 0.05% NaOH, 0.2% NaOCl, followed by 0.36% HCl, 1x/year	UF operated at 50% flux; disc filter prior to UF; UF selected after pilot tests with UF and CP due to a consistent filtrate quality and lower operating and land use costs despite a wide ranging raw water quality, open intake industrial area, UF: Dow
Magong, Taiwan [87] 23,000 m^3/d UF 5,500 m^3/d RO	no chlorine	no coagulant	antiscalant	BW: 1x/hour (with air scrub) no CEB, only CIP 1x/month, 2%w/w $H_2C_2O_4$ equal to 20,000 mg/l followed by 0.2%w/w NaOCl (2000 mg/l) (equal to a continuous dose of 0.4 and 4.0 mg/l respectively)	CP since 2002, replaced by UF in 2008, open intake and self-clean filters (130 µm), SDI_{15} and $MFI_{0.45-15}$ < 2.1 achieved despite poor feed quality (SDI_5 = 20), UF: Dow
Outside feed – uncontained, submerged and vacuum driven, mostly PVDF (Zenon/Zeeweed, Siemens/Memcor), some PAN					
Zhejiang, China [52, 88] 76,800 m^3/d UF 34,560 m^3/d RO	NaOCl	4-10 mg/l Fe, lamella sedimentation	not specified	not specified	open intake, coagulation, flocculation, clarification prior to UF to remove organics/floatable substances, UF: Zenon

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Plant characteristics	Chlorine & SBS	Coagulant	Antiscalant & acid	CEB/CIP	Remarks
Inside feed − always in housing, always pressurized, mostly PES (Norit, Hydranautics, Inge membranes), some PS (Koch) and CA (Aquasource)					
Palm Jumeirah, UAE [89] 64,000 m^3/d RO (2 x 32,000 m^3/d RO) (2 x 92,000 m^3/d UF)	1.5-25 mg/l ClO_2 1x/day for 15-30 min, dechlorination (SBS) in case of hypochlorite break through (not expected in normal operation)	0.3 mg/l as Fe ($FeCl_3$ used)	1-2 mg/l antiscalant, shock dose with 275 mg/l H_2SO_4 to pH 3.5, 1x/day for 20 min (design value, not known if actually employed)	CEB: 200 mg/l Cl_2 1x/day CIP: 0.5% oxalic acid + 0.25% ascorbic acid once every 9 months	chemical wastes neutralized by acid/alkali, CIP and BW mixed with RO reject and tertiary sewage before discharge, SWRO CIP anticipated once every 18 months, UF: Norit
Colakoglu, Turkey [90, 91] 6,700 m^3/d	not specified	'coagulant' (not further specified)	not specified	CEB: acid (every day) CIP: 1x/6 month	cites 2 more plants (Palm Jumeirah; Turkey) and several pilot plants where NTU<0.1 and SDI_{15} < 3 were met, UF: Norit
Qingdao power plant, China 5,600 m^3/d RO [52]	not specified	coagulation, flocculation, lamella sedimentation	not specified	not specified	open intake, UF selected because of reliability, difficult feed water; UF: Norit
Qingdao power plant, China 13,000 m^3/d RO [52]	not specified	coagulation/ flocculation	not specified	not specified	UF selected because of reliability, difficult feed water, UF: Hydranautics/Hyflux
Jeddah, Saudi Arabia [92, 93] 56,500 m^3/d UF 25,500 m^3/d RO	CaOCl shock chlorination at intake, dechlorination with SBS prior to RO	design dose of 1 mg/l as Fe is not necessary, no coagulant aid used either	H_2SO_4 to pH 6.7 antiscalant	20 mg/l Cl_2 every 8-20 hours, CEB with H_2SO_4	2 stage media filters prior to UF, stable operation without $FeCl_3$, combination of BW and CEB sufficient, one CIP (of UF and RO) in 6 months, UF: Hydranautics
Rome, Italy [94] 23,500 m^3/d UF	not specified	none, $FeCl_3$ use optionally	not specified	20 mg/l Cl_2, 1x/day CEB: HCl, 1x/three days	open intake, DAF prior to UF, UF filtrate treated by UV prior to the SWRO, UF: Inge
Liaoning, China [95] 28,000 m^3/d	0.3-1.0 mg/l residual Cl_2	$FeCl_3$ or $FeSO_4$ and anionic flocculant if needed	not specified	CIP: 1x/month	coagulation, sedimentation prior to UF, UF backwash discharged to sea, UF: Koch
Other					
Fukuoka, Japan [96] 50,000 m^3/d RO	0.5-1.0 mg/l Cl_2 residual 1x/week (of UF and RO)	not specified	acid	not specified	subsurface (infiltration) intake prior to UF; 60% recovery SWRO (HF cellulose triacetate)
Pinghai power plant, China 5,600 m^3/d RO [52]	not specified	coagulation/ flocculation	not specified	not specified	UF selected because of reliability, difficult feed water
Cyprus 20,000 m^3/d RO [52]	no chlorine	no coagulant	not specified	not specified	UF selected because of ease of operation
Addur, Bahrain 37,850 m^3/d RO [52, 97]	not specified	not specified	not specified	not specified	stable performance, SDI requirements met, single stage media filtration (sand)

Abbreviations: HF: hollow fiber, CP: conventional pretreatment, DMF: dual media filtration; GMF: granular media filtration, DE: diatomaceous earth, DAF: dissolved air flotation, AEB: air enhanced backwash, CEB: chemical enhanced backwash, CIP: cleaning in place, SBS: sodium bisulfite; $FeCl_3$/$FeSO_4$: ferric chloride, ferrous sulphate, PFS: polyferric sulphate, NaOCl: sodium hypochlorite, Cl_2: chlorine, ClO_2: chlorine dioxide, H_2SO_4: sulfuric acid, oxalic acid (HOOC-COOH): strong dicarboxylic acid, TOC: total organic carbon, DOC: dissolved organic carbon, NTU: nephelometric turbidity unit, SDI_{15}: 15-min. silt density index, UV: ultraviolet light, CA: cellulose acetate, PES: polyethersulfone, PS: polysulfone, PVDF: polyvinylidene fluoride, PAN: polyacrylonitrile

Table 7: Pilot studies of ultrafiltration (UF) or microfiltration (MF) pretreatment prior to SWRO.

Pilot characteristics	Chlorine & SBS	Coagulant	Antiscalant & acid	CEB/CIP	Remarks
Outside feed – in housing and pressurized, mostly PVDF (Pall/Asahi, Dow/Omexell, Memcor), some PS and PAN					
MF-SWRO pilot in 9 locations 15-360 days [98]	not specified	$FeCl_3$ used in 3 locations	not specified	10 mg/l Cl_2 every 20-40 min. CEB: 500 mg/l Cl_2 1x/day CIP: caustic, chlorine, acid	high filtrate quality (NTU<0.1, SDI<1.3-2) in all locations regardless of feed quality, MF: Pall Aria
MF-SWRO pilot Med. Sea 1 year [99]	not specified	(1): no chemicals (200 days) (2): 2 mg/l $FeCl_3$	(2): HCl (pH 6.8)	(2): 10 mg/l NaOCl or 25 mg/l H_2O_2 every 17-19 min., air scour	better results with chemicals (100% SDI<2); H_2O_2 less effective than NaOCl for CEB, MF: Pall Microza
MF-SWRO pilot CP with coagulation, sedimentation, filtration, Florida, 1 year [100]	MF: filtrate treated with UV + chlorine residual of 1-2 mg/l for 30 min. 1x/day CP: with and without chlorine, residual of 2-3 mg/l	$FeCl_3$ MF: 0-3.5 mg/l as needed CP: 0-25 mg/l and 0-4 mg/l polymer	antiscalant, pH to 5.8 if needed	MF: CEB: 500 mg/l NaOCl 1x/day CIP: 1000 ppm NaOCl, 1% NaOH, 1% citric acid 1x/month	CP did not always meet desired NTU and SDI; MF (recommended): BW: 4x/h, 30 sec, 1 l/sec BW/air scrub: 4x/h, 60 sec, 0.44 l/sec CEB: 1x/day 500 mg/l NaOCl, CIP: 12x/year UV disinfection of filtrate, MF: Pall Microza
UF-SWRO pilot, Zhejiang, China 80 days [101]	(1) no chlorine (2) no chlorine (3) 6 mg/l NaOCl	(1) without coagulant (2) 2.4 mg/l PFS (as Fe) (3) 2.4 mg/l PFS (as Fe)	not specified	AEB / CEB: 20 mg/l NaOCl for 60 sec. every 30 min.	UF filtrate with very low turbidity, which improved even further with coagulation and chlorination, UF: Dow/Omexell-SFP
UF-SWRO pilot, Qingdao, China 5 months [102]	not specified	0-1.5 mg/l $FeCl_3$	not specified	CEB A: 500 mg/l NaOCl CEB B: 400 mg/l HCl CEBs every 12, 24 or 120 h CIP: 20,000 mg/l oxalic acid, 1000 mg/l NaOH, 2000 mg/l NaOCl (3 x in 5 months)	optimized design: at 60 $l/m^2/h$: CEB A every 120 h, at 80-85 $l/m^2/h$: CEB A every 12 h, CIP every 4-6 weeks, CEB B less effective; UF: DOW
DOW UF guidance manual, emprirical values [103]	0.5 mg/l continuously		not specified	CEB 1 (alkali): 0.1% NaOCl + 0.05% NaOH CEB 2 (acid): 0.1% HCl or 2% citric acid or 2% oxalic acid CIP: 2x CEB concentrations except for citric/oxalic acid	CEB frequency: every 1-3 months, CEB duration: BW + 10 min. soak CEB volume: 1-1.5 m^3/h CIP frequency: if TMP increase >1 bar CIP duration: 2 h recycle and soak
UF-SWRO pilot France, Med. Sea 4 months [104]	not specified	not specified	not specified	CEB with air, with and w/out Cl_2 every 40-45 min.	CEB with Cl_2 needed to prevent UF fouling; UF filtrate SDI_{15} of 1-2; UF removed TOC only slightly, but no SWRO fouling was observed, UF: Polymem (PS-based, outside-in)

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Pilot characteristics	Chlorine & SBS	Coagulant	Antiscalant & acid	CEB/CIP	Remarks
Outside feed – uncontained, submerged and vacuum driven, mostly PVDF (Zenon/Zeeweed, Siemens/Memcor), some PAN					
MF-SWRO pilot UF-SWRO pilot GMF-SWRO pilot San Francisco Bay [105]	not specified	MF/UF: with and without coagulant (10 mg/l) GMF: coagulation, flocculation	not specified	not specified	all: acceptable turbidity and SDI; MF/UF: TOC reduced by 50% with in-line coag. (only 16% removed without); GMF: TOC reduced by 35%; UF/MF: Memcor and Zenon in parallel
Pilots with UF, DE precoat, and high settling Tampa Bay, Florida 3 months [50]	not specified	not specified	not specified	not specified	UF significantly outperformed the other treatments, consistently producing feed water with an $SDI_{15}<1$ regardless of feed turbidity; UF: Zenon
UF-SWRO pilot GMF-SWRO pilot Sydney, Australia [106]	not specified	UF: no coagulant GMF: $FeCl_3$	not specified	not specified	UF: better microorganisms removal, but $SDI_{15} > 3$ for 2 months, no coagulant use; GMF: reliable performance ($SDI_{15} < 3$) and better TOC removal; UF: Zenon Zeeweed
MF-SWRO pilot GMF-SWRO pilot Med. Sea [107]	not specified	MF: not specified GMF: 6 mg/l $FeCl_3$ and 0.15 mg/l coagulant aid	MF: acid (pH 6.8) GMF: acid (pH 6.8)	MF: 100 mg/l Cl_2, 1x/day	MF: better SDI_{15} and microorganisms removal but worse in DOC removal than GMF, MF: immersed outside-in
Inside feed – always in housing, always pressurized, mostly PES (Norit, Hydranautics, Inge membranes), some PS (Koch) and CA (Aquasource)					
UF-SWRO pilot Trinidad 100 days [108]	not specified	1.3 mg/l $FeCl_3$ (as Fe)	not specified	CEB: NaOCl / acid (pH 2.5) altern. every 6 h	filtrate SDI of 1-2 despite highly variable feed SDI and without any pre-treatment to the UF; coagulation decreased the SDI even further; UF: Norit
UF-SWRO pilot 2 stage DMF-SWRO East-/West Coast USA several runs, 4-6 weeks [109]	no chlorine	$FeSO_4$ UF: 1.5 mg/l as Fe DMF: 0.5 mg/l as Fe	not specified	UF: CEBs with NaOCl every 6-18 h HCl 1x/day	DMF: poorer performance on East Coast, both UF and DMF failed to properly handle the West Coast waters; UF: Norit
UF-SWRO pilot CP Texas, USA one year [110]	no chlorine	$FeCl_3$ UF Hydran.: 0.1-0.25 mg/l UF Norit: none CP: 6-7 mg/l	not specified	Norit: CEB 1: HCl every 3 h, CEB 2: 200 mg/l NaOCl every 12 h	Hydranautics at 102 $l/m^2/h$ (90-92% recovery), Norit at 85 $l/m^2/h$ (93-94% recovery), RO cleaning every 6 months with UF vs. every 6 weeks with CP, life cycle costs 3-8% cheaper with UF; UF: Hydranautics and Norit

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Pilot characteristics	Chlorine & SBS	Coagulant	Antiscalant & acid	CEB/CIP	Remarks
MF-SWRO pilot precoat filters Tampa Bay, Florida [111]	not specified	not specified	not specified	not specified	both systems: SDI_{15} <1, MF needed more frequent backwashes than anticipated; precoat favored (more proven technology, lower cost) with coag.-flocc. ($FeCl_3$) and ClO_2; MF: Hydranautics Hydrasub
UF-SWRO pilot Tampa Bay, Florida 2 months [112]	not specified	not specified	not specified	not specified	UF allowed a higher flux of 20.4 $l/m^2/h$ and 65% recovery in the SWRO; no cleaning necessary in test period; UF: Hydranautics
UF-SWRO pilot Jeddah Port, Red Sea 2 years [112]	not specified	with and without $FeCl_3$ 0-0.7 mg/l as Fe	with and w/out acid (pH 6.7) with and w/out DMF prior to UF	AEB every 15-60 min. CEB: 20 mg/l Cl_2 every 6 h for 2 min. later 50 mg/l Cl_2 with caustic (pH 12.5) and acid (pH 1.5-2)	best treatment: Fe-dose of 0.3 mg/l with acid; UF with DMF can easily cope without $FeCl_3$; SDI averaged 1 after AEB was correctly used; acid or caustic CEB only necessary when no Fe-dosing was employed; UF: Hydranautics
UF-SWRO DMF [113, 114]	not specified	UF: 0.2 mg/l as Fe DMF: 0.7 mg/l as Fe	not specified	not specified	coagulant not needed in UF with good feed water but can control UF fouling and provide some degree of organics removal; UF/MF can reduce CIP of SWRO from 3-8 to 1-2 times per year; UF: Hydranautics
MF/UF-SWRO manuals, emprirical values [115, 116]	optional, intermittent to 1-5 mg/l	0.5-1 mg/l $FeCl_3$	not specified	CEB: 50 mg/l Cl_2 6x/day CIP 1x/30-60 days	Based on experience with fluxes of 75-81 $l/m^2/h$; UF: Hydranautics
UF-SWRO pilot Addur, Bahrain 16 months [117]	chlorine 1 mg/l in winter, 2 mg/l in summer	0.25 mg/l as Fe (found to be sufficient)	not specified	CEB: 50 mg/l NaOCl every 2-3 h, CIP: NaOH (pH 12), citric acid (1%) if necessary	rapid biofouling of SWRO with CP; pilot UF showed stable operation; heavy fouling of UF without chlorination; UF: Inge Multibore
UF-SWRO (open intake), UF-SWRO (well) SWRO (well only) Pacific USA 2 months [118]	UF (open): not given wells: NaOCl, SBS	UF (open): 0.375 mg/l (as Fe) UF (well): 5 micron filters well only: 5 micron filters	not specified	UF (open): no CEB, CIP frequency >30 days wells: CEB 30 mg/l NaOCl 1x/day, CIP at end of test	UF on a well may not have economic benefits (operating directly on well feed had minimal impact on RO performance); UF operated on surface feed delivered consistent good quality; UF: Koch
UF-SWRO pilot DMF-pilot Gibraltar, 4 months [119]	both: chlorine, 3 mg/l SBS	UF: 1 mg/l $FeCl_3$ DMF: 'coagulant'	both: 3 mg/l antiscalant	UF: 5 mg/l Cl_2 every 30 min.	SDI: feed: 13-25, UF filtrate: <0.8, DMF filtrate: 2.7-3.4; coagulation improved UF performance; UF: Aquasource
UF-SWRO pilot Zhejiang, China 60 days [120]	(1) no chlorine (2) 6 mg/l NaOCl	(1) 2.2 mg/l PFS (as Fe) (2) 2.2 mg/l PFS (as Fe)	not specified	CEB A / CEB B: 200 mg/l / 10 mg/l NaOCl 600 mg/l / 60 mg/l HCl	flux A: 69-95 $l/m^2/h$, B: 86-130 $l/m^2/h$; both showed stable permeate quality; NaOCl had little influence on SDI; both UF: inside-out

continued on next page

Pilot characteristics	Chlorine & SBS	Coagulant	Antiscalant & acid	CEB/CIP	Remarks
Parallel tests of submerged/outside-in and pressurized/inside-out membranes					
UF-SWRO (sub/Memcor) UF (press/Norit) one year, Moss Landing, California [121]	shock chlorination	$FeCl_3$ Run 1: 5 mg/l Run 2: without	not specified	Run 1: no CEB, CIP 1x/week Run 2 Memcor: CEB 500 mg/l NaOCl 1x/day CIP less than 1x/week Run 2 Norit: CEB 200 mg/l NaOCl + acid 2x/day, CIP 1-2x/week	Memcor: 12 month test, Norit: 6 months, feed: power plant cooling water (seawater), fluctuating in quality
UF (sub/PVDF) UF (press/PES) and GMF-SWRO pilot Hong Kong, China [122]	all: shock chlorination, dechlorination	sub. UF: 1.7-2.8 mg/l as Fe press. UF: 0.5-2.8 mg/l as Fe GMF: 1.4-8 mg/l as Fe	not specified	sub. UF: air scour, CIP every 1-16 weeks press. UF: CEB every 6-8 h altern. with acid, alkali, Cl_2 CIP every 2-8 weeks GMF: air scour, regular soaking with SBS	all systems met the min. SDI and turbidity requirements during most of the study but exhibited operational concerns that might limit their long-term performance (both UF systems experienced irreversible fouling); adjustment in chemical dosing and CIP could warrant long-term implementation of UF
Other					
UF-SWRO pilot CP-SWRO Med. and Red Sea 2 years [123]	UF: not specified CP Med: 1.2 mg/l Cl_2 CP Red: no chlorine	UF: 0.3 mg/l Fe CP: 0.3-0.7 mg/l Fe	CP Med: 2-stage media filter CP Red: 1-stage sand filter	UF: 20 mg/l Cl_2 every 15-30 min., acid (pH 2) and caustic 2x/day	UF produced lower SDI and turbidity than CP, especially during stormy weather, when a shut-down of the CP-SWRO was necessary; UF material: PES
UF-SWRO pilot Qingdao, China [124]	no chemicals were injected into the UF-SWRO system, neither through the feed water nor the backwash water				sand filter prior to UF; UF filtrate of very good quality (NTU <0.01, 95% $SDI_{15} < 3$); high recovery (80%), low flux (60 $l/m^2/h$) was best mode of operation; UF material: PAN
MF with flocculation MF with sand filtration MF with DMF [125]	not specified	1 mg/l $FeCl_3$ (all pilots)	not specified	not specified	DMF with in-line coagulation of 1 mg/l gave the lowest MFI and SDI_{15}, MF: Whatman GM (CA)
membrane pretreatment (literature review) [126]	not specified	0-2 mg/l $FeCl_3$	1.05-3 mg/l antiscalant	10-200 mg/l NaOCl	these are min./max. values based on an extensive literature review

Abbreviations: HF: hollow fiber, CP: conventional pretreatment, DMF: dual media filtration; GMF: granular media filtration, DE: diatomaceous earth, DAF: dissolved air flotation, AEB: air enhanced backwash, CEB: chemical enhanced backwash, CIP: cleaning in place, SBS: sodium bisulfite; $FeCl_3$/$FeSO_4$: ferric chloride, ferrous sulphate, PFS: polyferric sulphate, NaOCl: sodium hypochlorite, Cl_2: chlorine, ClO_2: chlorine dioxide, H_2SO_4: sulfuric acid, oxalic acid (HOOC-COOH): strong dicarboxylic acid, TOC: total organic carbon, DOC: dissolved organic carbon, NTU: nephelometric turbidity unit, SDI_{15}: 15-min. silt density index, UV: ultraviolet light, CA: cellulose acetate, PES: polyethersulfone, PS: polysulfone, PVDF: polyvinylidene fluoride, PAN: polyacrylonitrile

2.3.2 Integrated membrane systems (IMS)

An IMS uses ultra- or microfiltration (UF/MF) membranes for prefiltration of the feedwater in SWRO systems [22]. As the prefiltration membranes provide a fixed barrier to particles, their removal rating depends upon the pore size of the active layer of the membrane [114]. MF membranes are capable of filtering suspended particles such as sand, silt, plankton and bacteria from the feedwater, while UF membranes also remove some viruses and high molecular weight dissolved organics [5]. Although UF/MF systems are nearly 100% efficient in terms of particles and microorganisms removal, they are much less efficient in removing dissolved organic matter [51].

While most SWRO membranes have a spiral wound configuration, UF/MF membranes are typically hollow fibers or small capillaries. Depending on the system, the flow takes place either from the inside to the outside or from the outside to the inside of the membrane (Table 6). Inside-out systems are contained in a module, are pressurized, and most consist of polyethersulfone (PES). Outside-in systems can either be housed in modules and are pressure-driven, or the membranes are submerged in a tank and the water is drawn by a vacuum of 0.7-0.8 bar from the outside to the inside of the membrane. A main advantage of pressurized modules is that chemical cleaning volumes are reduced by a factor of 3 compared to submerged systems. Outside-in configurations can furthermore tolerate higher feed solids loadings, and the use of polyvinylidenfluoride (PVDF) for outside-in membranes allows the use of air scour due to the good mechanical strength and flexibility of the material. However, inside feed fibres tend to have higher permeability, due to the selection of PES rather than PVDF [127].

UF/MF pretreatment was originally developed for highly polluted waste waters and has also become widely accepted for brackish waters today [50]. As a pretreatment to SWRO, it has long been considered as an expensive option that was only used as a last resort to deal with difficult source waters [54]. Approximately 16 SWRO plants have been implemented with UF pretreatment as of late (Table 6). This is still a fairly low number taking the many pilot studies into account that have demonstrated the good performance of UF, and to a lesser extent of MF-SWRO systems, over the last ten years (Table 7). Plant operators tend to continue using conventional pretreatment [39, see also Table 5], and the still limited performance data on full-scale IMS can be seen as the true bottleneck for the breakthrough of UF/MF technology in SWRO [86].

UF/MF will possibly gain more popularity in the future with the advent of high-flux SWRO membranes, as the propensity for fouling at higher water fluxes requires a more effective pretreatment [5]. Moreover, SWRO projects have increased in plant size over the past years. New projects are therefore often located in industrial or shipping areas, which provide difficult feed water conditions, and use open intakes to ensure a sufficient feedwater flow. For these reasons it seems likely that more SWRO projects will require a sophisticated two or three stage conventional pretreatment in the future [51]. As the economic case can be a close decision when UF/MF is compared to an advanced conventional pretreatment, the operational benefits of membrane prefiltration under difficult feed water conditions may turn the balance in favor of UF/MF [113, 114][a]. According to Busch et al. [52], a diversification and extension of the driving forces can be observed.

[a] For example, UF/MF pretreatment was found to be more expensive than conventional pretreatment in a case study for a SWRO plant in the Eastern Mediterranean in 2002, leading to an increase in RO permeate cost of 1-2 US¢/m^3 [123]. A more recent case study, also for an Eastern Mediterranean location, showed that the additional cost of UF/MF is amortized over the life-time of the project due to savings on chemicals and consumables and may even result in a cost reduction by 0.7 US¢/m^3 compared to conventional pretreatment depending on the frequency of SWRO cleaning [113, 114].

The authors expect that increasing emphasis will be placed on the lower environmental impact through lower chemical use and sludge production of UF/MF, which is expected to become the main driving force behind UF/MF pretreatment in some Australian and Californian projects in the future.

The reputed benefits of membrane pretreatment prior to SWRO can be summarized as follows: UF/MF pretreatment produces a constant and high quality feedwater regardless of source water fluctuations. This results in a reduced fouling potential of the SWRO membranes, a lower cleaning frequency and hence a lower chemical use and labor intensity. It also results in a longer membrane life, i.e., a lower membrane replacement rate, and therefore savings in material and energy use in the manufacturing process of the SWRO membranes. Moreover, a better quality of the feedwater offers the potential to operate the SWRO membranes at a higher flux rate. In SWRO applications, the flux can range from 12 $l/m^2/h$ for surface seawater with high fouling potential and conventional pretreatment up to 17 $l/m^2/h$ for good feedwater quality, e.g., as from a clean beach well. At a higher SWRO flux a lower membrane surface area is needed to produce the same permeate flow, resulting in a more economical use of materials and space. The pretreatment system would be smaller as well because less feed water needs to be treated. Membrane pretreatment can thus save about one third in plant area size [22] and about one third in the membrane replacement rate [114].

Chemical use in UF/MF

UF/MF pretreatment reduces the fouling potential of the SWRO membranes by transferring the risk to the UF/MF membranes. Although the UF/MF configuration can tolerate a higher feed *solids* loading than spiral wound SWRO membranes [22], the build-up of material may still cause fouling on the UF/MF membranes, which may lead to an increase in energy demand and periodic cleaning [128]. Furthermore, UF/MF is not very efficient in removing *dissolved* organic matter of low molecular weight from the feed water [51], which can cause severe fouling of the SWRO membranes. Additional pretreatment prior to UF/MF may therefore be needed to counter the accumulation of material and to maintain a high flux through the UF/MF membranes, and to increase the removal capability of dissolved organics in order to prevent fouling of the SWRO membranes.

MF and UF were originally introduced as a “chemical-free” alternative to conventional pretreatment [129] that may eliminate the need for coagulant dosing with good feed water quality [113] and the need for a continuous presence of free chlorine [54, 57]. However, the use of a disinfectant, either continuously or intermittently, and the use of inline coagulation, which is the dosing of a coagulant without a sedimentation or granular media filtration step, seem to be common practice in many UF systems in order to improve the performance and filtrate quality. Moreover, UF/MF membranes usually require intermittent chemically enhanced backwashing (CEB) and cleaning in place (CIP).

Performance data on more than 40 UF-SWRO systems could be obtained from the recent literature (years 2000 to 2009). Of these, only 16 provided information on full-scale *operational* plants (Table 6), the remaining were *pilot systems* (Table 7). Examining the available literature led to the following results and conclusions concerning chemical use in integrated membrane systems.

Coagulant use

Of the 16 operational plants (Table 6), seven were operated with a coagulant and seven without or with optional use of a coagulant. Doses, which were reported in three cases, ranged between 0.3 and 10 mg/l as Fe [52, 130, 131]. Of the papers which reported *no* coagulant use, three mentioned additional pretreatment prior to the UF, either DAF [94], sedimentation [95], or two stage media filtration [92, 93]. One plant was operated at only 50% of the UF design flux [86]. In another system, it can be assumed that coagulants are not used because of a subsurface intake preceding the UF [96]. Inasmuch as these cases allow for a conclusion, UF pretreatment often requires some kind of other pretreatment if it is to be operated without coagulants, either in the form of natural or engineered filtration systems, sedimentation, or flotation. However, UF operation on open intakes without additional pretreatment or coagulation has been reported in three cases [52, 87].

Of the 26 UF/MF pilot studies, about three quarters either reported coagulant use or investigated the need for coagulant dosing (Table 7). Doses ranged between 0.2-2.8 mg/l as Fe (ten studies) and 0.1-10 mg/l as $FeCl_3$ (ten studies). Many studies found that pre-coagulation improved the UF/MF performance in terms of UF/MF fouling or filtrate quality [e.g., 99, 101, 112, 119]. Although UF generally performed well in terms of particles and microorganisms removal, a few studies found UF/MF to be less efficient in terms of TOC removal than conventional pretreatment [105, 107], or observed irreversible fouling [122]. The poor performance in TOC removal had been anticipated where no coagulant was used [106, 107], and improved considerably or even outperformed conventional pretreatment where a coagulant was added [105][b]. In the case where irreversible fouling occurred, it was concluded that an adjustment in the chemical dosing combined with proper CIP is necessary to ensure long-term performance of the IMS [122].

These findings underline that in-line coagulation prior to UF/MF seems to be the rule rather than the exception, especially where water is received from an open intake without any additional pretreatment or where TOC removal is important, and that a poor performance is often attributed to an inadequate or no chemical pretreatment. Only two pilot tests reported a good performance without coagulant use, with similar reasons as for the full-scale plants that operated without coagulants. In one case, no chemicals were injected into the UF-SWRO system, neither through the feed water nor the backwash water, which may be attributed to a sand filter preceding the UF [124]. In the other case, the UF-SWRO received its feedwater from a beachwell [118].

Chlorine use

Of the 16 operational UF-SWRO systems (Table 6), six reported the addition of a disinfectant to the feedwater (either Cl_2, NaOCl, CaOCl or ClO_2). Two apparently chlorinate the feedwater *continuously* to achieve residual chlorine levels of 0.3-1.0 mg/l [86, 95], which is similar to conventional pretreatment where chlorine is added continuously. Three reported *intermittent* disinfection of the pretreatment line, either with a *low chlorine dose* of 0.5-1 mg/l once a week [96] or *shock chlorination* [92] up to 1.5-25 mg/l for 15-30 minutes once a day at the seawater intake followed by dechlorination after the UF [89]. Three reported the use of chlorine in CEB. The doses were 10 mg/l twice a day, 20 mg/l twice a day, and 15 mg/l once every hour, i.e., with every backwash [86, 92, 94]. For comparison, a dose of 15 mg/l to every backwash corresponds to a continuous dose of 0.15 mg/l to the feedwater if one assumes a 10% water loss for backwashing. Only two plants reported that chlorine is not used in the UF feed, but did not specify if chlorine is used in CEB and CIP [52].

Of the 26 UF/MF pilot studies (Table 7), ten reported chlorine use in CEB only [98, 99, 102, 104, 107–110, 112, 123], and seven reported chlorine addition to the UF feed *and* CEB [101, 117–122]. The given chlorine doses to the feed water were 1 mg/l in

[b] MF/UF with in-line coagulation at a high dose of 10 mg/l reduced TOC levels by 50% compared to 16% removal by UF/MF without coagulant and 35% achieved by conventional pretreatment.

winter and 2 mg/l in summer [117], which is comparable to the levels in the operational plants [e.g., a dosage of 1 mg/l results in a chlorine residual of 0.5 mg/l, 86]. Two other studies gave a rather high chlorine dosage of 6 mg/l to the feedwater [101, 120].

Chlorine levels in CEB varied between 5-20 mg/l every 15-40 minutes [i.e., every backwash, 98, 99, 101, 119, 123], to 20-50 mg/l every 2-6 hours [112, 117], up to 100-500 mg/l once or twice per day [98, 100, 102, 107, 110, 120, 121]. These values show a higher upper range than the results of a review carried out by Fritzmann et al. [126], who reported chlorine levels of 10-200 mg/l in CEB of UF/MF membranes.

For Pall membranes, chlorine was applied in doses of 10 mg/l every 20-40 minutes, 500 mg/l once per day, or a combination of both [98–100]. In Dow plants, treatment is generally similar. Doses are slightly higher with 15-20 mg/l but CEB is slightly less frequent with once every 30-60 minutes. A dose of 500 mg/l once or twice per day has also been reported [86, 101, 102]. One full-scale Dow plant operated without any CEB [87]. For Norit membranes, doses of 200 mg/l were employed once or twice per day [110, 121]. Both Pall and Dow membranes have an outside-in configuration, whereas Norit has an inside-out configuration. Other membrane suppliers which produce inside-out UF membranes are Hydranautics, Inge and Aquasource. Their chlorine dose levels range from 5 mg/l every 30 minutes [119, for Aquasource] to 20-50 mg/l every 2-6 hours [92, 93, 112, 115–117, for Hydranautics, Inge] in pilot plants. Doses in full-scale plants are lower and less frequent with 20 mg/l every 8-24 hours [92–94].

Two papers noted that heavy fouling of the UF was observed without chlorination [104, 117]. Only one plant explicitly reported a good UF performance without chlorine addition or any other chemicals to the feed water nor the backwash water [124].

CEB and CIP

As mentioned in the previous sections, UF/MF membranes usually require three intermittent cleaning processes (Tables 6 and 7) to maintain permeability [22, 50, 132]:

- a frequent membrane backwash every 20-60 minutes for 1-2 minutes with filtered water to counteract the accumulation of suspended solids and bacteria, often enhanced with an air scour (AEB) to accelerate the removal of particles and foulants (short-term permeability maintenance),
- a chemically enhanced backwash (CEB) on a weekly or daily basis to restore the membrane's performance, using chlorine, hydrogen peroxide or chlorine dioxide, with acid and base conditioning (medium-term maintenance),
- cleaning in place (CIP) on a monthly basis or if severe fouling occurs using similar chemicals as for CEB at potentially higher concentrations (long-term maintenance).

The membrane backwash, containing the natural solids from the sea and typically coagulants, can either be discharged into the sea along with the concentrate, or dewatered and transported to a landfill (cf. section 3.3.3, page 71). The CEB and CIP cleaning solutions can either be conveyed to a tank for initial treatment and then disposed of into a sewer, or discharged into the sea along with the concentrate (cf. section 3.3.6, page 77). The practice depends on the existing discharge regulations, if any. Most UF/MF plants seem to discharge their backwash and cleaning wastes without any treatment.

CIP is usually a two step process, involving the recirculation of a caustic solution containing chlorine to remove organic foulants followed by a citric acid solution to remove inorganic foulants, and each followed by a rinse. The waste can be combined and neutralized, adding base or acid as necessary [98]. The range of chemicals used in CEB and CIP extends from generic chemicals like sodium hypochlorite, sodium hydroxide or hydrochloric acid to commercially available cleaning ‘cocktails’ [108].

Both pressurized and submerged configurations use frequent CEB and CIP, although the procedure and handling for CIP is more complicated in the submerged system. For inside-out designs, the membrane backwash is carried out by reversing the flow direction and by increasing the flux in order to expel particles from the membrane. Outside-in configurations often also utilize air during backwashing, which either passes through the fiber (air enhanced backwash) or along the outside of the fiber [air scour, 22]. Air scouring, with a minimum increase in energy demand, was found to improve the backwash efficiency and to reduce the filtrate water consumption [104].

Conclusions

The majority of UF/MF systems showed a reliable and often superior performance compared to conventional pretreatment in terms of particles and microorganisms removal, but some also performed less well in terms of dissolved organics removal. MF/UF pretreatment may therefore not always be an adequate solution for the prevention of organic fouling [133], unless coagulation prior to UF/MF is used to improve the adsorption and removal of organics as fine particulates. Coagulation may not be needed in every case. The need for coagulants and the optimum dosage should be established by pilot testing [114], as overdosing may cause operational problems of the prefiltration membranes [91] or the SWRO membranes [123], and would represent an unnecessary cost factor and environmental burden. The majority of UF/MF systems also used chlorine, either continuously in the UF feedwater or intermittently in the UF feed or backwash water.

It can be concluded that most of the chemicals, which are used in conventional pretreatment, are also commonly used in UF/MF systems. However, total chemical use in UF/MF is claimed to be significantly lower or ‘minimal’ compared to conventional levels [50, 86, 113]. For example, Pearce [114] assumes that UF/MF systems can be operated with 43% of the coagulant dose of conventional pretreatment (i.e., 0.3 mg/l instead of 0.7 mg/l as Fe) and that chlorine use can be reduced to intermittent chlorine cleaning.

To summarize, the reported coagulant levels in this review ranged between 0.2 and 2.8 mg/l as Fe in UF/MF pilot plants and between 0.3 and 10 mg/l as Fe in UF/MF full scale plants, with values of 0.2-2 mg/l assumed to be the rule and 10 mg/l the exception (Tables 6, 7). This compares to ranges of 0.2 to 20 mg/l as $FeCl_3$/$FeSO_4$ (0.1-8 mg/l as Fe assuming 40% active ingredient) in conventional pretreatment, where ranges of 1 to 6 mg/l (0.4-2.4 as Fe) are assumed to be the rule and 20 mg/l the exception (Table 5).

While coagulant use seems to be similar in UF/MF and conventional pretreatment, chlorine use appears to be higher in some UF/MF-SWRO plants due to a combination of continuous chlorination and intermittent shock chlorination during CEB and CIP.

Chlorine was reported to be used continuously in several UF/MF systems in concentrations between 0.3-2 mg/l, which is similar to conventional pretreatment where low-level chlorination of 1-2 mg/l is applied. However, many conventional plants use intermittent chlorination. For example, an intermittent dose of 10 mg/l for one hour per day equals a continuous dosage of ‘only’ 0.4 mg/l. As intervals in conventional pretreatment are often even longer, e.g., once per week, chlorine use may be even lower.

Many UF/MF systems additionally employ CEBs with considerable chlorine doses, which may compare to an additional chlorine level of 0.5-2 mg/l if the chemicals were used continuously. For instance, a UF-SWRO plant in Asia reported feedwater chlorination in a dosage of 1 mg/l and chlorination of every backwash once every hour in a dosage of 15 mg/l [86]. Assuming a 10% water loss for backwashing, an intermittent chlorine dosage of 15 mg/l corresponds to a continuous chlorine dosage of 0.15 mg/l to the feedwater, or a total chlorine dosage of 1.15 mg/l.

In conclusion, the reported chemical use in UF/MF pretreatment does not seem to live up to the expectations that were initially imposed on the technology of being a ‘chemical-free’ or ‘low-chemical’ process, although low-chemical approaches seem to be feasible.

A comparative life cycle analysis (LCA) of conventional and membrane pretreatment systems found that the overall chemical use of UF/MF systems is lower than of conventional pretreatment systems. This, however, has only a minor effect on the overall ‘environmental burden’ of the system, as most of the reduction in the overall environmental burden of an IMS stems from a reduction in the overall energy demand, and not from the lower chemical use [126]. The lower chemical use of the UF/MF system may be attributed to the savings in SWRO cleaning rather than savings in pretreatment chemicals. An LCA and aspects of operability led to the selection of a UF/MF system over a classic two stage GMF system for a desalination project south of Perth with a capacity of 140,000 m^3/d, which is expected to operate without any coagulants [134].

The chemical use of an IMS depends on the process design and the feedwater quality. In order to implement a membrane filtration successfully, the filtration time, the backwashing and CEB intervals need to be optimized. One option to postpone backwashing and CEB is by having additional pretreatment prior to membrane filtration. This may include natural systems such as beachwells or engineered systems such as media filters. Another option is by lowering the flux, which will increase the total membrane area to be installed and thus the capital investment [108]. It would also be possible to operate without coagulant pretreatment, but with more frequent cleaning [114].

Saving two RO cleans per year can reduce the total water cost of an UF/MF pretreated RO system below that of a conventional system, purely based on savings in RO replacement costs, chemicals, and cleaning downtime. A well-designed IMS operated on good feed water quality may thus considerably reduce chemical use. The “occasional use of commodity chemicals is all that is required” in that case, with “much lower costs than the proprietary chemical cleaning regimes required for RO” [114]. However, an IMS does not seem to be superior to conventional pretreatment in terms of chemical use when in-line coagulation, chlorination and frequent CEB is employed. With such a system, the only remaining benefit may stem from the reduced cleaning frequency of the SWRO membranes, which may cause a reduction in chemical use of the overall system.

2.3.3 Distillation

Most MSF plants today are of the brine circulation design, which reduces the feedwater and therefore the pumping and chemical requirements compared to the once through design. Plants with a brine circulation design are subdivided into three sections, i.e., the heat input, the heat recovery and the heat rejection section (Figure 9). Most of the seawater that passes through the heat rejection section, which acts as a heat sink, is discharged as cooling water into the sea. It is typically mixed with the concentrate from the last distillation stage. Only a small portion of the intake water is treated (the make-up water), mixed with recycled brine from the last stage of the heat rejection section, and used as feedwater to the heat recovery section. The flow in this section resembles a turbulent, rapidly flowing river that may be up to 20 m wide and 100 m long in the largest MSF systems, with a maximum temperature of 90-120°C reached in the heating section.

MED plants can have many possible configurations, mainly distinguished by the arrangement of heat exchanger tubes. In the common horizontal tube arrangement, the feedwater enters all effects in parallel (Figure 9) and is sprayed onto the outside surfaces

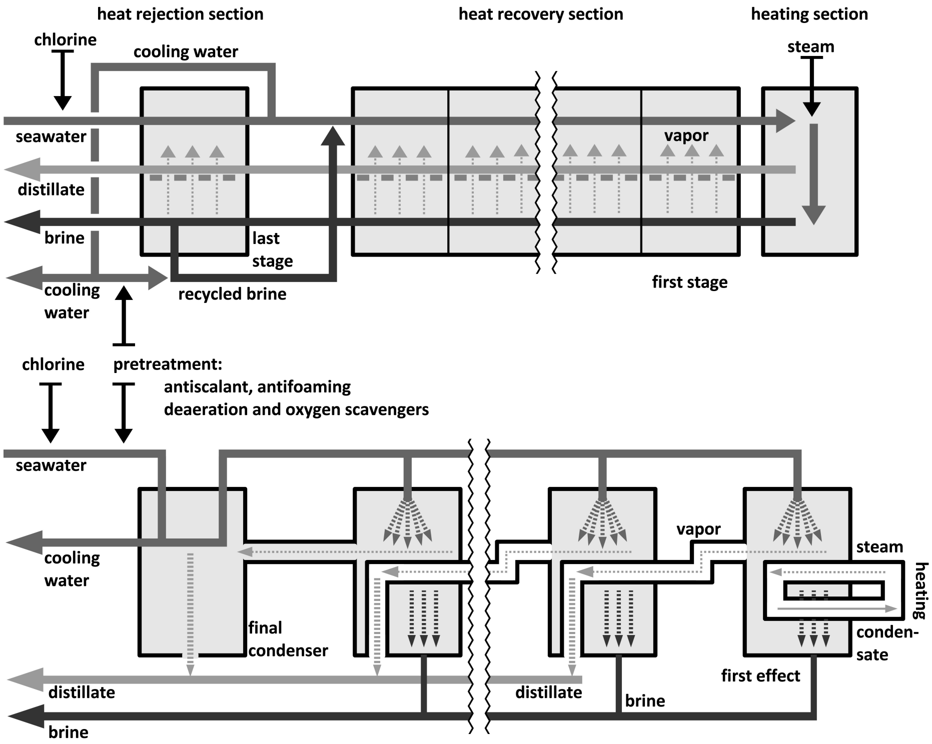

Figure 9: Simplified flow-scheme of a MSF distillation plant (top) with brine recycle configuration and an MED distillation plant (bottom) with horizontal tube arrangement including conventional pretreatment steps and different waste streams [adapted from 15, 21].

of evaporator tubes to produce a thin film, which rapidly raises the water temperature to the boiling point. High temperature MED plants operate at 110°C and low temperature MED plants at 70°C [135], with the latter being more common. As the MED process results in a very low temperature drop of 1.5-2.5°C per effect, a sufficient number of effects can be incorporated even at low temperatures so that comparatively high performance ratios are achieved [136]. MED plants can be configured to function with less cooling water, resulting in a higher temperature rise (of over 20°C) of the reject than with MSF.

The MSF process is the most commonly employed distillation technology due to its robustness and capability of large production capacities per unit. However, MED technology is increasingly being used due to its reduced pumping requirements and thus its lower power use compared to MSF. Large MED plants often incorporate thermal vapor compression, where the pressure of the motive steam is used in addition to the heat of the steam in order to increase the efficiency of the process. Moreover, MED plants have a lower potential for scaling due to lower operating temperatures and a lower potential for corrosion due to the use of other construction materials, such as corrosion-proof plastic materials and coatings, aluminum or titanium [136, 137]. A disadvantage of MED is that scales form on the outside surfaces of the tubes and therefore cannot be mechanically removed by circulating sponge balls through the tube system, as in MSF plants.

Table 8: Energy consumption of desalination and conventional water supply options.

	Electrical energy [kWh/m^3]	Main energy form	Thermal energy [MJ/m^3]	Performance ratio [kg/2326 kJ]
BWRO [5]	0.5−3.0	electrical		
SWRO [5]	2.5−7.0	electrical		
MSF [5]	3.0−5.0	steam/thermal	250−330	7.0−9.0
MED thermal vapor compression [5]	1.5−2.5	steam/thermal	145−390	8.0−14
Surface water treatment [34, 113]	0.2−0.4	electrical		
Waste water reclamation [34]	0.5−1.0	electrical		
Long distance transport [a]	1.6−2.8 [b] 12.0 [c]	electrical		

[a] depends on the transport distance and the elevation gap between source and destination, e.g., normal distribution costs are around 0.6 kWh/m^3 [113, based on UK experience]
[b] power required to convey surface water to San Diego, Los Angeles and Orange County [34]
[c] power required if water was conveyed to Perth via the Kimberley pipeline [70]

Conventional steps in most distillation plants operating on surface seawater include control of marine growth, usually by chlorination (see also page 64), control of scaling by dosing of an antiscalant agent (see also page 75), reduction of foaming by dosing of an antifoam agent, and deaeration or use of oxygen scavengers to inhibit corrosion. The cooling water of MSF/MED plants is chlorinated but not de-aerated, i.e., it may contain disinfection by-products and corrosion products. The cooling water is commonly blended with the brine, which contains antiscalant and antifoam chemicals and is generally de-aerated prior to blending with the cooling water (Tables 13-15, page 79ff.).

2.3.4 Energy use

The energy demand of desalination depends on a range of factors including recovery, pretreatment design (e.g., conventional vs. membrane filtration), the type of distillation process (e.g., MSF vs. MED) or SWRO membranes used (e.g., low energy membranes), the efficiency of pumps and motors, the type and efficiency of the energy recovery system installed (if any), and environmental conditions (e.g., feed water temperature). Energy demand also depends on the product water specifications. For example, employing a second SWRO pass for boron removal will increase the energy demand of the process. Table 8 summarizes the typical energy requirements of the main desalination processes and compares them to other water supply options.

Reverse Osmosis

Modern SWRO plants can achieve a *specific* energy demand of <2.5 kWh/m^3 and a total energy demand <3.5 kWh/m^3 by using state of the art equipment (such as pressure exchangers, variable frequency pumps and low-pressure membranes) and under favorable conditions (i.e., a low fouling potential, a temperature >15°C, a salinity[c] <35).

[c] The UNESCO definition of Practical Salinity Units (psu) is used, which is the conductivity ratio of a seawater sample to a standard KCl solution and hence a *dimensionless* value. As salinity reflects the amount of total dissolved solids (TDS) in ocean water, it was traditionally expressed as parts per thousand (ppt). A salinity of 35 ppt equals 35 g of salt per 1,000 g of seawater, or 35,000 ppm (mg/l), or in approximation 35 (psu).

Table 9: Energy consumption of SWRO with different pretreatments per cubic meter of *product* water at 20°C, a feed salinity of 40, a total recovery of 41%, and using work exchangers [53].

	Total energy demand [kWh/m^3]	Specific demand[a] [kWh/m^3]	Pre-treatment[b] [kWh/m^3]	Waste water and sludge treatment [kWh/m^3]
FL, SM, 1 GF	4.01	3.79	0.035	0.019
MF/UF	4.24	3.81	0.215	0.042
DAF + 2 filters	4.37	3.78	0.395	0.024
DAF + MF/UF	4.64	3.83	0.580	0.052

[a] 1st and 2nd RO pass, including cleaning operations

[b] without seawater extraction, screening and pumping

Abbreviations: FL: flocculation, SM: static mixer, GF: gravity filter, DAF: dissolved air flotation, UF: ultrafiltration, MF: microfiltration

Other energy consumptions (internal pumping, auxiliaries, administration buildings, laboratories, post-treatment, water transfer to supply network) account for the difference of 0.1-0.2 kWh/m^3 between the total energy demand and the sum of the given specific and single demands.

The real energy demand may be higher under less favorable conditions. For example, the calculated *specific* energy demand of a state of the art facility with a feed salinity of 40 and a temperature of 20°C (typical for Eastern Mediterranean seawater), a total recovery of 41%, and equipped with the most efficient energy recovery system, is approximately 3.8 kWh/m^3 (Table 9). An additional 0.2-0.8 kWh/m^3 is required for pretreatment, waste water and sludge treatment (depending on the feedwater quality), administration buildings and laboratories, post-treatment and drinking water pumping to supply network, which leads to a *total* energy consumption of about 4-4.6 kWh/m^3 [53, 138].

For instance, the Spanish National Hydrological Plan assumes a *total* energy value of 4 kWh/m^3 under the assumption that plants are equipped with state of the art technologies [139], which is similar to the energy demands reported for other large SWRO projects in Israel [3.9 kWh/m^3, 82] and California [4.5 kWh/m^3, 132]. Older or smaller SWRO plants without energy recovery may use up to 5 kWh/m^3 at 50% recovery. Values given for recent large SWRO projects that also include the *transfer* of water to the supply grid ranged between 4.2-5.3 kWh/m^3 [71, 85, 140, for further examples see Table 5, page 30].

The Affordable Desalination Collaboration operated a demonstration plant in California over two years using state of the art, off-the-shelf technology and set a world record in *specific* energy consumption of 1.58 kWh/m^3 with a low-energy membrane operated at 42% recovery. However, this result is currently not realistic in full-scale applications as it was achieved at the expense of permeate water quality and process recovery. The *specific* energy demand of SWRO plants usually increases with recovery, but the *total* energy demand decreases with the recovery rate as less feedwater must be pumped and treated to obtain the same volume of permeate at a higher recovery. Optimizing the energy demand of the whole process is a complex undertaking, as the single sub-systems of a SWRO plant, particularly its pretreatment, first and second passes, are closely interrelated [53]. At the most affordable point for a single stage 190,000 m^3/d plant, a total treatment energy in the range of 2.75-2.98 kWh/m^3 was demonstrated [141].

A report published by the U.S. National Research Council estimates that the practical upper limit of energy savings in RO may be about 15% from current levels, assuming a system operating at 40% recovery, using a 95% energy recovery device and a seawater RO membrane with twice the permeability of today's best membranes. Improvements in module design appear to have the greatest potential for reducing the overall energy costs, unless a breakthrough in an alternate technology to RO is achieved [5].

Energy recovery devices for SWRO fall into two categories. Pressure and work exchangers transfer the concentrate pressure directly to the feed stream, which allows for a very efficient energy transfer, whereas all other energy recovery devices first transfer the concentrate pressure to mechanical power and then convert the mechanical power back to feed pressure. The efficiency of devices ranges from 70-85% for turbochargers, 80-88.5% for Pelton turbines, 90-95% for work exchangers to 95% for pressure exchangers [5, 39, 53]. A booster pump typically compensates for the remaining pressure loss [142]. A two pass SWRO plant with a work exchanger may require about 3.6-4.0 kWh/m^3 of energy, which is about 0.7-0.8 kWh/m^3 higher with a Pelton turbine [53].

The energy demand of the pretreatment (Table 10) is lowest for flocculation with a static mixer and one or two stage gravity filtration (0.015-0.02 kWh per m^3 *filtrate* water). It increases if the static mixer is replaced by a flocculation basin (0.10-0.12 kWh/m^3), and if additional flotation or sedimentation steps are added (0.14-0.16 kWh/m^3) [53]. The energy demand of a more extensive conventional pretreatment is comparable to the energy demand that is generally given for UF/MF pretreatment in the literature, which is 0.10-0.20 kWh/m^3 [53, 54, 108, 143]. However, plant operators give lower energy demands of only 0.03-0.09 kWh per m^3 filtrate water. The most energy intensive option would be UF/MF pretreatment with additional pretreatment such as flotation, with an estimated energy demand of 0.25 kWh/m^3 [53]. Table 9 shows the pretreatment energy demand of a plant per m^3 *product* water operated at 41% total recovery. Depending on the feed water quality, to which the pretreatment is customized to, the energy demand may account for more than 10% of the overall energy demand of the plant [53].

When comparing energy requirements, the whole process should be taken into account. For instance, the overall energy costs in a SWRO system with UF/MF pretreatment may be lower than for conventional pretreatment due to a lower energy consumption in the SWRO stage. A better feedwater quality results in lower SWRO fouling and a reduction in RO pressure drop caused by fouling [126].

Distillation

MSF distillation plants, which have an operating temperature up to 120°C, require about 250-330 MJ of thermal energy and 3-5 kWh of electrical energy for the production of one cubic meter of water (Table 8) . MED plants, which operate at temperatures below 70°C, require 145-390 MJ of thermal and 1.5-2.5 kWh of electrical energy per m^3 of water.

Although distillation processes require more energy than SWRO, they are still the first choice in countries of the Middle East (cf. section 1.4, page 10). This has several technical and economical reasons, including difficult feed water conditions for SWRO plants and the availability of low cost energy. Dual-purpose co-generation facilities predominate in the region, which integrate MSF or MED distillation with power generation.

Because MSF and MED are capable of using 'low value' and 'waste' heat, it is not straightforward to compare the total energy use of distillation with reverse osmosis. Waste heat is heat that is released to the environment, such as steam leaving a back-pressure turbine that can no longer be used to produce electricity. Low value heat is heat of low temperature with little value for industrial processes, such as steam extracted from

Table 10: Energy consumption of different SWRO pretreatments. The values refer to the specific demand of the pretreatment only without intake and initial screening.

	[kWh/m^3 filtrate]	[kWh/m^3 product][e] (for 40% recovery)	[kWh/m^3 product][e] (for 50% recovery)
FL, SM, 1 filter [53][a]	0.015	0.037	0.030
FL, SM, 2 filter [53][a]	0.020	0.049	0.040
FL, FB, 1 filter [53][a]	0.100	0.244	0.200
FL, FB, 2 filter [53][a]	0.120	0.293	0.240
SED, 1 filter [53][a]	0.140	0.341	0.280
SED, 2 filter [53][a]	0.150	0.366	0.300
DAF, 1 filter [53][a]	0.150	0.366	0.300
DAF, 2 filter [53][a]	0.160	0.390	0.320
MF/UF low [53][a]	0.100	0.244	0.200
MF/UF high [53][a]	0.200	0.488	0.400
DAF + MF/UF [53][a]	0.250	0.610	0.500
Norit Xiga, Seaguard [108, 130][b] (inside-out, pressurized, PES/PVP, 0.2-0.4 bar TMP)	0.030	0.073	0.060
Inge Multibore [117][c] (inside-out, pressurized, PES, 0.25 bar TMP)	0.050	0.122	0.100
Dow SFP [87, 131][d] (outside-in, pressurized, PVDF, air scour, 0.5 bar TMP)	0.090	0.220	0.180
Zenon Zeeweed [50, 106, 144] (inside-out, submerged/vacuum, PVDF, air scour, 0.1-0.35 bar)	no information		

[a] without intake/screening, assuming a feed pressure of 1 bar for initial pretreatment
[b] value of 0.03 kWh/m^3 for the UF excluding the intake. The intake in a full-scale SWRO plant with this pretreatment accounts for a site-specific energy demand of 0.08 kWh/m^3 and screening for <0.01 kWh/m^3, amounting to a total energy demand of 0.12 kWh/m^3 in this case.
[c] value of 0.05 kWh/m^3 for a pilot plant and including ultrafiltration, backwash and CEB. The intake (1.2 km offshore) accounts for an additional 0.02 kWh/m^3 [145]. Pilot plants typically have a higher specific energy demand than full-scale plants, which can be assumed to have a specific energy demand of about 0.01 kWh/m^3 for the UF at 0.4 bar transmembrane pressure [146].
[d] UF energy demand in two operational plants including backwash and air scrub but without intake
[e] To obtain the energy demand normalized to one cubic meter of *RO permeate (product)* water, the values given for one cubic meter of *pretreated (filtrate)* water are divided by the recovery rate of the plant (i.e., 0.4 for 40% recovery).
Abbreviations: FL: flocculation, SM: static mixer, FB: flocculation basin, SED: sedimentation, DAF: dissolved air flotation, UF: ultrafiltration, MF: microfiltration, PES: polyethersulfone, PVP: polyvinyl-pyrollidone, PVDF: polyvinylidene fluoride, TMP: transmembrane pressure.

Table 11: Water consumption of different desalination processes depending on recovery rate.

	Product recovery rate [as % of feedwater]	Feedwater required for 1 m^3 of product	Concentrate resulting from 1 m^3 of product
BWRO	60–85	1.7–1.2 m^3	0.7–0.2 m^3
SWRO	35–60	2.9–1.7 m^3	1.9–0.7 m^3
MSF	10–20 (35–45)*	10.0–5.0 m^3	9.0–4.0 m^3
MED	20–35 (35–45)*	5.0–2.9 m^3	4.0–1.9 m^3

* brackets: without cooling water requirements

a condensing turbine, that could still be used to generate electricity, but that is sometimes wasted depending on practical circumstances such as electricity demand [5].

In a comparative life cycle assessment of different desalination processes it was concluded that the environmental 'load' of SWRO is one order of magnitude lower than the load of thermal processes if these are operated with a conventional boiler, but comparable if the thermal processes are entirely driven by waste heat. MED was found to be more efficient than MSF and was also more energy efficient than RO in one evaluation under the assumption that waste heat is used. This can also be seen if only the electrical energy demand is compared (Table 8). For instance, modern cruise ships often choose MED as it requires only 20-33% of the electrical energy of RO and as the heat energy can be obtained from the ship's engines [5]. The environmental load of distillation processes can be significantly reduced (by 75%) if integrated into other industrial processes [147–149].

The comparative life cycle assessment also found that the environmental load of material use and disposal (section 2.3.6) has little weight (10%) compared to plant operation (90%) due to the high energy demand of all desalination processes [147–149].

2.3.5 Water use

The consumption of water (as feedwater to the desalination plant) depends on the recovery rate of the process, and the water use for backwashing of filters. The concentrate from the RO process and the backwash from filters (with or without treatment) is discharged back into the sea. SWRO plants require between 2.9-1.7 m^3 of intake water and produce between 1.9-0.7 m^3 of concentrate at recoveries between 35-60% (Table 11).

Depending on the quality of the feed water, conventional pretreatment filters are backwashed in intervals ranging from once every 8 hours to once every few days [22], but typical is once per day [50]. Backwashing usually lasts for 8-20 minutes at flow rates of 35-55 $m^3/m^2/h$ (compared to filtration rates of 7.5-15 $m^3/m^2/h$). Some plants use concentrate for backwashing, which can also be accompanied by an scour at a rate of 55-90 m/h. The filtrate volume required for a backwash is typically 2-3% of the system filtrate capacity per filtration stage [22]. If one assumes a filtration rate of 10 $m^3/m^2/h$ and a backwash flow of 45 $m^3/m^2/h$ for 15 min. once a day, the water loss is 5%. For example, two conventional pretreatment systems in the Mediterranean and Red Sea were operated with water losses of 4% and 4.4% for backwashing, respectively [123]. The water losses in UF/MF pretreatment are generally higher than in conventional pretreatment due to a higher backwashing frequency of normally at least once per hour. The recovery of UF pretreatment systems prior to SWRO ranges between 90% and 95.5% [92–94, 98, 103, 110, 112, 118], corresponding to water losses between 10% and 4.5%.

Most thermal plants have considerably lower recovery rates than SWRO plants. As they use cooling water for temperature control, the seawater flow rate to thermal plants has to be 2 to 4 times higher than the feed to RO plants for the same amount of product water extraction. The cooling water is discharged along with the concentrate and mixing of both reject streams usually takes place before discharge into the sea.

2.3.6 Material use

The manufacturing process of equipment for construction and the replacement of parts consumes materials and energy, with secondary environmental impacts resulting from the production process and the extraction and transport of raw materials. All material and energy flows can be balanced in a life cycle analysis [147–149].

The operation of RO plants will result in worn-out membranes which are usually disposed in landfills. The standard life-time of SWRO membranes is 3 to 7 years, depending on the feedwater quality and the efficiency of the pretreatment. In some cases, 10 year old spiral wound membrane elements may still be operational. A few companies recover used membranes and clean them for further use in another application. However, the composite nature of modern membranes probably makes it difficult to separate the single materials for recycling at the end of their useful life [39]. Modern membranes typically consist of cellulose acetates, polyamides, polyetheramides and polyethersulfones. The most widely used material is a thin-film composite polymer which combines a thin polyamide layer with a microporous polysulfone support layer [5].

For illustration, a total of 15,904 polyamide thin-film composite membranes are being used in the 200,000 m^3/d SWRO plant in Valdelentisco, Spain [150]. Each membrane has an active surface area of 37 m^2, totalling 588,448 m^2 (59 ha) for the entire plant, which have to be replaced and disposed of every few years. The world's largest SWRO plant with a capacity of 330,000 m^3/d in Ashkelon, Israel, has 27,000 membrane elements with an active surface area of about 99 ha (or 200 football fields). Efforts to obtain more specific data on the energy and materials use in the manufacturing process of the membranes from literature and membrane suppliers were not successful.

UF/MF membranes used for prefiltration are usually replaced every 5 to 10 years [114]. The most widely used materials for UF/MF membranes are polyethersulfone (PES) for inside feed configurations (e.g., Norit, Inge, Hydranautics membranes) and polyvinylidene fluoride (PVDF) for outside feed configuration (e.g., Zenon, Pall, Dow, Memcor). Other materials include cellulose acetate (CA, e.g., Aquasource) and polyacrylonitrile (PAN). As UF/MF membranes are usually made from a single polymer [22], recycling of the material is theoretically possible, but is currently not practiced in most plants and by most membrane suppliers. Disposal options include landfilling or incineration. A benefit of UF/MF pretreatment is that the improved feed water quality may result in a lower SWRO membrane replacement rate, which can be reduced from 15-20% per year (membrane life-time of 5 to 7 years) with a conventional pretreatment to 10-13% per year (8 to 10 years) with UF/MF pretreatment, which is a 33% reduction [114].

For a full-scale two-pass SWRO plant in Jeddah with a combined UF and media filtration pretreatment, a UF replacement rate of 14.3% (7 years) was given. The replacement rate of the SWRO membranes was only 8% (13 years) as compared to 12% (8 years) for conventional pretreatment. The replacement rate of BWRO membranes in the second RO stage was also 8%, regardless of the pretreatment type [92]. For a full-scale UF-SWRO plant in Dubai, an annual SWRO replacement rate of 11% (9 years) is specified [89].

In conventional pretreatment, media filters have to be replaced about once every 10 years [92]. The filter beds consist of sand (single media filters) or sand and anthracite (dual media filters). Cartridge filters, which are typically made out of polypropylene or other soft polymeric materials [22], have to be replaced every 2-8 weeks depending on the raw water quality and the performance of the pretreatment [50]. For example, the cartridge filtration stage of the SWRO plant in Valdelentisco, Spain, consists of a total of 300 cartridges made from polypropylene [150]. Polypropylene is used for a wide variety of plastic parts, including food packaging or textiles. It can therefore be assumed that recycling of the cartridge filters is in principle a feasible alternative to landfill disposal. If UF pretreatment is used, additional cartridge filters may not be needed [50]. As the UF membranes provide a fixed barrier with a better removal capability than the cartridge filters, the installation of cartridge filters is often a precautionary measure.

In RO systems, stainless steels with a high corrosion resistance or non-metallic materials prevail, such as concrete or plastic. Stainless steels are by definition all iron-carbon alloys with a minimum chromium content of 10.5%. Different types of stainless steels are available, such as duplex, ferritic, austenitic or super-austenitic stainless steel. The corrosion resistance of steel is generally considered good when the corrosion rate is less than 0.1 mm/a [151]. When appropriate construction materials are used and the plant is designed properly, e.g., by eliminating dead spots and threaded connections, corrosion should be minimal [16]. Significant amounts of corrosion by-products are therefore not to be expected in the concentrate discharge of SWRO plants. All metallic parts and all non-composite non-metallic materials have a high potential for recycling.

MSF systems can be manufactured from a variety of materials but alloys of copper and nickel and various molybdenum bearing grades of austenitic stainless steels predominate, e.g., carbon steel (type 316L) clad with stainless steel for flash chambers or condensing section walls, and copper nickel alloys (types 90–10 and 70–30) for condenser tubing and condenser plates [149]. Limits are normally placed on corrosion by-products detectable in the distilled water, primarily to ensure equipment longevity. Copper and iron are often measured in the distillate with limits of 0.02 mg/l each [148]. Copper levels in the discharge can be a long-term environmental concern (cf. section 3.3.5, page 76).

In MED distillation plants, the prevalent construction materials are epoxy-coated carbon steel for effect vessels, external structure shapes and internal supports; aluminum, aluminum brass, titanium or copper nickel for effect tubing and tube plates; and stainless steel (grade 316) for pumps or demisters [149]. As the harshest operation conditions are encountered in the first three rows of the evaporator, most modern MED plants use titanium in this section of the plant. The other evaporator tubes are usually made from more economical materials such as aluminum alloy (AL-brass 76/22/2) [152].

2.4 Outfalls

The most widely used method of concentrate disposal for all desalination processes (SWRO and distillation) is surface water discharge. It is a relatively low-energy, low-technology and low-cost solution, assuming that the length of the pipeline is reasonable and the concentrate does not need further treatment. However, it has the potential for negative impacts on aquatic organisms, but the implementation of suitable mitigation measures, such as a good site location, an advanced outfall design with diffusers, or the pre-dilution with additional waste water such as cooling water prior to discharge, can likely minimize most potential negative environmental effects.

The discharge design primarily influences the mixing behavior in the near-field region, which extends up to a few hundred meters from the outfall location. In the near-field, a velocity discontinuity between the effluent and the ambient flow arises from the initial momentum flux and the buoyancy flux of the effluent. It causes turbulent mixing, which leads to an entrainment of seawater and thereby decreases differences in salinity, temperature or residual chemicals between the effluent and ambient water body. Ambient currents may deflect the jet trajectory, inducing higher dilution, whereas ambient density stratification has a negative effect on vertical spreading. Boundary interactions can occur, for example, at the water surface, the sea bed, or at pycnoclines. They generally define the transition from near-field to far-field mixing processes. The far-field can extend up to several kilometers and is dominated by ambient processes, such as passive diffusion, which cause a further slow mixing of the plume [153].

Diffusers
The use of multi-port diffusers can effectively increase the mixing process of the concentrate in the discharge site by increasing the volume of seawater in contact with the concentrate and by creating turbulent mixing conditions. A number of factors affect the dilution potential of diffusers, including the exit velocity and the volume of the concentrate, the depth of nozzles below the sea surface, the vertical angle of nozzles, and the number and spacing of nozzles [154]. The concentrate typically exits the diffuser nozzle at a high velocity and is directed in an upward slope towards the sea surface. With such a design, a salinity level of one unit above background levels can be achieved at the edge of the near-field mixing zone.

Two broad categories of concentrate outlet structures can be distinguished: rosette-style diffusers, which consist of several outlets risers above the seafloor with a small number of nozzles attached to each riser, and pipeline-style diffusers, which consist of nozzles arranged along a pipe instead of a rosette (Figure 10). All large Australian SWRO projects including the Victoria [154], Sydney [155], Perth [156] and Gold Coast plants [157] use or are proposed to use diffuser systems.

Sub-surface discharge
Brine disposal can also take place via a subsurface discharge structure. In coastal areas, beach wells or percolation galleries beneath the beach or seafloor can be used to induce mixing in the groundwater table to slowly dissipate the plume into the surf zone.

In Long Beach, California, an ocean floor demonstration system is currently being tested which combines seabed drains for the intake and a discharge gallery for the outfall, both located in the seafloor sediments [41, see also page 24].

A discharge gallery has also been in use at the Marina Coast Water District desalination plant (1,000 m^3/d) in California for ten years, which is one of the first plants to use such a system for brine disposal. In contrast to the demonstration facility in Long Beach, the injection well is located in the beach sediments, where the concentrate is diluted through mixing with natural groundwater, and is subsequently dissipated into the surf zone. A long-term monitoring programme concluded that there was not a detectable increase in salinity of the receiving waters due to brine discharge [39].

Subsurface outfalls are considered to be an effective way of minimizing the environmental impacts of concentrate discharge, at least in some locations where suitable hydrogeological conditions exist, and are more feasible for smaller SWRO plants. Another mature technology is deep-well injection of concentrate into a deep geological formation, usually inland, and isolated from drinking water aquifers. It is usually used for larger flows due to high development costs [5]. Other options for brine disposal include sewer discharge, evaporation ponds, land application or zero liquid discharge (ZLD).

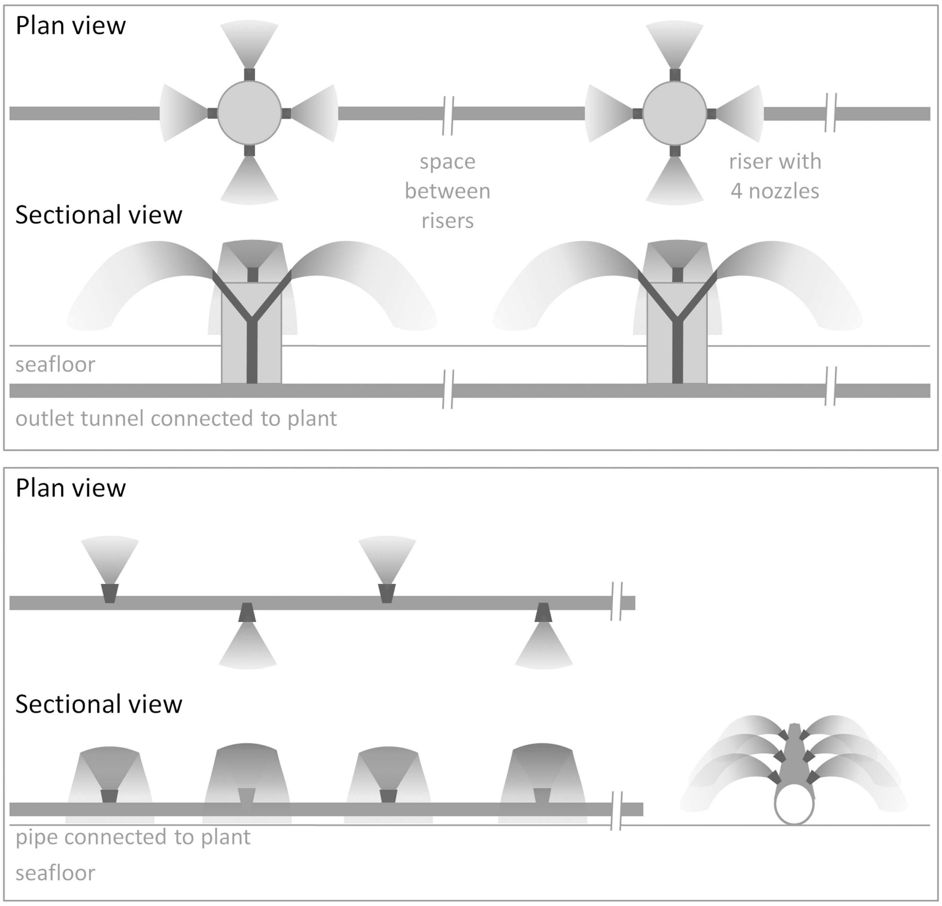

Figure 10: Rosette-style (top) and pipeline-style diffusers (below) for concentrate discharge.

They are mainly used where surface discharge is not possible, e.g., for inland BWRO plants, but usually not for SWRO plants [158].

Co-discharge with cooling water
Co-location of MSF/MED distillation plants with power plants is common practice, and some large SWRO plants were also co-located or are proposed to be co-located to power plants. Examples include the Carlsbad and Huntington Beach SWRO plants in Southern California [132, 159], the Tampa Bay SWRO plant in Florida [75], and the Ashkelon and Hadera SWRO plants in Israel [160, 161]. The main environmental benefits of co-locating SWRO plants to power plants are [162]:

- the use of existing intake and outfall structures, which reduces construction impacts,
- reduced land use and landscape impacts as the facility is constructed in an industrial area, and does not require additional power transmission lines,

- if the intake water is taken from the cooling water discharge conduits of the power plant, the required energy demand of the SWRO process can be reduced by 5-8% because of a higher membrane permeability at higher water temperature (mostly relevant for ambient seawater temperatures <20°C)[d],
- if cooling water is reused as feed water to the desalination process, the total amount of feedwater intake is reduced, limiting the impingement and entrainment effects to the level that is caused by the existing power plant,
- the concentrate can be blended with the cooling water before discharge, which significantly reduces the salinity of the concentrate before disposal.

Co-discharge with wastewater
Another option for co-discharge exists with wastewater treatment plant effluents, as proposed for two small SWRO plants in California [Santa Barbara and Santa Cruz, 39, 163] and Europe's largest SWRO plant in Barcelona [46]. The main advantage is that the salinity of the concentrate is very effectively diluted. A dilution ratio of 1:1 is sufficient to reduce the salinity of the concentrate to ambient seawater salinity levels, as the SWRO concentrate usually has twice the ambient salinity. However, there are several issues associated with the practice of blending SWRO concentrate with wastewater treatment plant effluents, and the discharge through an existing wastewater treatment plant outfall has found a limited application to date.

One consideration is the potential for whole effluent toxicity of the blended discharge that may result from an ion imbalance of the blend of the two waste streams [158]. Tests carried out in California clearly indicate that blending of wastewater effluent and desalination concentrate may have negative effects on some aquatic species, such as sea urchins and starfish (echinoderms), which were found to be most sensitive to the exposure of a blend of wastewater and concentrate. They are the only major marine taxa that do not extend into freshwater. As wastewater effluent has a freshwater origin with a different ratio of key ions than seawater, the ion imbalance may be responsible for the observed toxic effects [16]. Furthermore, residual contaminants in the waste water may have negative effects on marine life. However, both effects are attributed to the waste water and not to the concentrate from the desalination process. Consequently, they may occur wherever wastewater treatment plant effluents are discharged into the sea.

Another consideration is that wastewater may be considered as a resource, which should not be wasted to the ocean in water-scarce areas, as recycling is usually preferable to disposal in the concept of waste management. The question must also be raised if a new desalination project is necessary or could be reduced in size if the existing waste water sources were reused to full potential instead of being discharged into the ocean. Another argument for waste water reuse using desalination technologies is the elimination of a waste product. Effluents from conventional waste water treatment plants still contain diverse contaminants, including nutrients, metals, or micropollutants such as pharmaceutical and personal care products, which are a burden for many rivers, estuaries and coastal seas. Purifying and reusing waste water does not only produce a new water supply, but also eliminates a waste product if the remaining waste stream from the process is treated using zero liquid discharge (ZLD) technologies.

[d] As higher feed temperature also results in a higher salt passage, a higher feed temperature may also result in higher energy consumption if a second RO stage has to be implemented [22].

Water reuse is practiced in many parts of the world, but the use of desalination technologies in water reuse has been limited so far (cf. section 1.3). The world's largest facility treating waste water with an output capacity of 310,000 m^3/d is located in Sulaybia, Kuwait. It uses UF followed by RO to treat secondary effluent waste water. The energy demand of waste water desalination is lower than for SWRO due to the considerably lower salt content of the water, which is another environmental benefit. An expansion of waste water desalination is therefore expected in the future. A second advantage is that most of the water is already where it is most needed, i.e., near urban areas, avoiding long distance transport. The purified water can be used for industrial purposes, landscaping activities in urban areas, or aquifer recharge. From a technical point of view, the product can even comply with WHO drinking water standards [16, 164], but direct potable reuse, e.g., as practiced in Windhoek, Namibia, has found a very limited application to date.

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The main physico-chemical characteristics of the concentrate produced by SWRO, MSF and MED desalination plants are discussed – along with potential marine impacts – in the next chapter and summarized in Tables 13-15 on page 79ff.

Marine environmental impacts | 3

3.1 Introduction

The list of potential impacts of desalination plants on the environment is long and in some aspects similar to other development projects. The main concerns for the marine environment are the construction of intake and outfall structures, which may cause the temporary or permanent destruction of coastal habitats, the impingement and entrainment of marine organisms with the intake water, which may have an effect on ecosystem population dynamics, and the concentrate and chemical discharges into the sea, which may affect water, sediments and marine life if not well designed and managed. This chapter gives an overview on the main concerns for the marine environment.

3.2 Seawater intakes

Open seawater intakes usually result in the loss of eggs and larvae of fish and invertebrate species, spores from algae and seagrasses, phytoplankton and zooplankton, as well as smaller marine organisms that are drawn into the plant with the seawater (entrainment). Due to the pretreatment in desalination systems, which involves chlorination at the intakes to control marine growth, and the removal of suspended solids, it must be assumed that the survival rate of organisms which are drawn into the plant is minimal.

Parts of this chapter were based on:

S. Lattemann. Protecting the marine environment. In A. Cipollina, G. Micale and L. Rizzuti, editors. Seawater desalination, green energy and technology, pages 271−297, Springer-Verlag, Berlin Heidelberg, 2009.

S. Lattemann and T. Höpner. Environmental impact and impact assessment of seawater desalination. Desalination, 220: 1−15, 2008.

S. Lattemann, K. Mancy, H. Khordagui, B. Damitz and G. Leslie. Desalination, resource and guidance manual for environmental impact assessments. United Nations Environment Programme (UNEP), Nairobi, Kenia, 2008.

S. Lattemann and T. Höpner. Impacts of seawater desalination plants on the marine environment of the Gulf. In A. Abuzinada, H. Barth, F. Krupp, B. Böer and T. Al Abdelsalaam, editors. Protecting the Gulf's marine ecosystems from pollution, pages 191−205, Birkhäuser Verlag, Switzerland, 2008.

T. Höpner and S. Lattemann. Chemical impacts from seawater desalination plants, a case study of the northern Red Sea. Desalination, 152(1-3): 133−140, 2003.

Entrainment causes the loss of a large number of eggs, larvae and plankton organisms. The question however is if this represents a *significant, additional* source of mortality for the affected species, which negatively affects the ability of the species to sustain their populations, and which may affect the productivity of coastal ecosystems. These secondary ecosystem effects are difficult to quantify [39]. Plankton organisms are generally prevalent in coastal surface waters and have rapid reproductive cycles. Fish and invertebrate species produce large numbers of eggs and larvae to compensate for a high natural mortality rate as part of their reproduction strategy. The mortality caused by entrainment in a single facility therefore seems unlikely to have a substantial negative effect on population and ecosystem dynamics. The situation is different when cumulative sources of mortality (other power or desalination plants) exist and when endangered species, species of commercial interest, or marine protected areas are potentially affected by the intakes. While it is relatively straightforward to estimate the levels of entrainment for a single desalination project, it is difficult to evaluate the indirect impacts on the ecosystem, especially in places where cumulative sources of mortality are involved.

Furthermore, open intakes usually result in the loss of larger marine organisms when these collide with screens at the intake (impingement). Impingement mortality is typically caused by suffocation, starvation, or exhaustion due to being pinned up against the intake screens or from the physical force of jets of water used to clear screens of debris [39, after 165]. Experience from the Gulf indicates that consumption of cooling seawater by power-desalination plants may be a significant source of mortality. Several cases of massive fish kills were reported in the vicinity of power-desalination plants in the Gulf [166]. Impingement may also be a significant source of mortality for endangered or protected marine species, such as sea turtles or sea snakes, and even the intrusion of a three meter long whale shark into the intake of a SWRO plant in the Red Sea has recently been reported after a storm had damaged the intake structure [93]. Similar to entrainment effects, the cumulative ecosystem effects are difficult to estimate.

The cumulative impingement impacts of eleven power plants located on the Southern California coast were recently evaluated. Desalination plants will be co-located to two of these power plants in the near future and will use water from the cooling water discharge conduits as feedwater. The combined impingement mortality from the once-through cooling (OTC) systems of all eleven power plants was estimated to amount to 8-30% of the *recreational* fishing totals for Southern California [39, after 167]. However, it should be noted that power plants typically require much larger feed flows than desalination plants. If the intake velocity of the feedwater is reduced to velocities of about 0.1 m/s, which is comparable to background currents in the oceans, it can be expected that mobile organisms will be able to swim away from the intake area.

The California Coastal Commission concluded in 2004 that “the most significant potential direct adverse environmental impact of seawater desalination is likely to be on marine organisms: This impact is due primarily to the effects of the seawater intake and discharge on nearby marine life; however, these effects can be avoided or minimized through proper facility design, siting, and operation.” The U.S. Clean Water Act, section 316(b), requires that “the location, design, construction and capacity of cooling water intake structures reflect the best technology available for minimizing adverse environmental impact” (for impact mitigation by best available techniques see also chapter 6).

Table 12: Calculated salinity of SWRO plant reject streams.

	Feedwater salinity										
Recovery	30	31	32	33	34	35	36	37	38	39	40
30%	43	44	46	47	48	50	51	53	54	56	57
35%	46	48	49	51	52	54	55	57	58	60	61
40%	50	51	53	55	56	58	60	61	63	65	66
45%	54	56	58	60	62	63	65	67	69	71	72
50%	60	62	64	66	68	70	72	74	76	78	80
55%	66	69	71	73	75	77	80	82	84	86	89
60%	75	77	80	82	85	87	90	92	95	97	100

For feedwater salinities between 30 and 40 and recovery rates between 30% and 60%, assuming a permeate salinity of 0.3. The salinity values are derived by the equation $R_S = \frac{F_S F_F - P_S P_F}{R_F}$ where R_S is the salinity and R_F the flow rate of the reject stream, F_S the salinity and F_F the flow rate of the feed stream, and P_S the salinity and P_F the flow rate of the permeate stream.

3.3 Waste water disposal

The waste stream mainly contains the natural constituents of the intake seawater in a concentrated form. Depending on the process, environmental concerns may arise due to the high concentration of inorganic salts or the increased temperature of the waste stream, which may increase ambient salinity and temperature in the discharge site and may negatively affect local ecosystems. Furthermore, the pretreatment of the intake water involves chemical additives (Figure 8, page 26 and Figure 9, page 45), some of which are discharged along with the waste water. As seawater is a highly corrosive medium, the waste stream may also contain small amounts of metals that pass into solution when metallic parts inside the plant corrode. Although the following review of the waste water properties is formally subdivided into concerns related to the *physical* properties and those related to the *chemical* additives, *synergetic* effects of thermal and osmotic stress and effects caused by the exposure to residual chemicals should be anticipated. Tables 13 to 15 at the end of this chapter (page 79) provide a summary of the single physical and chemical effluent parameters discussed in the following sections.

3.3.1 Salinity, temperature and density

The salinity of the concentrate is largely a result of the plant recovery rate, which in turn depends on the salinity of the source water and process configuration. SWRO plants have higher recovery rates than distillation plants and therefore higher reject stream salinities, which typically range between 65 and 85[a]. Although the brine blow-down from the last stage in MSF distillation plants may have a salinity of almost 70, the brine is effectively diluted with a threefold amount of cooling water. Dilution results in a salt concentration that is rarely more than 15% higher than the salinity of the receiving water, while the SWRO brine may contain twice the seawater concentration. The brine and cooling water discharges of thermal plants are 5 to 15°C warmer than ambient seawater, whereas the temperature of the SWRO concentrate is similar to ambient values.

[a] The UNESCO definition of Practical Salinity Units (psu) is used, see footnote on page 46.

The concentrate discharge may lead to an increase in salinity in the discharge zone. The salinity increase can be controlled by pre-dilution with other waste streams, such as cooling water, or dissipation by a multi-port diffuser system, and discharge into a mixing zone that can effectively dissipate the salinity load due to strong wave action and currents (see also section 2.4, page 52). The increased temperature of the concentrate and cooling water discharge from distillation plants causes thermal pollution in the discharge site.

The density difference between the waste water and ambient seawater is a controlling factor for mixing and spreading of the discharge plume in the receiving water body. In shallow coastal waters, density is a function of salinity and temperature[b]. Due to the high salt content, the SWRO reject stream has a higher density than ambient seawater. For example, ambient salinity levels of 36-40 and seasonal temperature variations of 15-30°C, as typical for Mediterranean surface water, result in density variations of 1,023-1,030 kg/m^3. A SWRO plant with a feedwater salinity of 36, operating at 50% recovery, would produce a concentrate with a salinity of 72 (Table 12). At 20°C, the density of the concentrate is 1,053 kg/m^3, which is negatively buoyant compared to an ambient density of 1,025 kg/m^3. The plume would sink to the seafloor, unless it is adequately dissipated, forming a water mass of elevated salinity that would spread over the seafloor and might diffuse into the sediment pore water in the vicinity of the outfall pipe.

As salinity and temperature have opposite effects on density, the reject streams of distillation plants can either be positively, neutrally or negatively buoyant. They are often positively buoyant due to the influence of large amounts of cooling water discharge. For example, seawater salinities of 45 and temperatures of 33°C are characteristic of seawater in the Gulf. The reject water of a MSF distillation plant would be negatively buoyant compared to the ambient density (1,028 kg/m^3) at a salinity of 50 and a temperature increase of 5°C (1,030 kg/m^3), and positively buoyant at a temperature increase of 10°C (1,027 kg/m^3). However, the exact values should be calculated for the ambient and operating conditions over the course of a year in order to make reliable predictions. A discharge calculator which computes the effluent properties at the discharge point is currently under development within the MEDRC-funded project BrineDis [168]. It allows the input of up to three different effluents, which are merged at the discharge point, to allow for the blending of concentrate with wastewater or cooling water.

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The reject streams of SWRO and distillation plants generally affect different realms of the marine environment. The SWRO concentrate, if not dissipated, will spread over the sea floor and may affect benthic communities, whereas the reject streams of distillation plants will likely affect the whole water column for two reasons: the outfalls are usually located directly at the shoreline, i.e., in shallow water, and the plants have large discharge flow rates. Beyond the immediate discharge area, the plume will most likely affect pelagic (open water) organisms due to neutral or positive buoyancy. However, it must be pointed out that mixing and dispersal processes are largely influenced by site-specific oceanographic conditions. To analyze plume spreading in a specific project site, the existing oceanographic conditions need to be investigated and accompanied by modeling studies and density calculations.

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[b] Density is a function of salinity, temperature, and pressure (ρ (S, T, p)) and is calculated from in situ measurements of conductivity, temperature and pressure using the equation of state for seawater. For simplicity, often only the last two digits are used, which is the density anomaly sigma σ (S, T, p) = ρ (S, T, p) - 1,000 kg/m^3. Pressure is only relevant in deeper water layers and can be ignored in surface water with $\sigma_t = \sigma$ (S, T, 0).

A comprehensive review of the scientific and grey literature concerning monitoring and bioassay studies for desalination projects has been carried out and published in [17]. The studies used a wide range of approaches and methods to investigate the environmental impacts of desalination plant discharges. They were usually short-term – without ecologic baseline or effects monitoring, limited in scope – addressing only one effect such as elevated salinity, and localized – not taking transboundary effects into account like the dispersal of pollutants. In a nutshell, most fell short of recognizing the potentially synergetic effects of the single waste components on marine organisms and the complexity of potential responses from the ecosystems. While the possible risk of damage to the marine environment in close proximity to desalination plants is at hand, no final conclusions can yet be drawn concerning the long-term or cumulative impacts of desalination plants on the marine environment. The results from the more conclusive studies, and the main conclusions from this review, are summarized in the following.

Salinity is a vital environmental parameter for marine life but increased salinity can also be harmful or even lethal to marine life. In general, toxicity will depend on the sensitivity of a species to increased salt levels, its life cycle stage, the exposure time and the natural salinity variations of the habitat to which the species is adapted.

For instance, a salinity study on the Mediterranean seagrass *Posidonia oceanica* indicated that a salinity of 45 may cause 50% species mortality in 15 days and that a salinity of 43 may reduce plant growth rates by 50% [169]. In contrast, two related seagrasses from Western Australia, *P. australis* and *P. amphibolis*, seem to be more adapted to naturally high salinity levels, as densest covers are generally observed at a salinity of 40 to 50 [156, after 170]. The available studies indicate that even related species such as these *Posidonia* seagrasses can have a different tolerance of hypersaline conditions.

Some macro fauna taxa such as echinoderms (e.g., sea urchins, starfish), which are strictly marine species, also seem to be more sensitive to salinity variations than for example macro invertebrates which also occur in estuaries and which are able to adapt to a wide range of salinities. Young life cycle stages, such as sea urchin embryos, are also considered to be more sensitive than adults. Most marine organisms can adapt to minor deviations in salinity and might recover from extreme, short-term exposures to increased salinity. For example, *P. oceanica* plants that survived in a salinity of 43 over 15 days were able to recover when returned to normal conditions [169]. However, only few species will be tolerant of high salt concentrations over extended periods of time.

Marine organisms normally occur in those environments to which they are adapted and which provide favorable environmental conditions. This includes salinity ranges but also other environmental and biological factors such as temperature, food supply or competition among species. Natural salinity values vary between 30 and 37 in the Atlantic Ocean, between 36 and 40 in the Mediterranean Sea, between 37 and 43 in the Red Sea, and can range up to 60 in naturally saline environments of the Gulf. Salt concentrations that exceed considerably and continuously the ambient levels to which the native species are adapted may result in osmotic stress. This will drive mobile animals away from the discharge site and can cause a die-off of the sessile fauna and flora.

For example, salinity increases near the outfall of the Dhekelia SWRO on Cyprus were reported to be responsible for a decline of macroalgae forests, and echinoderm species were observed to have vanished from the discharge site [171]. Observations on the distribution on marine species from naturally hypersaline environments in the Gulf indicate that salinities above 45 alter the benthic community considerably [172]. This stresses the importance of salinity as a controlling environmental factor.

Similarly, thermal discharges may have an effect on species distribution by changing the annual temperature profiles in the discharge site. This could enhance biological processes by increasing seawater temperatures to favorable conditions in winter, but could result in thermal stress when critical values are exceeded in summer. Marine organisms could be attracted or repelled by the warm water, and species more adapted to the higher temperatures and seasonal pattern may eventually dominate in the discharge site of the distillation plant. In extreme cases, the thermal discharge may cause a die-off of sessile marine species. For instance, waste water discharges from the Taweelah MSF plant in the UAE probably caused a die-off of nearby mangrove stands [173].

Salinity and temperature thresholds should therefore reflect the local conditions, taking the sensitivity of endemic species and seasonal salinity and temperature variations into account. Some general and site-specific thresholds are given in the following:

- The World Bank guidelines for power plants recommend that the temperature of heated water be reduced prior to discharge to ensure that the discharge water temperature does not result in an increase greater than 3°C of ambient temperature at the edge of a scientifically established mixing zone, which takes into account ambient water quality, receiving water use, potential receptors and assimilative capacity among other considerations. For mixing zone regulations, it is recommended to use 100 m from the point of discharge when sensitive aquatic ecosystems are absent [174].
- Based on extensive field and laboratory studies of the seagrass *Posidonia* in Spain, it has been recommended to avoid discharges of desalination concentrate nearby Posidonia meadows, or to dilute the discharge salinity so that it neither exceeds a value of 38.5 in any point of the meadow for more than 25% of the observations (on an annual basis) nor a value of 40 in no more than 5% of the observations. These values compare to ambient salinities in the Western Mediterranean of 37-38 [175].
- The licence for the Perth SWRO plant in Western Australia specifies that salinity should be within 1.2 units of ambient levels within 50 m from the discharge point and within 0.8 units of background levels within 1,000 m from the discharge point. This requires a dilution factor of 45:1 at 50 m distance in all directions of the diffuser. After two years of operation, it was concluded that the actually achieved dilution factors ranged from 50-120:1. Whole effluent toxicity (WET) tests were performed with 5 native species at commissioning and after 12 months of operation, which showed that the required dilution factor to achieve a 99% species protection level is about 15:1, i.e., 99% of the species in the marine ecosystem will be protected at this dilution [176].
- For the Sydney SWRO project under construction, WET tests have been performed using various effluents and concentrations from a pilot plant and 5 indicator species. The toxicity testing verified that salinity was the key source of toxicity. A dilution factor of 30:1 at the edge of the near field is calculated to achieve salinity levels within natural variation levels of 1 unit above ambient, and to achieve the desired species protection level of 95%. Baseline studies are currently conducted for a comprehensive field monitoring programme to verify the results after start-up [177].
- For the Gold Coast desalination plant, field measurements during start-up confirmed an effective dilution factor of 60:1 at the edge of the near-field mixing zone at 60 m distance from the diffuser with no discernible difference to ambient salinity at that point. A minimum dilution ratio of 47:1 had been predicted by the hydrodynamic models. WET tests using effluent from the full-scale plant effluent were performed

on six distinct species from more than three trophic levels representative of the local ecosystem. The results imply that a minimum dilution factor of 9:1, corresponding to a salinity of 37.6 or an increase of 2.3 above ambient, should be achieved at the edge of the mixing zone to obtain a 95% species protection level [178].

- The most extensive WET testing was carried out for the Olympic Dam SWRO project. Basted on WET tests with 15 species from four trophic levels, it was calculated that a dilution of 45:1 should protect 99% of the marine species in the area, corresponding to a salinity increase of 0.7 units above ambient. The hydrodynamic modeling studies predicted that this dilution would be achieved within 300 m from the outfall in 90% of all times. In 100 m distance from the outfall, the minimum dilution would be 8:1 corresponding to a maximum salinity increase of 3.7 units or 9% above ambient. The maximum extent of the 45:1 dilution contour would be ≤1.1 km for 99% of the time. A dilution of 85:1 or a salinity increase of 0.4 units above ambient, which ensures 100% species protection at all times, would be achieved within 3.9 km of the outfall [179].
- In the U.S., EPA recommendations state that the salinity variation from natural levels should not exceed 4 units in areas permanently occupied by food and habitat forming plants when natural salinity is between 13.5 and 35 [132].
- For the proposed Carlsbad SWRO project in California, long-term salinity tolerance and toxicity tests were carried out. 18 marine species were held in a tank containing a blend of concentrate and power plant cooling water at a salinity of 36 (expected to occur in the mixing zone in 95% of the time, compared to an ambient salinity of 33.5). All organisms remained healthy and showed normal activity and feeding behavior during the 5 month test. Three indicator species were also exposed to salinities of 37-40 over an extended period of time with 100% survival and normal behavior at the end of the 19 day test. In 300 m distance from the point of discharge, the salinity near the bottom is expected to reach 34.4 under average and 40.1 under extreme conditions. An initial dilution factor of 15.5:1 has been assigned at the edge of the mixing zone [180].
- For a SWRO plant in Okinawa, Japan, a maximum salinity of 38 in the mixing zone and an increase of 1 unit where the plume meets the seafloor was established [181].

The World Bank standard of a 3°C differential at a generic scale of 100 m is, for example, regulatory practice for cooling water discharges in Qatar, but has been proven problematic for many industries to achieve. Cooling towers could be implemented where discharge regulations cannot be met. Cooling towers can reduce the cooling seawater volumes by approximately 60%, probably less in warmer climates, but the need for additional chemical additives within the towers, such as antiscalants, should be balanced against the benefits of a reduced thermal pollution [182].

The strict salinity thresholds, which have been established in some locations, underline that even minor salinity increases may be harmful to marine ecosystems, and should thus be avoided by advanced discharge designs. The total salt load, however, is not a concern for semi-enclosed sea areas, such as the Gulf, the Mediterranean or the Red Sea (cf. section 3.4, page 82), as natural evaporation exceeds the water abstraction rates by desalination plants by several orders of magnitude. The key to avoid impacts is to effectively dilute and disperse the salinity to ambient concentrations. The same argument, however, does not hold good in every case, particularly with regard to the chemical additives. While the salt is of natural origin, the additives are of anthropogenic origin, and some may have a tendency for accumulating in the environment. Dilution is therefore not a proper means of impact mitigation for some of these compounds.

3.3.2 Residual biocides

In most desalination plants, chlorine is added at the intake to control biogrowth on the screens, in the intake pipe and in the pretreatment line. The initial chlorine concentration declines inside the plant due to the oxidant demand of the seawater resulting from reactions with organic seawater constituents[c], and abiotic degradation (decomposition).

The common practice in Kuwaiti distillation plants, for instance, is to maintain a residual chlorine concentration of 0.5 mg/l at the outlets, but levels of total residual chlorine actually discharged into the sea are typically in the range of 0.1 mg/l [166]. At the outfall of a power-desalination plant in the UAE, the residual chlorine concentration was 0.25 mg/l in the year 1998 [185], which was reduced to 0.15 in 2005 [186]. Peak concentrations were up to 0.6 mg/l for one hour at high tide [186].

While chlorine levels can be assumed to range between 0.1 and 0.5 mg/l at the outlets of distillation plants depending on the dosage and oxidant demand of the seawater, they are very low to non-detectable in the reject streams of SWRO. This is because the water is usually dechlorinated with sodium bisulfite (SBS) after a short reaction time and before the water enters the RO units in order to prevent membrane damage by oxidation.

Following discharge, a further rapid decline in chlorine levels by up to 90% can be expected [187] due to the oxidant demand of the receiving water, dilution and decomposition. Environmental concentrations are hardly available. Field measurements in Kuwait Bay indicate that residual chlorine is between 0.03 and 0.5 mg/l in the waters adjacent to distillation plants [10, 188]. According to one study, a level of 0.5 mg/l is expected to be reduced to 0.05 mg/l at a distance of about 1 km [189].

Chlorine use in desalination and coastal power plants

Low-level chlorination of 0.5-1.0 mg/l with residual oxidant levels of 0.1-0.2 mg/l in the cooling water flow is routinely employed in coastal power plants [184]. This is generally similar to the practice in some distillation plants, where low chlorine doses are employed. The immediate decline after chlorine dosing is due to the considerable oxidant demand of the seawater. For instance, coastal power stations in the UK may require a tenfold chlorine dose to obtain a sufficient residual oxidant level in the cooling water [190].

The practice of low-level chlorination deliberately applies a chronic but not acute toxicity to the *sessile* species within the cooling water flow, which may also potentially affect the organisms in the receiving water in close proximity to the discharge point. It probably also causes the mortality of a proportion of the entrained planktonic organisms, depending on taxa, life stage and the thermal regime involved [190]. Following discharge, the effluent plume mixes with fresh seawater, and the sequential oxidant demand

[c] The great oxidizing capacity of chlorine leads to a high reactivity with water constituents. In freshwater, chlorine content is usually expressed as free available chlorine (FAC), which is the sum of molecular chlorine (Cl_2), hypochlorous acid (HOCl) and hypochlorite ion (OCl^-). In wastewater, where high levels of ammonia are present, chloramines are primarily formed, which also have oxidizing capacity and are referred to as combined chlorine. Coastal seawater has typically low ammonia concentrations but contains about 65 mg/l bromide (up to 80 mg/l in the Gulf), so that chlorination leads to the rapid formation (99% conversion in 10 seconds) of hypobromous acid (HOBr) and hypobromite (OBr^-), which are the main active species in seawater. Hypobromite/hypobromous acid reacts with organic seawater constituents, especially N-containing compounds, forming bromamines, which are as toxic as hypobromite/hypobromous acid, e.g., to mussels. Total residual chlorine (TRC) refers to the sum of FAC and combined chlorine, while total residual oxidant (TRO) also includes other oxidants such as bromine species [183, 184]. In accord with many publications, the terms chlorine or residual chlorine will mainly be used in the following, although TRC would be more precise.

rapidly negates the remaining toxicity of the water. This is why almost no measurable residual oxidant can be found beyond the point of discharge of coastal power plants [190].

For instance, TRO concentrations were measured along the length of the dispersing cooling water plumes of two seawater cooled power stations in the UK which employ low-level chlorination. In one plant, TRO concentrations of 0.02 mg/l were observed in the vicinity of the outlet (up to a distance of 600 m) and dropped below the detection limit (<0.01 mg/l) beyond 1,575 m. In the other plant, TRO levels of 0.02 mg/l near the outlet decreased even faster with distance [184]. The European risk assessment report for sodium hypochlorite evaluates the environmental exposure and effects caused by cooling water discharges from coastal power plants. Based on the measured value of 0.02 mg/l nearby the outfalls, which decreased to zero after mixing with the receiving water [184], it is concluded that there is no need for further risk reduction measures. Potential effects – if any – are only to be expected in the near vicinity of the outlet [191].

The European reference document on the application of best available techniques (BAT) in industrial cooling systems generally recommends to optimize the dosing regime based on monitoring, as industrial cooling processes are very site- and process-specific. As a primary BAT approach, the emissions of free residual oxidant from a once-through cooling (OTC) system should be ≤ 0.2 mg/l at the outlet for continuous chlorination of seawater as a 24-hour average, and ≤ 0.5 mg/l for intermittent and shock chlorination as an hourly average within one day [192]. The World Bank Pollution Prevention and Abatement Handbook likewise recommends a maximum total residual chlorine/bromine concentration of 0.2 mg/l for effluents from thermal power plants [174]. Where shock chlorination is applied, the maximum value is 2 mg/l for up to 2 hours, not to be repeated more frequently than once in 24 hours, with a 24-hour average of 0.2 mg/l.

The BAT and World Bank reference values are not mandatory, as discharge regulations are normally established at the national or site-specific level. While some countries may follow these recommendations, others may adopt more or less strict regulations. In Qatar, for instance, the EU and World Bank standard is challenged by recently introduced regulations which require an incremental reduction of the maximum chlorine concentration permitted in discharged cooling seawater from 0.2 to 0.05 mg/l [193]. As most industries struggle to meet the new requirements, they make an effort to facilitate the Ministry of Environment in revising the regulations [194]. Worldwide, regulatory authorities exert pressure on industries to diminish their chlorine use, driven by the acute toxicity of chlorine and the formation of chlorination by-products [195]. In Venice Lagoon, for example, chlorine use has been banned because of its adverse side-effects [196].

To reduce the discharge of free residual oxidant, the required chlorine dosage should be established based on target species behavior and the seawater quality parameters. For example, different chlorine doses may be needed in summer and winter time. Pulse-chlorination has been applied in some coastal power plants and other seawater cooling applications and can be considered as best available technique [183, 192]. The environmental benefit of pulse chlorination is that a significant reduction in total chlorine use in the range of 30% to 50% can be achieved whilst ensuring an effective control of macrofouling in OTC systems, as for example, proven in power plants in The Netherlands. Pulse chlorination does not apply chlorine as a toxicant, but as a trigger which forces bivalves to switch between open and closed valves, i.e., aerobic and anaerobic metabolism, which leads to exhaustion – and ultimately death. The lowest practicable levels of total residual oxidant that can be achieved with pulse chlorination in environmental conditions is between 0.05 and 0.15 mg/l as a monthly mean [183].

To conclude, chlorine dosing levels in distillation plants and coastal power plants are similar at the lower end of the concentration range, but dosing levels in distillation plants exceed those in coastal power plants at the upper end of the concentration range. While power plants routinely use low-level chlorination in doses of 0.5-1.0 mg/l with resulting oxidant levels of 0.1-0.2 mg/l in the cooling water, distillation plants reported dose levels of 0.4-4.0 mg/l with resulting oxidant levels of 0.1-0.5 mg/l at the point of discharge. Environmental data is scarce, with levels of 0.02 mg/l being observed in the vicinity of power plant outlets and 0.03-0.5 mg/l being observed in the vicinity of distillation plant outlets. The EU BAT value for power plants is a chlorine residual ≤ 0.2 mg/l at the outlet for continuous chlorination and ≤ 0.5 mg/l for intermittent chlorination. The information available on chlorination practices in distillation plants suggests that waste water discharges can be in compliance with BAT levels for coastal power plants where low-level chlorination is used or where the oxygen demand of the seawater is sufficiently high so that discharge concentrations of ≤ 0.2 mg/l are met at the outlet.

Potential impacts of chlorine use

Although residual chlorine levels in the discharge are likely to meet BAT standards and further decrease quickly in the environment, the potential for adverse effects still exists in the mixing zone of the plume. According to Taylor [190], very few instances of acute toxic effects have been observed beyond a power station outfall where *low-level chlorination* is used. However, impacts will mainly depend on the dosing levels and specific local conditions. Observations from distillation plants are scarce. In one case, effluents with high chlorine levels were reported to affect mud flats in the Bay of Kuwait [197], which can probably be attributed to the Doha power and desalination plant [198].

Chlorine is a very effective biocide and its toxicity has been confirmed in many toxicological studies. Based on studies from a wide spectrum of marine species, the U.S. EPA recommends a short-term water quality criterion of 13 μg/l (criterion maximum concentration, CMC) and a long-term criterion of 7.5 μg/l (criterion continuous concentration, CCC). These are estimates of the highest chlorine concentration in seawater to which an aquatic community can be exposed briefly (CMC) or indefinitely (CCC) without resulting in an unacceptable effect. The California Ocean Plan specifies a daily total residual chlorine maximum of 8 μg/l in seawater and a 6 month median of 2 μg/l which should not be violated by a discharge [199]. Although the toxicity depends very much on species sensitivity and life cycle stage, the data indicates that chlorine is very toxic to aquatic life at concentrations in the low parts per billion (ppb or μg/l) range [200].

As part of the EU environmental risk assessment of sodium hypochlorite, ecotoxicity data of freshwater and marine species were reviewed and a predicted no effect concentration (PNEC) of 0.06 μg/l total residual chlorine was established based on fish, invertebrate and algae toxicity data [191]. If the predicted environmental concentration (PEC) exceeds the PNEC, a need for limiting the risks is usually identified. Residual chlorine concentrations in seawater of 0.02 mg/l, as observed near power plants up to a distance of 600 m from the outlet [184], clearly exceed the PNEC value derived by the EU risk assessment by two orders of magnitude. It is difficult to understand the reasons underlying the conclusion of the EU risk assessment for the power plant cooling water scenario that there is “no need for ... risk reduction measures beyond those which are being applied already”. The main argument given to back this decision is that environmental chlorine levels will “rapidly drop to zero when reaching surface water”, which means that “potential effects − if any − may only be expected in the near vicinity of the outlet”. The problem is that paraphrasing terms such as “rapid” or “near” are fuzzy criteria which give little guidance and a lot of leeway to environmental regulators and operators.

From a regulatory viewpoint, aquatic pollutants are typically regulated at the point of discharge (emission standards, ES) or as water quality objectives within the receiving water body (ambient standards, AS) or both (combined approach). While ES encourage source control principles, such as effluent treatment, AS can be associated with the concept of a mixing zone, where gradual mixing in the water body reduces the pollutant concentration to the AS, which must be met at the edge of a defined mixing zone [201]. The ES/AS ratio expresses the necessary dilution that must be attained through physical mixing or – to some extent – through biological decay and chemical transformation processes. The ES/AS ratio is 27 for chlorine, based on a discharge limit of 0.2 mg/l (EU BAT and World Bank guidelines) and a water quality objective of 7.5 μg/l (U.S. EPA).

In an open-coast location, high dilution rates can easily be achieved, but in less exposed locations, mixing zones can extend over considerable areas in the water body. Most regulations do not provide any information on where the AS apply. This may result in highly variable interpretations, from very restrictive (where the mixing zone is reduced to a very small area, i.e., the AS applies to almost the entire water body and must basically be met at the end-of-pipe) to non-restrictive (the boundary of the mixing zone is wherever mixing is complete). Thus, a “combined approach” requires a regulatory mixing zone definition which takes the characteristics of the water body into account. In order to meet mixing zone regulations, properly sited outfalls with optimized high efficiency mixing designs are typically needed [168].

Chlorination by-products (CBPs)

Potential impacts also result from the formation of chlorination by-products. Due to many possible reactions of hypochlorite and hypobromite with organic seawater constituents, by-product diversity is generally high.

Trihalomethanes (THMs) such as bromoform account for most of the compounds. Increased THM levels near distillation plants of 9.5 μg/l [188] and up to 83 μg/l [202] have been reported. These values are by a factor of three higher than the mean bromoform concentrations reported in the effluents of different coastal power plants that use chlorine for disinfection, and where values ranged between 3.5 μg/l and 25.16 μg/l [190]. Samples taken within the dispersing plume of power plants showed bromoform concentrations up to 14 μg/l and were traceable (0.26±0.1 μg/l) even in the far field up to 15 km from the outlet [184]. Samples taken within the discharge site of cooling seawater from liquefied natural gas production in the Gulf, to which chlorine doses of 0.5-1.5 mg/l were applied (similar to power and desalination), showed bromoform concentrations up to 105 μg/l below the outfall and were traceable (< d.l. of 0.1 μg/l) beyond 5 km from the outfall [203]. Significantly increased concentrations of dibromoacetonitrile (DBAN) were also found in many effluents, but concentrations of other halogenated organics, such as haloacetic acids, are usually considerably lower. Oil pollution may give rise to compounds like chlorophenols or chlorobenzenes [184, 202, 204, 205].

While the toxicity of the applied oxidant (i.e., chlorine) is known to decline rapidly with dilution and because of the oxidant demand of the ambient seawater, the same cannot necessarily be said of the more chemically stable by-products. For THMs like bromoform, the main route of loss from the water is through volatilization, with a half-life in a water body 1-2 m deep of 1-2 days, and reported aerobic biodegradation half-lives of around 1 month. The half-lives of other chlorination by-products in seawater were reported to be between several days and several weeks [190].

The concentrations of chlorination by-products reported for the effluents of coastal desalination and power plants were far below reported *acute* toxicity levels. However, recorded data in the literature is limited and long-term *chronic* exposure studies have not been published [184]. Some of the by-products such as chlorinated hydrocarbons are persistent, some compounds tend to accumulate in the fatty tissue of aquatic organisms, and some show a chronic mutagenic and carcinogenic toxicity [192].

It is not possible to derive toxicity data for all by-products. Ecotoxicological data in connection with the assessment of seawater chlorination, however, suggest that the ecotoxicities of the brominated THMs are not markedly different from chloroform. In the EU risk assessment, it was therefore concluded that the aquatic toxicity of total THMs can be broadly assessed by using the PNEC for chloroform, which is 146 µg/l for freshwater species [191]. A study reviewing existing water quality standards for CBPs reported only one existing value, a limit of 12 µg/l chloroform as average annual concentration [190].

Comparing the PNEC for total residual chlorine (0.06 µg/l) and chloroform (146 µg/l, representative of bromoform in absence of any actual data), it can be concluded that the toxicity of free oxidants is considerably higher than the toxicity of the chlorination by-products. The effluent and ambient concentrations of residual chlorine near power and desalination plants are well within a range that can be *acutely* toxic to some marine species, while the reported bromoform levels are far below reported *acutely* toxic levels. Since residual chlorine is rapidly degraded and removed from surface water by further reaction with organic material, the primary mitigation measure for chlorine is to minimize the discharge concentration and the mixing zone in which quality standards are exceeded. Since chlorination by-products are more persistent, and some have chronic carcinogenic and mutagenic properties, the focus should be on reducing their formation. Dechlorination will remove chlorine toxicity and will considerably reduce the potential for by-product formation. However, studies investigating the toxicity of chlorinated-dechlorinated seawater observed increased mortality [206, 207] and chronic effects [208] of test species even in dechlorinated seawater. The observed effects were assumed to be due to the presence of halogenated organics formed during chlorination.

Alternatives to chlorination

Due to environmental and health issues raised by residual chlorine and disinfection by-products, several alternative pretreatment methods have previously been considered for use in desalination. The methods have been successfully used in freshwater treatment or reuse applications, and some have been developed and piloted for application in a limited number of (usually smaller) seawater desalination plants. None, however, has gained wide acceptance over chlorination so far.

Especially in distillation plants, which require large feedwater flows and therefore large quantities of disinfectants, chlorination is usually the preferred option due to its low cost and high efficiency. A comparison between several alternative pretreatment methods for distillation plants in terms of environmental and health impacts, effectiveness, technical feasibility and costs found that no alternative method superseded chlorination followed by dechlorination [209]. Although this study was published in 1992, dechlorination in distillation plants has not been reported for a single plant.

Several alternatives to chlorination have also been put forward for SWRO. Chemicals included sodium bisulfite (SBS) [210], monochloramine [211, 212], ozone [210], chlorine dioxide (page 69) and 2,2-dibromo-3-nitrilopropionamide (DBNPA, page 70) [213]. A non-chemical alternative is *UV-light* of 200-300 nm wavelength. It is not presently used in large desalination plants, but some smaller plants have claimed to use UV light successfully [214]. *SBS* is a reducing agent, which exerts a biocidal effect by depleting oxygen levels, and is therefore not effective against anaerobic bacteria. *Ozone* converts bromide into hypobromous acid, which is also the main active species in seawater chlorination, and therefore does not eliminate environmental concerns [209].

Reported advantages of *monochloramine*[d] are that it will less likely degrade high molecular organics into assimilable nutrients [211, 212], and that it will less likely produce chlorination by-products in seawater and RO permeate [215], but field testing failed to yield conclusive positive results in some Middle-Eastern desalination plants [216]. Chloramine has a reportedly lower germicidal effect than chlorine, and can be tolerated by some thin-film composite SWRO membrane, e.g., Filmtec membranes. This implies that dechlorination may not be required when chloramine is used as a disinfectant. However, since chloramine is formed by adding ammonia to chlorine, it is possible that free chlorine will still be present under certain conditions of pH, temperature and the ratio of chlorine to nitrogen. As free chlorine can be damaging to the membranes, dechlorination should therefore still be considered [217, 218].

Chlorine dioxide

For seawater applications, there is a growing interest in the application of chlorine dioxide (ClO_2). The substance is – like chlorine – a powerful oxidant, but requires a shorter contact time and dosage. Efficient antifouling control is achieved at concentrations in the range of 0.05-0.25 mg/l, which can be increased intermittently to 0.4-0.5 mg/l for some hours per day [192]. Power plants which use chlorine dioxide report either no residual oxidant levels or a maximum of 0.1 mg/l at the point of discharge [219]. Chlorine dioxide is also used in some distillation plants in the Gulf region [220] and is used in the Tampa Bay SWRO plant in Florida [75]. In Tampa, chlorine use was abandoned due to elevated disinfection by-product levels. Chlorine dioxide is now used followed by sodium bisulfite to remove residual oxidants ahead of the RO membranes [5].

In a pH range of 6–8.5, chlorine dioxide remains as a dissolved gas in solution. Unlike other oxidants such as chlorine or ozone, it does not readily react with bromide to form bromine, or with ammonia to form chloramine. Furthermore, it does not favor addition and substitution reactions which would produce chlorination by products such as THMs and DBAN. Analytical tests performed at the discharge point of a power plant confirmed the absence of these compounds in the water under the measurable limit of 0.1 μg/l [196]. A study on a large coastal power station in Spain with an OTC system reported significantly reduced THM formation when chlorine dioxide was used instead of hypochlorite, irrespective of reaction temperature or reaction time. THM levels ranged from 0.3 μg/l (0.5 mg/l ClO_2 dose, 10 min. reaction time, 15°C) to 460.5 μg/l (0.4 mg/l ClO_2 dose, 60 min., 60°C) [192, after 221]. These values, however, are partly higher than reported bromoform concentrations between 3.5 and 25.1 μg/l in the cooling waters from power plants in Northern Europe which used chlorine for control of marine growth.

Minimal impact is predicted by the intermittent discharge of chlorine dioxide into marine waters which undergo rapid mixing based on toxicity tests with species from three trophic levels. Effects were only observed at high chlorine dioxide doses of ≥25 mg/l. The study found chlorine dioxide to be markedly less toxic, with NOEC concentrations a thousand times higher than for total residual chlorine. However, the study concludes that the results probably underestimate the effects from continuous exposure, citing 96 h LC_{50} values for fish species in the range between 20 and 170 μg/l [222].

[d] Monochloramine is often used synonymously to the term chloramine (NH_2Cl). The term chloramine also refers to a family of organic compounds with the formulas R_2NCl and $RNCl_2$, with R as an organic group.

DBNPA
2,2-dibromo-3-nitrilopropionamide is used in RO units for oil platforms and pulp factories, and increasingly also for water reuse and desalination. For example, DBNPA has been successfully used in a BWRO plant in Malaga [213], in a SWRO plant in Curacao [131], and is also being considered for the Fujairah SWRO, the largest facility in the Middle East (200,000 m^3/d), as a further increase in shock chlorination was not sufficient to control marine growth in that plant [223]. DBNPA use is compatible with UF, MF and chlorine, but the presence of reducing agents such as SBS should be avoided [224].

DBNPA is a non-oxidative biocide which is effective against anaerobic and aerobic bacteria, fungi and algae. It deactivates biofouling organisms quickly, and is rejected by the membranes. This makes it suitable for on-line addition to the pretreatment system and RO feedwater, but offline use 2-3 times per week is currently recommended for desalination plants by the chemical manufacturer. The trains are taken offline and soaked with DBNPA solution with an active biocide concentration of 20 ppm (100 ppm with 20% active ingredient). Another non-oxidizing biocide used in desalination is isothiazole. Due to a slower reaction time, it is more commonly used in membrane preservation [213].

DBNPA did not totally meet the criteria of a "ready biodegradation" classification, but it rapidly degrades at environmentally realistic concentrations and conditions via different pathways. Degradation begins at the time of introduction to the system. The ultimate degradation products are carbon dioxide, ammonia, and bromide [224–226].

DBNPA rapidly hydrolyzes above pH 6 and in warm waters[e]. Despite the expected rapid decay in ambient conditions, concern was raised about the first degradation product, dibromoacetonitrile (DBAN), which is about three times more toxic to fish and has a longer half-life than DBNPA[f] [228]. DBAN is also a major by-product in seawater chlorination (cf. page 67). The third and most stable of the hydrolysis products, dibromoacetic acid, however, is about 100 times less toxic to fish than DBNPA[g] [227].

A second reaction route is with organic matter present in surface waters, which is equal to or even faster than the hydrolysis route via DBAN, and rapidly yields a one-bromine derivative, monobromonitrilopropionamide (MBNPA), before losing the second bromine to give cyanoacetamide (CAM) in the range of hours. MBNPA is half as toxic as DBNPA. A high ratio of TOC to DBNPA, as expected under environmental use conditions, thus results in much lower aquatic toxicity than one would predict from just an LC_{50} value for DBNPA and hydrolysis of DBNPA [228].

The compound furthermore decomposes under the influence of sunlight (<1% active substance remaining after four weeks), and can be rapidly reduced by a number of sulfur-containing species such as bisulfite. Both processes lead to CAM, which is also biodegradable. DBNPA is furthermore microbially and chemically degraded in soils, and its low lipid affinity suggests that it does not bioaccumulate [227].

The hydrolytic, photolytic, chemical, and microbial decomposition of DBNPA and its degradates suggests that these compounds will not persist in the environment [227]. The dominant degradation pathways under use conditions are reaction with nucleophilic substances (such as bisulfite) or organic material (TOC) in surface water [226], giving rise to degradation products (MBNPA, CAM) which are less toxic than DBAN.

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[e] Half-life of 145 hours at 0°C and pH 7.7, 5.8 hours at 25°C and pH 7.7, 2 hours at 25°C and pH 8.0 [227].
[f] Half-life of DBAN was extrapolated to be about 6 days [228].
[g] Sequence of degradation products by hydrolysis: dibromoacetonitrile (DBAN), dibromoacetamide (DBAM), dibromoacetic acid, glyoxylic acid, and oxalic acid.

An ecological risk assessment for the use of DBNPA in OTC systems, based on a set of toxicity data for freshwater and marine species and a conservative exposure scenario, led to the conclusion that DBNPA does not present a significant risk to the aquatic environment. The test data for all organisms indicated that once the toxic threshold is achieved, the entire population of organisms of that species is affected. The exposure scenario assumes that DBNPA is added weekly for up to 3 hours, leading to a calculated effluent concentration of about 19 ppm. Because of the episodic treatment regime and the short half-life of the compound in surface water, organism exposure to DBNPA is assumed to be episodic and short, so that chronic exposures are expected to be extremely rare in the environment [229]. The potential for localized mortality is only given if a large amount of biocide is discharged. DBNPA is one of only a few biocides registered with the U.S. EPA for OTC systems, which allows discharge in accordance with a National Pollutant Discharge Elimination System (NPDES) permit [226].

To conclude, the intermittent *offline* use of DBNPA in SWRO plants as proposed by [213] does not seem to present a significant risk to the marine environment. The treatment that is proposed is similar to the use of DBNPA in OTC systems. The exposure of marine organisms can therefore be assumed to be episodic and short. As a further precautionary measure, it is recommended to inactivate excess DBNPA with sodium bisulfite before discharge, or to divert the cleaning solution to a storage tank for self-decomposition before discharge to a waste water treatment facility or surface water discharge.

Conclusion
To conclude, no alternative biocide has gained wide acceptance over chlorination in desalination applications so far. The environmental assessment of seawater chlorination comprises two different groups of chemicals, the oxidants and the chlorination by-products. Both differ in terms of ecotoxicity, bioaccumulation and biodegradation. The toxicity of the free oxidants is high, even at low concentrations, but they decompose quickly and do not bioaccumulate. Toxicity to non-target organisms, if any, will therefore be limited to the mixing zone. Some of the chlorination by-products, however, can be expected to be more persistent, to bioaccumulate, and/or to show a chronic mutagenic and carcinogenic toxicity, possibly also beyond the mixing zone. To minimize negative effects from both chlorine and by-products, chlorine doses and discharge levels should be reduced as far as is consistent with an effective pretreatment strategy. BAT is low-level or pulse-chlorination with ≤ 0.2 mg/l chlorine at the outlet. In some regions, stricter values have been implemented or chlorine use has been banned completely. Emission standards for chlorine should be combined with ambient water quality standards and clear mixing zone regulations in a combined approach. In this respect, where the release of chlorinated effluents is associated with an increased temperature, the synergetic effects of thermal stress and residual chlorine, which have been demonstrated in many studies, need also to be taken into account [191].

3.3.3 Coagulants

In SWRO plants, conventional pretreatment relies on a combination of different chemicals for coagulation-flocculation and media filtration for the removal of suspended solids (section 2.3.1). Primary coagulants are typically metal salts like ferric chloride ($FeCl_3$) and ferrous sulfate ($FeSO_4$). Their cations neutralize the negative surface charge of the suspended particles and the formed hydroxide flocs adsorb and enmesh smaller colloid particles, so that particles are aggregated into larger and more filterable solids [22]. Conditioning with sulfuric acid (H_2SO_4) to pH 6-7 and dosing of coagulant aids, which have similar properties to both polymers (high molecular weights) and electrolytes (charged compounds), can enhance the coagulation process. UF pretreatment often involves in-line coagulation with metal salts only and in lower doses (section 2.3.2).

Doses are normally correlated to the amount of suspended material in the intake water and typically range between 1-6 mg/l $FeCl_3/FeSO_4$ (0.4-2.4 mg/l as Fe assuming 40% active ingredient) in conventional pretreatment (Table 5) and between 0.2-2 mg/l as Fe in full-scale UF plants (Table 6), although values up to 8 mg/l as Fe respectively 10 mg/l as Fe have occasionally been reported. Acid and coagulant aid doses in conventional pretreatment are typically between 20-50 mg/l and 0.2-2 mg/l, respectively.

The filter backwash water, which contains the natural suspended material and the coagulant chemicals, can either be discharged into the sea or can be dewatered and the sludge disposed off in a landfill. Discharge seems to be the standard practice in UF plants, whereas an increasing number of SWRO plants with conventional pretreatment have a waste water and sludge treatment step before discharging the supernatant.

For example, backwashing in the Ashkelon plant was carried out for 10 to 20 minutes every hour, producing about 6,500 m^3/h of backwash with an intermittent peak iron content of 40-50 mg/l and resulting in an estimated iron discharge of 500-650 tons in 2007. The backwash plume has been observed to disperse over a considerable distance with the potential to affect a nearby marine reserve [80, Figure 11]. The iron load corresponds to an equivalent continuous average discharge of 3.5 mg/l, which was reduced to 1.8 mg/l Fe in 2008, which is below the discharge standard of 2.0 mg/l. The standard shall be reduced to a mean of 0.3 mg/l in the future. The Ministry for the Environment has furthermore inserted a request to treat the backwash waters by dewatering and landfilling for new tender documents in Israel, which shall remove 90% of the iron [81].

When discharged to the sea, the filter backwash may significantly increase the amount of suspended matter in the discharge site, which may be an aesthetic problem as iron salts can turn the mixing zone of the backwash plume into a deep red-brown color. The coagulant chemicals, which are also commonly used in conventional drinking water treatment systems, are generally non-toxic to aquatic life. Iron is a natural element in seawater and a nutrient required for algae growth. It is often limited in open ocean waters but usually not along the coastal shelf. Still, a main concern of the Israel Ministry for the Environment is a potential eutrophication of coastal waters. The discharge of large sludge volumes may also cause physical effects that can have negative impacts on marine life. Increased turbidity and lower light penetration could affect the productivity of benthic macroalgae, seagrasses or corals if present in the discharge site, and sedimentation of the material may blanket benthic plants and animals.

Prior to disposal on land, several levels of treatment may be required, including clarification, thickening and sludge dewatering, depending on the feedwater quality and the volumetric sludge production. A worst case scenario would require a thickener followed by a sludge dewatering system, using lamella settlers, a belt press or centrifuge in a separate building with odor control, e.g., as in the Tampa Bay SWRO facility. Small sludge amounts may be dewatered in a simple and relatively inexpensive sludge drying bed on-site or the liquid sludge may simply be disposed of in a landfill without treatment [230]. The clarified backwash water, which contains about 1% of the particulate material retained in the pretreatment filter, is normally discharged into the sea [132].

For a large SWRO plant in Israel, the cost of backwash treatment was calculated to be 5 US¢/m^3, which represents an approximate 10% increase in the current product water cost of 60 US¢/m^3. The total cost includes capital and operational costs, transportation

Figure 11: Discharge of filter backwash water from the Ashkelon plant in Israel. Photo: courtesy of Rani Amir, Director of the Marine and Coastal Environment Division, Israel Ministry of the Environment. A red plume appears approximately every hour for 10-20 min. when untreated filter backwash water, with high iron concentration up to 42 mg/l, is discharged with the concentrate. The plume has been observed along the seashore up to a distance of 3 km from the outfall [80].

of the solids to a landfill over a larger distance and the landfill cost [231]. This can be assumed to be a worst case situation. Electricity costs for treatment are assumed to be 0.02-0.05 kWh/m^3 depending on the sludge amounts [53, Table 9, page 47].

Many SWRO plants, especially smaller and older projects, discharge the backwash waters without treatment (Table 5). More recent projects where the sludge is also discharged into the sea are the Ashkelon plant in Israel and the Hamma plant in Algeria with capacities of 320,000 m^3/d and 200,000 m^3/d, respectively. However, plants in Israel will either collect the backwash water in a storage tank before blending it with the concentrate to avoid turbidity peaks, or will have sludge treatment in the future. Especially in upcoming projects, such as the Sorek project with a projected capacity of 820,000 m^3/d, sludge treatment is the favored option of the Ministry for the Environment, who estimate the cost increase to be about 1 US¢/m^3 [81].

To conclude, a trend towards sludge treatment can be observed in an increasing number of countries despite the cost increase, and most of the new, large SWRO projects underway in Australia, the U.S. and Europe will also have a full sludge treatment. The Sydney SWRO plant in construction (250,000 m^3/d, potential upgrade to 500,000 m^3/d), for instance, will have treatment facilities on site where the sludge will be dewatered and then sent to landfill [63], although sludge disposal to the ocean was initially considered in the EIA and project description [155, 232].

The environmental effects statement (EES) for the Victoria SWRO project in Melbourne (410,000 m^3/d, potential upgrade to 550,000 m^3/d) differentiates between the reference project, which assumes sludge disposal in a landfill (20-100 m^3/d) and recycling of the clarified water (supernatant) to the head of the plant, and alternative options for sludge disposal [154]. The EPA, an inquiry commission and an independent expert group support the disposal of pretreatment wastes to landfill rather than discharge into the ocean, because it is the "best practice approach being adopted by desalination plants currently being commissioned or designed in Australia and overseas". Inclusion of pretreatment sludge into the saline effluent would furthermore require additional toxicity evaluation, a higher level of dilution and a larger mixing zone, and may potentially create toxicant, nutrient, discolouration and deoxygenation effects. The quantity of sludge might be reduced by the use of membrane filtration technologies, which were also outlined in the EES as possible variations within the reference project. It is assumed that membrane filtration requires less coagulant to remove suspended solids and organics from the intake seawater, and hence could reduce quantities of wet sludge [233].

All other projects in Australia with capacities up to 140,000 m^3/d, such as the Perth and Gold Coast projects (in operation) and those under development (Olympic Dam, Perth II) either have or will have a sludge treatment [69, 157, 179].

The largest operational SWRO plant in the U.S. (Tampa Bay, Florida, 90,000 m^3/d) also has a full sludge treatment. In California, the biggest and most advanced projects are the plants in Carlsbad and Huntington Beach. While the EIA for the Carlsbad project states that the sludge will be dewatered on site to sludge concentrations of 20% or higher and disposed of in a landfill, the EIA for the Huntington Beach project considers ocean disposal. However, the California Coastal Commission expects that both projects will have the same requirements, that is landfill disposal. The preference is for a facility to avoid ocean discharge − through landfilling or routing to a wastewater treatment system. Only if those alternatives are infeasible or unavailable, a desalination facility may be able to receive a permit for ocean discharge that is subject to water quality parameters [234].

The largest SWRO plant in Europe in Barcelona, Spain (200,000 m^3/d) will have a complete sludge treatment as well, as does a smaller plant near Marbella [46]. Little, however, is known about other SWRO plants in Spain. Although all plants within the Spanish AGUA programme (cf. also page 17) are generally required to have a sludge treatment according to tender documents and EIAs [46], not all Spanish plants have such a treatment in practice, possibly due to special circumstances. In the plants in Almeria, Torrevieja and Aguilas for example, coagulants are often not required due to a good raw water quality [235]. In these cases, the only remaining solids in the backwash are of natural origin. In principle, all plants should have a sludge treatment, but even industry insiders who frequently visit the facilities are not sure about the practice, as plant managers may not be willing to share this information [60].

The backwash sludge is a waste product without any real beneficial reuse to date. It cannot be used in agriculture because of its salt content [46]. Some potential reuse and recycling applications that were considered for the Melbourne SWRO project included acid treatment of the sludge to regenerate a lower grade coagulant which could be directly recycled at the plant or could be used as a raw material to produce other useful chemicals, such as an iron-based catalyst for arsenic removal from contaminated water or phosphate removal from wastewater [154]. In Sydney, research into beneficial reuse continues and includes ferric recycling for reuse as a coagulant in the plant, other treatment options to reduce the amount of saline water in the sludge, and washing of the sludge to reduce its salt content [61]. Although some of the considered reuse options are technically feasible, clear market opportunities have not yet been identified considering the costs and logistics involved in converting the waste into a secondary product [154].

3.3.4 Antiscalants

Scales may be formed when the solubility limits of certain salts in the concentrate are exceeded. Scaling depends on the ion composition of the source water, the recovery rate and the temperature of the desalination process, and the precipitation kinetics of the sparingly soluble salts. Kinetics may result in long induction times of some salts even in supersaturated solution. Calcium carbonate is the main scale forming species in SWRO systems, whereas solubility limits for sulfate scales and silicates are generally not exceeded due to the high ionic strength of seawater [22]. Calcium carbonate is also the main scale forming species in distillation plants, which additionally encounter sulfate and magnesium hydroxide scaling due to increased temperature [236]. Scale formation can be controlled either by addition of sulfuric or hydrochloric acid (against carbonate scales), the dosing of a special scale inhibitor (antiscalant), or a combination thereof.

Acids react stoichiometrically with calcium carbonate and must therefore be added in relatively high concentrations of 20-50 mg/l to the feed stream. The pH of the acidified feed is usually between 6.8-7.0, compared to a natural seawater pH of about 7.8-8.1 depending on alkalinity and salinity. The concentrate pH is typically between 7.1-7.2, which is 0.1-0.4 units higher than the pH of the acidified feed, but 0.6-1.0 units lower than the ambient pH. [237][h]. A pH effect on the receiving water, which might be attributed to the reduced concentrate pH, is unlikely if the outfall is located in an area that provides good mixing conditions. Due to the good buffering capacity of seawater, any residual acidity will be neutralized quickly by mixing with surrounding seawater [15].

Antiscalants retard the nucleation process and impair crystal growth of scales in low, non-stoichiometric doses of 1-2 mg/l. The main generic groups are polyphosphates, phosphonates (organophosphorus compounds with stable carbon to phosphorus bonds), and organic polymers with multiple carboxylic groups [15, Figure 12].

Polyphosphate antiscalants are easily hydrolyzed to orthophosphate, which is an essential nutrient for primary producers. The use of polyphosphates may cause a nutrient surplus and an increase in primary production in the discharge site, which may lead to oxygen depletion when the plants die and their biomass decays. These effects of eutrophication were observed at the outlets of some larger distillation plants that used polyphosphate antiscalants in the 1990s [10, 13]. However, polyphosphates are only used on a limited scale in distillation plants today due to their instability at higher temperatures, but they might still be in use in some SWRO plants [15, Tables 3 and 5].

In contrast, the phosphonates and organic polymers are rather resistant to biological, chemical and physical degradation and generally have a slow to moderate rate of elimination from the environment through abiotic and biotic degradation processes. Most antiscalants are classified as ‘inherently biodegradable’, with half-lives of about one month or longer. They are not harmful to invertebrate and fish species as the dosing levels are considerably lower than the concentrations at which acutely toxic effects can be observed. However, some material data sheets of commercial antiscalant products classify these as ‘harmful to algae’. The effects are not completely understood, but it is assumed that the observed inhibition of algae growth is likely due to the products’ complexing abilities.

[h] The permeate pH is typically between 6.2-6.5, because HCO_3^- is rejected by the membrane but CO_2 is not, so that the permeate decreases in pH while the concentrate increases in pH relative to the acidified feed.

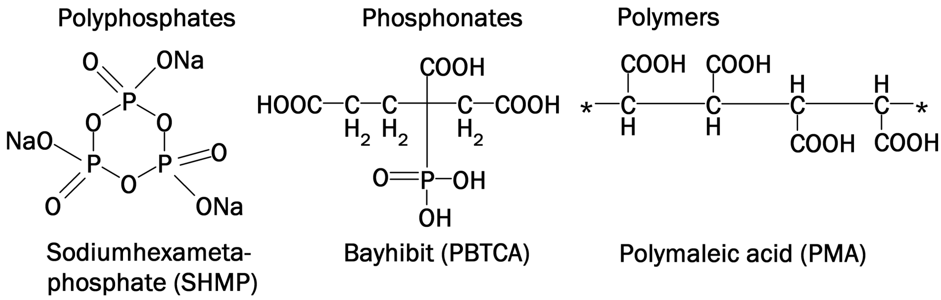

Figure 12: Chemical structures of a polyphosphate antiscalant (left), a phosphonate antiscalant (middle), and an organic polymer antiscalant [adapted from 15].

Antiscalants prevent scale formation by dispersing and complexing divalent cations, such as calcium and magnesium, which are also needed for algae growth [17].

It therefore also seems plausible that antiscalants may interfere with the natural processes of dissolved metals in seawater following discharge. In combination with a slow removal from the environment, this could be of concern in areas of high desalination activity, such as the Arabian Gulf (cf. The Gulf, page 13 and page 82).

3.3.5 Metals

Increases in metal concentrations in the discharge may result from the concentrating effect of the desalination process, which also increases natural metal ion concentrations in the concentrate (e.g., by a factor of two at 50% recovery), and from corrosion processes inside the plant. The first effect is more likely to be observed in SWRO plants due to their higher recovery, while the latter is more likely to occur in distillation plants due to the choice of alloys in the process. The discharge should be in compliance with any effluent limitations and water quality standards that have been established for the project site.

In SWRO plants, contamination with metals is generally below a critical level due to the use of non-metallic equipment and corrosion-resistant stainless steels. The concentrate may for instance contain traces of iron, nickel, chromium and molybdenum, depending on the type of steel used (e.g., 254SMO super austenitic steel). Copper-nickel alloys are common heat exchanger materials in distillation plants. Corrosion of these materials typically causes the contamination of reject streams with copper and nickel. Elevated copper concentrations in reject streams of 15-100 µg/l have been reported [15].

The presence of copper does not necessarily mean that it will adversely affect the environment. Natural copper levels range from an oceanic background of 0.1 µg/l up to 100 µg/l in estuaries [238], which makes it difficult to distinguish between natural copper levels and anthropogenic effects, e.g., as caused by oil pollution. The discharge levels of thermal plants, however, are within a range that could affect natural copper levels.

The U.S. EPA recommends a maximum copper concentration of 4.8 µg/l in seawater for brief exposure and 3.1 µg/l for long-term exposure [239]. Values of the same order of magnitude, i.e., a PNEC of 5.6 µg/l [240], and a water quality objective of 8 µg/l for the Mediterranean Sea [241], were determined for European saltwater environments. As outlined in the section on chlorine use (page 67), ambient water quality standards (AS) should be combined with effluent standards (ES) and clear mixing zone regulations in a combined approach. The ES/AS ratio is 5-32:1 for copper, based on a discharge concentration of 15-100 µg/l and a water quality objective of 3.1 µg/l (U.S. EPA). The ES/AS ratio expresses the necessary dilution that must be attained through physical mixing in the case of copper, as biological and chemical transformation processes are limited.

Copper, like most metals, is transported and accumulated in sediments, which is a major concern for point discharges, as this could lead to increased sediment concentrations in the discharge sites. This stresses the importance of estimating and evaluating total loads, in addition to concentrations (cf. The Gulf, page 82). Metals in sediments can be assimilated by benthic organisms, which often form the basis of the marine food chain, which may potentially lead to bioaccumulation and biomagnification.

3.3.6 Cleaning chemicals

Although much effort and care is invested into the design of pretreatment systems, RO membranes often develop biofilms and accumulate suspended material and scale deposits during operation. The initial stages of biofouling and scale formation can be detected by monitoring salt passage, permeate flux and membrane pressure, and cleaning is periodically needed to avoid irreversible membrane damage. Fouling inside distillation plants can reduce heat transfer, increase corrosion and cause material failure. Cleaning intervals are established on a case by case basis depending on the ambient seawater conditions and efficiency of the pretreatment scheme of the plant. Cleaning is typically carried out in 1-2 year intervals in SWRO plants operating on beachwell water [22] and more frequently, up to several times per year, in SWRO plants operating on surface seawater.

A chemical cleaning is often performed in two stages, first with an acidic solution and then with an alkaline solution [242]. The acidic solution (pH 2-3) is effective against metal oxides and scales, while the alkaline solutions (pH 11-12) removes silt deposits, organics and biofilms. The solutions may additionally contain detergents like dodecylsulfates, oxidants like sodium perborate, and organic-based or inorganic chelating agents such as EDTA or tripolyphosphates (Figure 13). The cleaning solutions are usually generic types or special brands recommended by the membrane manufacturers. After cleaning, or prior to storage, membranes are typically disinfected, using non-oxidizing biocides. For membrane storage over longer periods of time (e.g., during transport or plant shut-down), a chemical preservation solution may be required [15].

After the cleaning process is complete and the cleaning agents have been circulated through the membranes, the membranes are rinsed with product water several times. In many cases, the residual membrane cleaning solution and also the first rinse which contains most of the constituents from cleaning are neutralized and diverted to the sewer for processing. The ensuing rinses are typically disposed with the brine [39].

Discharge into the sewer may not be the standard practice in all locations. It is possible that the cleaning wastes are either discharged by direct blow-down into the sea immediately after cleaning, or the alkaline and acidic solution could be stirred into a buffer tank in order to achieve neutralization before conveying the mixture at a slow rate into the concentrate that is discharged into the sea [242]. For example, discharge practices on the Canary Islands involve neutralization before discharge into the sea. The chemical additives, mainly sodium polyphosphate and EDTA, have been reported as being not considered very important in terms of marine environmental impacts in this case [243].

The cleaning of MSF and MED plants is comparatively simple and usually involves acid washing at a pH of 2. Special inhibitors may be added to control corrosion in this highly acidic environment [15]. The cleaning solution is typically discharged into the sea, with or without pH neutralization before discharge. In MSF plants, the use of sponge balls, which are continuously circulated through the system to mechanically remove scale deposits from the interior surfaces of the heat exchanger tubes, has proven very effective.

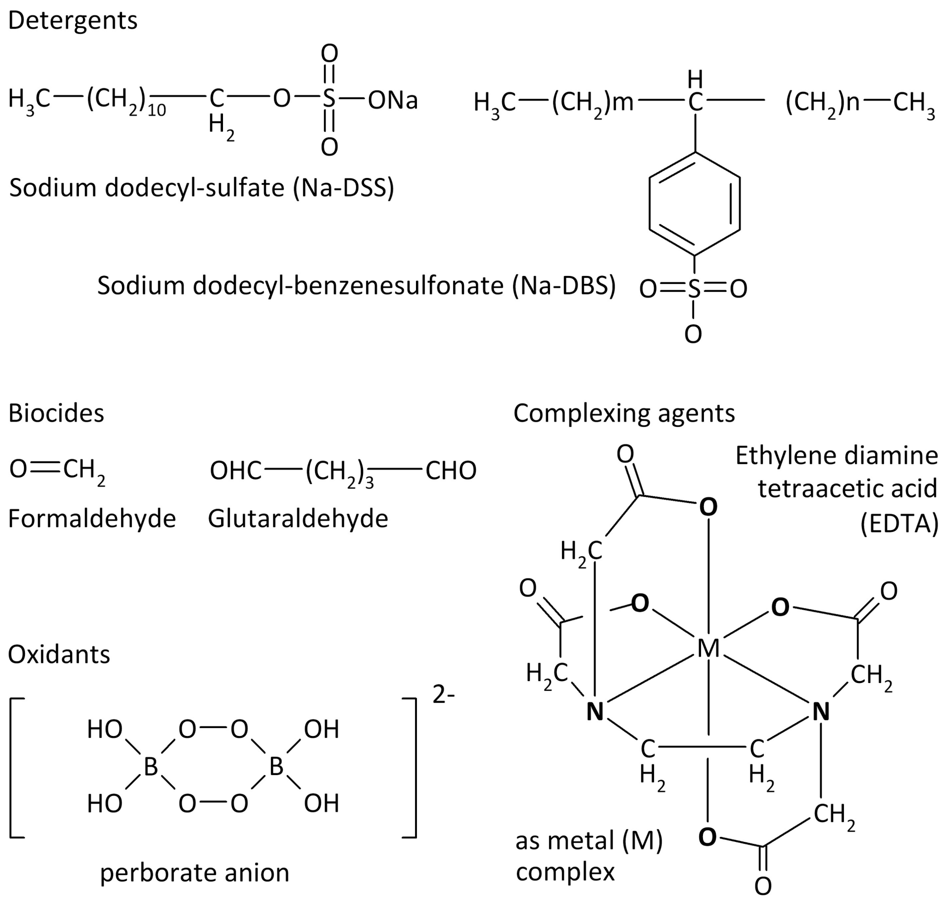

Figure 13: Chemical structures of different cleaning chemicals [adapted from 15].

The untreated discharge of cleaning solutions to surface waters may be harmful to marine life in the mixing zone. The German EPA, for instance, maintains three water hazard classes [244], which provide a good indication of the risks involved in the use of the different cleaning chemicals. Acids (depending on pH), perborate and phosphates are classified as having a low hazard to water, while EDTA and the detergents used for membrane cleaning are classified as 'hazardous'. EDTA is poorly degradable in the environment, and detergents like dodecylbenzene sulfonate have the potential to disturb the intracellular membrane system of organisms due to their surface active properties.

Some of the SWRO cleaning chemicals (e.g., EDTA, phosphates, detergents) are also commonly used in commercial household products. The recommended disposal for greywater generated from domestic processes is discharge into a municipal wastewater treatment plant. It is therefore highly recommended to recover and treat all cleaning solutions prior to discharge. This requires neutralization of the acidic solutions and specific treatment for detergents, oxidants, complexing agents, biocides or other compounds with detrimental effects on marine life and the coastal water body [15].

Table 13: Effluent properties of SWRO, MSF and MED plants with conventional process design: Physical parameters [16, 220].

	Seawater reverse osmosis (SWRO)	**Multi-stage flash (MSF)**	**Multi-effect distillation (MED)**
Physical parameters	SWRO: no use of cooling water in the process, but RO plants may receive their intake water from cooling water discharges of power plants	MSF/MED: it is assumed that the two waste streams from the desalination process are typically combined, i.e., the brine is diluted with major amounts of cooling water from the desalination process. Further dilution with cooling water from power plants may occur but is not considered here.	
Salinity (S) depending on ambient salinity and recovery	▪ SWRO: 65–85 ▪ BWRO: 1–25	▪ cooling water S: ambient ▪ brine S: typically 60–70 ▪ combined S: typically 45–50	▪ cooling water S: ambient ▪ brine S: typically 60–70 ▪ combined S: typically 50–60
Temperature (T) depending on ambient temperature	▪ if subsurface intakes are used: may be below ambient T due to a lower T of the source water ▪ if open intakes are used: close to ambient ▪ if power plant cooling waters are used as source: above ambient	▪ brine T: 3–5°C above ambient ▪ cooling water T: 8–12°C above ambient ▪ combined T: 5–10°C above ambient	▪ brine T: 5–25°C above ambient ▪ cooling water T: 8–12°C above ambient, up to 20°C possible ▪ combined T: 10–20°C above ambient
Plume density (ρ)	▪ lower than ambient (negatively buoyant plume)	▪ MSF/MED: plume can be positively, neutrally or negatively buoyant depending on the process design and mixing with cooling water before discharge, typically positively buoyant due to large cooling water flows	
Dissolved Oxygen (DO)	▪ if subsurface intakes are used: may be below ambient DO due to a lower DO of the source water ▪ if open intakes are used and if chlorine reducing agent is not overdosed: close to ambient	▪ brine: below ambient due to deaeration and oxygen scavengers ▪ cooling water: close to ambient (minor decreases of DO possible because of increases in temperature, as oxygen is less soluble in warmer water) ▪ combined: mixing of brine with cooling water increases the DO content of the combined effluent close to ambient, as turbulent mixing allows oxygen take-up from air	

Table 14: Effluent properties of SWRO, MSF and MED plants with conventional process design: Chemical parameters [16, 220].

Biofouling	**Seawater reverse osmosis (SWRO)**	**Multi-stage flash (MSF)**	**Multi-effect distillation (MED)**
Oxidants (1–2 mg/l) ▪ mainly chlorine ▪ chlorine dioxide used in some plants	▪ usually low-level chlorination with a dosage of 1–2 mg/l to the feed water, added intermittently or continuously ▪ SWRO: oxidants typically removed with sodium bisulfite (2–4 times the dosage of the oxidizing agent)	▪ MSF/MED: chlorine typically not removed by a dechlorination step ▪ discharge concentration of residual chlorine often reduced to 10–25% of the dosage level due to the oxidant demand of the seawater ▪ both the brine and the cooling water contain residual chlorine	
Halogenated organic by-products, typically trihalomethanes (THMs)	▪ SWRO: by-products may form during chlorination but levels are probably low due to dechlorination ▪ all processes: use of chlorine dioxide has been reported to reduce the risk of by-product formation	▪ MSF/MED: chlorination of seawater results in varying composition and concentrations of halogenated (chlorinated and brominated) organic by-products, mainly THMs such as bromoform	
Removal of turbidity (if solids are discharged back into surface waters, i.e., if the plant uses coagulation and has no sludge treatment step)			
Coagulants (1–30 mg/l) ▪ $FeCl_3$ or $FeSO_4$ Coagulant aid (<1–5 mg/l) ▪ e.g. polyacrylamide	▪ if filter backwash is discharged to surface waters: may cause turbidity, iron salts may cause effluent coloration ("red brines"), and increased sedimentation rates	MSF/MED: treatment not applied/not necessary	
Scale control additives (used in all desalination processes, can be a blend of several different antiscalants in combination with acid)			
Polymers (1–2 mg/l) ▪ polymaleic or polyacrylic acids, phosphonates	▪ dosage/discharge concentration below toxic levels to invertebrate and fish species; ▪ some products are classified as being 'harmful' to algae, presumably due to a nutrient inhibition effect, but dosing levels (1–2 mg/l) are still an order of magnitude below harmful levels (20 mg/l) ▪ some products classified as 'inherently' biodegradable: increased residence times in surface waters	▪ MSF/MED: antiscalants only present in the brine, not in the cooling water	
Phosphates (2 mg/l)	▪ SWRO: still used on a limited scale ▪ may cause eutrophication near outlets, as phosphates are easily hydrolyzed to orthophosphate, a major nutrient	▪ MSF/MED: as phosphates are not stable at high T, polymers are preferred	
Acid (H_2SO_4) ▪ dosage: 20–100 mg/l	▪ still applied in SWRO plants ▪ lowers pH from 8.1 (ambient) to 6–7 ▪ acidity quickly neutralized by seawater alkalinity		

Table 15: Effluent properties of SWRO, MSF and MED plants with conventional process design: Chemical parameters (cont.) [16, 220].

Foam control additives	**Seawater reverse osmosis (SWRO)**	**Multi-stage flash (MSF)**	**Multi-effect distillation (MED)**
Antifoaming agents (e.g. polyglycol)	▪ treatment not applied	▪ typically low dosage (0.1 mg/l), which is below harmful levels ▪ used in all distillation processes, but primarily in MSF ▪ antifoam only present in the brine, but not in the cooling water	
Corrosion			
Heavy metals	▪ usually corrosion-resistant stainless steels and plastics used ▪ concentrate may contain low levels of iron, nickel, chromium and/or molybdenum if low-quality steel is used	▪ metallic equipment usually made from carbon or stainless steel and copper nickel alloys ▪ concentrate may contain iron and copper; increased copper levels may be of concern but the available data is limited	▪ metallic equipment usually made from carbon and stainless steel, aluminium and aluminum brass, titanium, or copper nickel ▪ lower corrosion rates than in MSF reported, but no actual data on brine contamination levels available
Corrosion prevention	▪ not necessary	▪ as the feed water is deaerated, the brine is also deaerated before mixing with cooling water, which is not deaerated ▪ in MSF plants, the feed water may be treated with oxygen scavengers (e.g., sodium bisulfite), which may also remove residual chlorine as a side effect	
Cleaning solutions (if discharged into surface waters)			
Cleaning chemicals (used intermittently)	Alkaline (pH 11–12) or acidic (pH 2–3) solutions containing cleaning additives, e.g.: ▪ detergents (e.g., dodecylsulfate) ▪ complexing agents (e.g., EDTA) ▪ oxidants (e.g., sodium perborate) ▪ biocides (e.g., formaldehyde)	▪ Acidic (low pH) washing solution which may contain corrosion inhibitors such as benzotriazole derivates	

3.4 Potential cumulative impacts on sea regions

3.4.1 The Gulf

Due to their waste discharges, desalination plants were included in the list of major sources of land-based marine pollution in the Gulf by the United Nations Environment Programme (UNEP) and the Regional Organization for the Protection of the Marine Environment (ROPME) [2, 245]. The plants are mainly large distillation plants located on the shallow southern part of the Gulf. It can be estimated that the combined discharge of all desalination plants in the Gulf amounts to a waste water flow of more than 1,100 m^3/s, not taking cooling waters of co-located power plants into account. For comparison, the average discharge of the Shatt Al-Arab river, which constitutes the border between Iraq and Iran, is 1,456 m^3/s [246]. Although likely overestimated due to dams and water diversions, the Shatt Al-Arab is still the major source of freshwater influx into the Northern Gulf and an influencing factor for water mass characteristics in the sea region [247].

The waste water of distillation plants is mainly characterized by increased salinity, temperature and residual additives, including chlorine (cf. section 3.3.2, page 64), antiscalants (cf. section 3.3.4, page 75), corrosion products such as copper (cf. section 3.3.5, page 76) and intermittent cleaning solutions (cf. section 3.3.6, page 77). When considering the potential impacts of the waste discharges from desalination plants onto the marine environment, one has to distinguish between the salt and the chemical additives.

Although elevated salinity levels can be harmful to sensitive species (section 3.3.1, page 59), the total salt load is probably not a concern for whole sea areas. For example, desalination plants account for an annual water loss of 0.05% of the water in the Gulf, compared to 5.7% caused by natural evaporation, i.e., natural evaporation is a factor of 100 higher than water abstraction through desalination plants[i]. With an estimated exchange rate of seawater in the Gulf every 3 to 5 years [249], a further increase of the already naturally high salinity in the Gulf does not seem to be a likely scenario.

The daily chemical discharges of desalination plants into the Gulf can be estimated to amount to 23.7 metric tons (t) of chlorine, 64.9 t of antiscalants and 296 kg of copper (Figure 14). These values are estimates based on typical process designs[j]. Little is known about the *actual* chemical loads, their environmental fate and potential impacts on the marine environment. Only in a few cases, marine pollution by desalination plants was monitored and reported. The studies were usually short-term and localized.

For example, chlorine levels and chlorination by-products such as trihalomethanes, chlorophenols and chlorobenzenes (section 3.3.2, page 67) are measurable in the discharge sites of desalination plants. In other cases where pollution has been reported, links to desalination plants are more difficult to establish. For example, chlorine pollution has been reported to affect two mud flat areas in the Bay of Kuwait [197], which can possibly be attributed to the nearby Doha power-desalination plant. Screening of contaminants in marine sediments and biota has furthermore revealed low levels of halogenated pesticides in some locations of the Gulf [245]. The pesticides may stem from agricultural runoff and may have become halogenated as a result of seawater chlorination, either in power-desalination plants or due to other industrial discharges.

[i] Assuming a surface area in the Gulf of 250,000 km^2, an average water depth of 35 m, an evaporation loss of 2,000 mm per year [248] and an installed desalination capacity of 12.1 Mm^3/d (cf. section 1.4.1).

[j] Assuming a chlorine concentration of 250 µg/l in the brine and cooling water discharges of MSF and MED plants, copper levels of 15 µg/l in the brine of MSF plants, and a dosing rate of 2 mg/l of antiscalants to the feed water of MSF plants operated at 10% recovery, and a dosing rate of 2 mg/l of antiscalants to the feed water of SWRO plants operated at a limited (33%) recovery due to the high salinity in the Gulf.

Monitoring data concerning copper contamination in water, sediment and organisms attributed to desalination activity in the Gulf is also hardly available, although the latest state of the marine environment report mentions that copper and nickel levels are "relatively high near the outfalls of desalination and power plants" [245]. Eight heavy metals have recently been measured in the subtidal sediments near two power-desalination plants in the Gulf of Oman, outside the Gulf. Both plants have capacities of about 90,000 m^3/d and are thus comparatively 'small' distillation plants. Slightly elevated levels of copper, and to a lesser extent zinc, were found at varying distances with some high values close to the discharges, and are assumed to be corrosion products. Copper levels were slightly higher at the older of the two plants, which has been operating for more than 30 years. Maximum copper levels found in sediments were 13-16 mg/kg, which is below action trigger values established by UK, US and Australian authorities (40 mg/kg, 136 mg/kg, 65 mg/kg, respectively), and could hence be considered "slightly contaminated" [250].

Nothing is known about the environmental fate and effects of the antiscalant discharges to date. It has to be concluded that there is still a paucity of useful environmental monitoring data from the Gulf. Long-term or transboundary field investigations into the cumulative impacts of desalination plants on the Gulf's ecosystem are not existing [198].

Although this thesis is limited to the environmental issues in seawater desalination, it should be mentioned that desalination is only one cause of environmental concern in the Gulf, and maybe not the most pressing one. Oil is still the primary polluter in the sea region and considered as the greatest threat to the marine ecosystems [2, 251]. The level of petroleum hydrocarbons in the area exceeds that in the North Sea by almost three times [245]. Another serious threat to the integrity of coastal ecosystems is land reclamation in conjunction with the disposal of solid wastes and dredging, which causes the large-scale loss and degradation of coastal habitats. In some countries, a significant proportion of the shoreline is artificial, e.g., in Kuwait or Bahrain [2]. In Dubai, the most recent and prominent land reclamation projects is the Jumeirah Palm, which increased Dubai's shoreline by 100%, to be followed by Palm Jebel Ali and Palm Deira, and the World, which consists of 300 smaller artificial islands created off the coast of Dubai.

Other priority issues include nutrients, litter, persistent organic pollutants, heavy metals and radioactive substances, attributed to agricultural runoff, atmospheric deposition, sewage and industrial outfalls [2]. 20-30% of the sewage is assumed to be untreated or only partially treated [245]. The major industries in the Gulf include oil refineries, petrochemicals and chemicals, fertilizers, minerals, metals, cement, textiles and food processing, shipping and port operations, and power and desalination plants [2].

Some of these industries, such as for example oil and gas refining, use large quantities of seawater as a cooling medium with chlorine doses of 0.5-1.5 mg/l [203]. This is similar to the dosing levels in power-desalination plants, in which the major share of the cooling water discharges (60%) is attributed to power generation, and only the minor share (40%) to desalination [252]. An inventory of power plant capacities or other industries with cooling water requirements, similar to the IDA Worldwide Desalting Plant Inventory [24], is not available for the Gulf. It can be concluded that desalination accounts for only a part of the industrial cooling discharges into the Gulf and their thermal and chemical loads. Moreover, when comparing the mass and nature of heavy metals released by the discharges of desalination plants with the amount and nature of heavy metals probably released by other industries, atmospheric fallout and crude oil spills, desalination can be assumed to be only a minor contributor to the heavy metal load in the Gulf [166].

Considering that the Gulf is a very shallow, semi-enclosed, sedimentary sea basin with an average depth of only 35 m and a narrow opening of only 56 km to the open ocean, the question is why it has been able to withstand the manifold pressures so far. This may be due to the good mixing of the water column caused by wind and tidal action. The general circulation in the Gulf is driven by density gradients and is characterized by a cyclonic, counter clockwise pattern. Bottom water of high salinity flows along the southern shoreline out of the Gulf and is compensated by oceanic surface water through the Strait of Hormuz which flows along the Iranian coast to the north. With an anticipated turnover time of about 3 to 5 years, waterborne pollutants will eventually be flushed out of the Gulf. For other substances, such as metals, which do not degrade and tend to be transported into the sediments, the Gulf may act as a sink, with the risk of long-term accumulation of these substances [hydrological data from 248].

It is upon the ROPME Member States to further coordinate their efforts in a unified approach to protect the marine environment. Such an approach should include the establishment of a monitoring framework for the whole Gulf, and particularly ensure the enforcement of national, regional and international laws and conventions. Qatar, for instance, has one of the strictest discharge standards for chlorine in the world, but the question is – can it be implemented? As most industries struggle to meet the new requirements, they make an effort to facilitate the Ministry of Environment in revising the regulations instead [194]. If it is not possible to achieve a low chlorine discharge by low-level or pulse-chlorination, maybe dechlorination should be considered as an alternative. Chlorination-dechlorination has been proposed as early as 1992 [209], but has not been implemented in a single power-desalination plant to date. Also, control efforts have been below the desirable standards in many cases and have always been outstripped by further developments. Some reasons for this shortfall are the inadequate application and enforcement of measures and insufficient emphasis on preventative means [2]. Moreover, the lack of information exchange within ROPME states on a regular basis has been a major factor impeding the implementation of many environmental programmes [2]. ROPME is currently working with UNEP on an update of the state of the marine environment report [253], which will hopefully include a current and comprehensive inventory of the main pollutant sources of the Gulf at a transnational level.

3.4.2 The Red Sea

Similar to the Gulf, the Red Sea is a semi-enclosed body of water, in which desalination plants are considered a major source of land-based pollution [3].

The exchange of water is limited by the narrow strait of Bab el Mandeb, which is only 29 km wide and 130 m deep. However, the Red Sea itself is considerably larger and deeper than the Gulf with an average depth of 490 m. A net inflow of water into the Red Sea occurs through the Bab el Mandeb, which is partially balanced by an outward bottom current. These flows correspond to moderate turnover times of six years for the surface layer and slow turnover times of 200 years for the whole water body [254]. A pycnocline separates the surface and deep water layers at a depth of about 250-300 m. The Red Sea is known for its outstanding and fragile biological habitats. The southern parts, which are more strongly influenced by oceanic water, are dominated by fine grained sediments and associated mangroves, seaweeds and calcareous algae, while the nutrient-deprived northern parts are characterized by fringing coral reefs [255].

Although the total installed desalination capacity is lower than in the Gulf, the world's largest MSF plant in Shoaiba, about 100 km south of Jeddah, and other large facilities are found on the shore of the Red Sea (cf. The Red Sea, page 13).

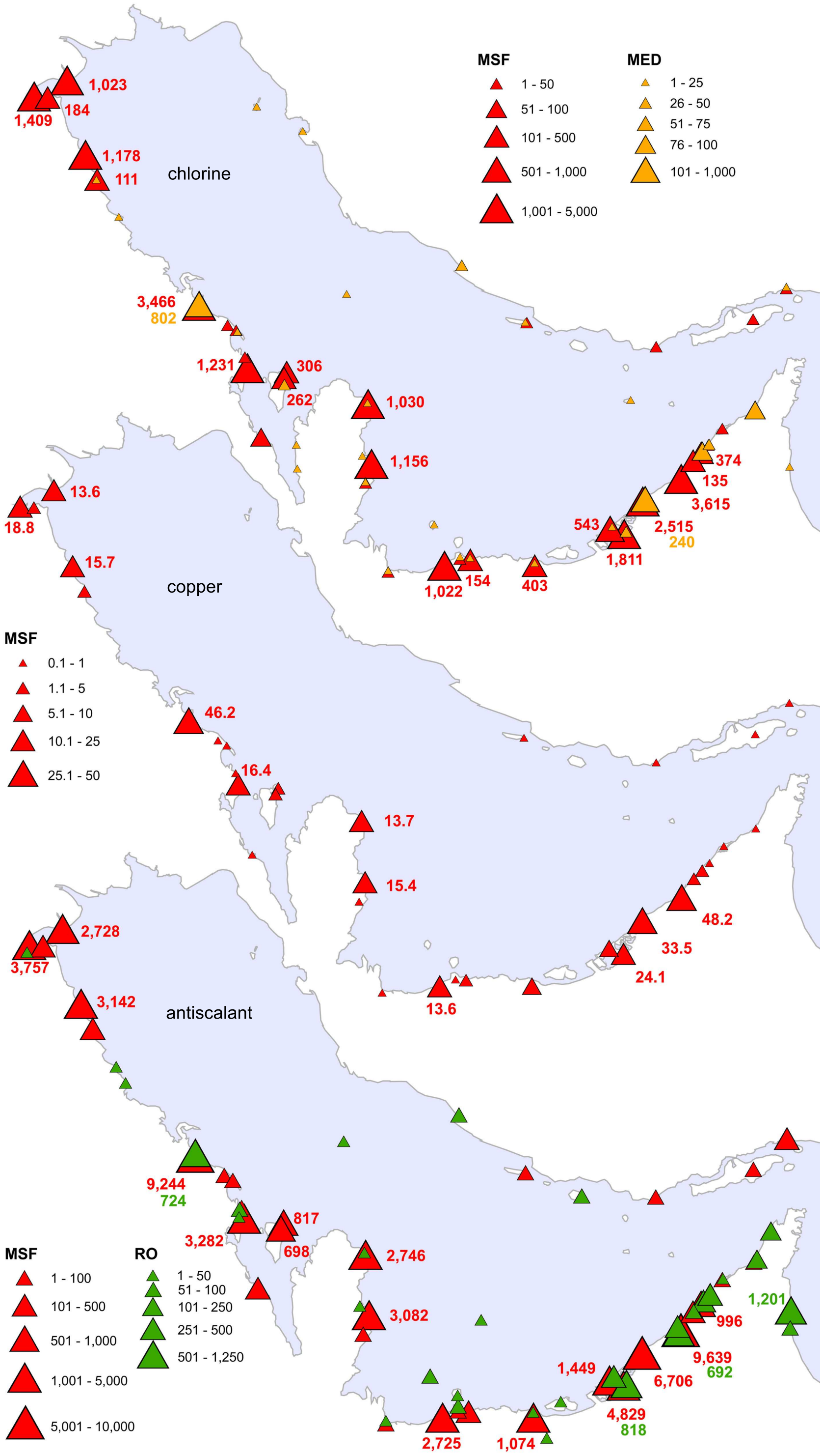

Figure 14: Estimated chemical discharges of chlorine (top), copper (middle) and antiscalants (bottom) into the Gulf in kg/d [7, updated after 198].

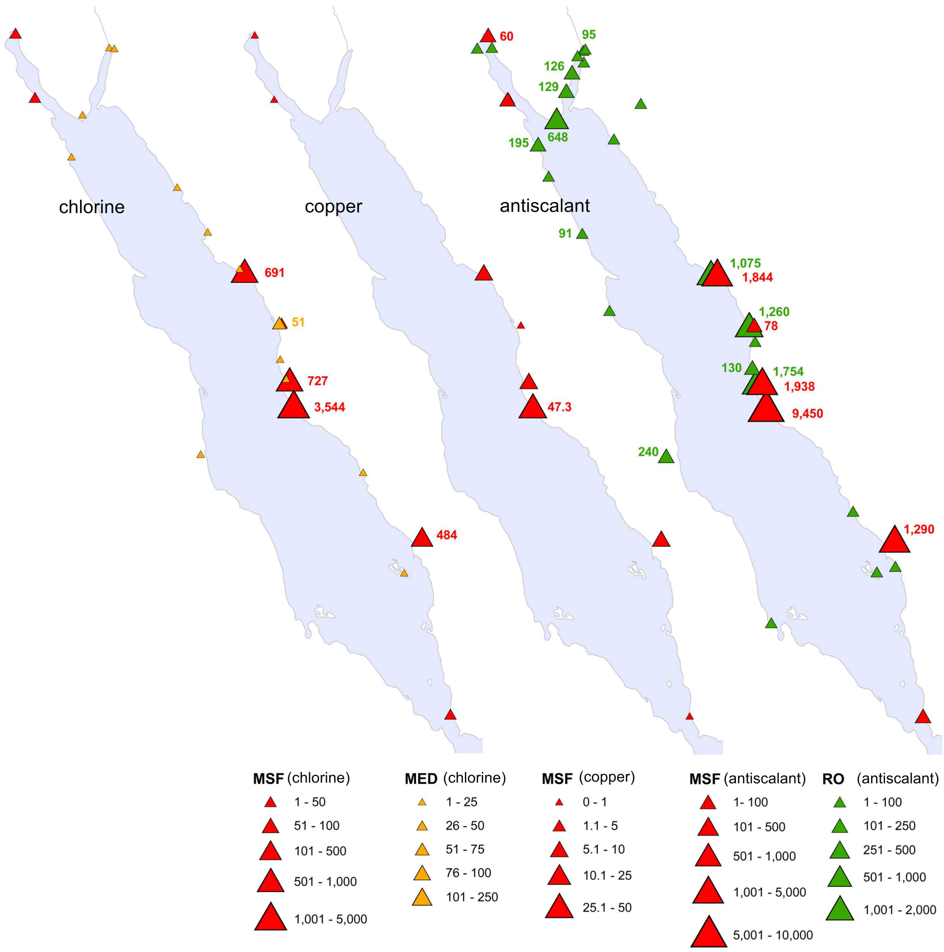

Figure 15: Estimated chemical discharges of chlorine (left), copper (middle) and antiscalants (right) into the Red Sea in kg/d.

The daily discharges of chemical additives from desalination plants into the Red Sea can be estimated to amount to 5.6 t of chlorine, 20.7 t of antiscalants and 74 kg of copper [7, updated after 255, Figure 15][k]. So far, no scientific basis exists that allows a conclusion on the actual impacts of these discharges on the Red Sea's ecosystem. The stratification of the water column, the existence of sills, and the long turnover times of the deep water layer bear the risk that pollutants will either have relatively long residence times or will remain almost indefinitely in the Red Sea. In combination with fragile and ecologically important ecosystems, it is probable that the Red Sea is very susceptible to disturbances by harmful materials. The rapid expansion of urban centers in Saudi Arabia has been achieved through the extensive use of desalinated water to meet demands of the population and industry [3]. A similar development will probably take place in the Gulf of Aqaba. The transition from the Gulf of Aqaba to the Red Sea, separated by the 250 m deep Strait of Tiran, is a smaller version of the transition from the Red Sea to the ocean.

[k] Based on the same assumptions as for the Gulf region.

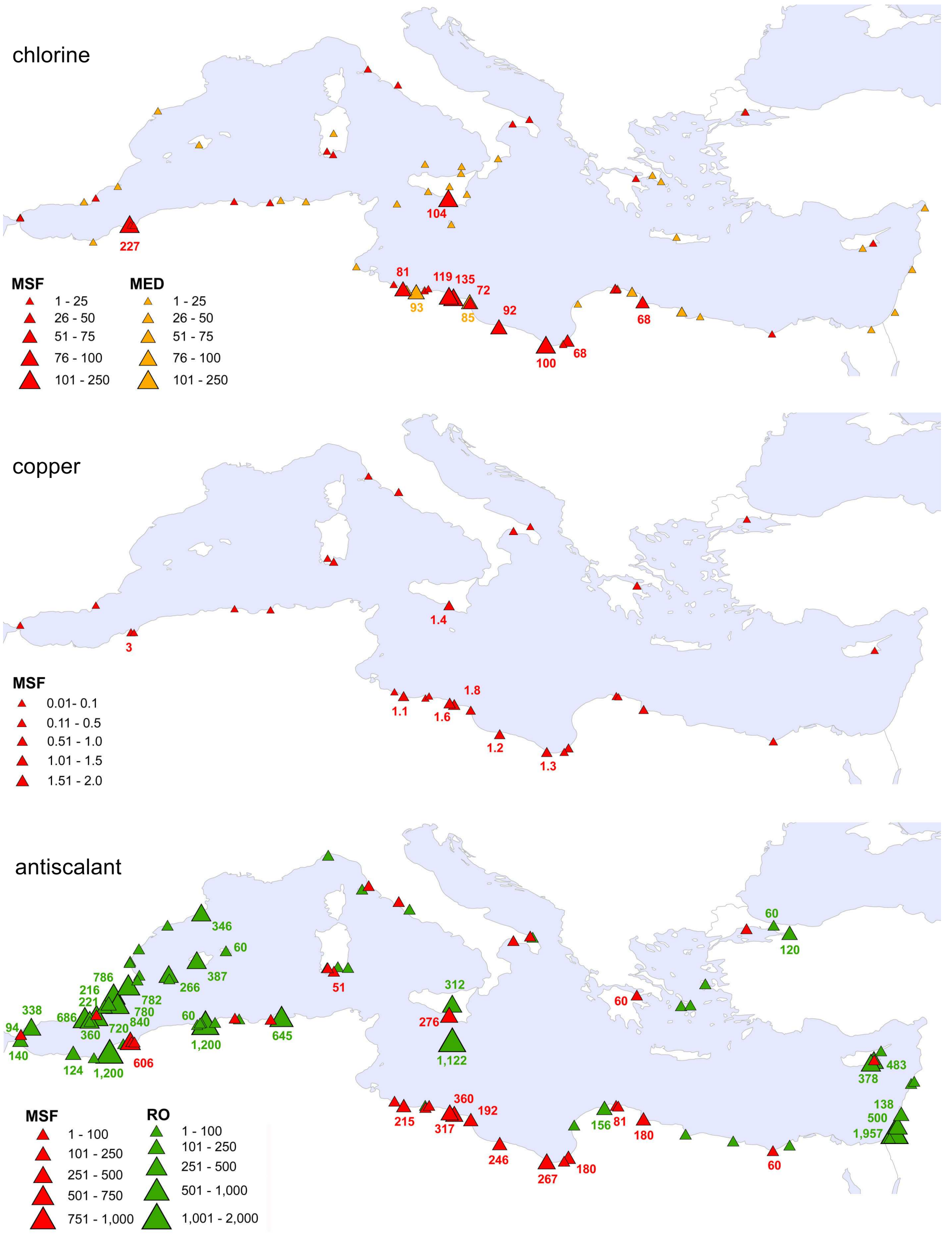

Figure 16: Estimated chemical discharges of chlorine (top), copper (middle) and antiscalants (bottom) into the Mediterranean Sea in kg/d.

The double semi-enclosure nature of the Gulf of Aqaba makes it one of the places most susceptible to any form of land-based pollution [255].

3.4.3 The Mediterranean Sea

The complex geomorphology of the Mediterranean basin is reflected in a complex surface water circulation, which is characterized by the formation of ring-shaped currents in most regional seas. As tidal currents are generally weak and therefore have little influence on the dispersal and dilution of pollutants, the surface circulation is the primary factor that controls the transport of contaminants within the Mediterranean Sea.

Transport between basins and out of the Mediterranean is limited by narrow, shallow straits. Vertically, the water column is not well-mixed due to the high average depth of the Mediterranean Sea of almost 1,500 m. Three different water masses can be distinguished within this stratification: The surface water consists of inflowing Atlantic water in the west, which moves in a counter clockwise direction along the Algerian coast to the east. In the eastern basin, high evaporation rates cause an increase in salinity. In combination with winter cooling, the surface water increases in density and sinks, forming the Levantine Intermediate Water (LIW).

The LIW flows in a depth of 200-500 m towards the west and enters the Adriatic and Balearic Seas, where strong cooling events cause a further increase in density and lead to the formation of East and West Mediterranean Deep Water (below 600 m depth), respectively. Mixing of the two deep waters is largely restricted by the 400 m deep sill that forms the Strait of Sicily. Above this deep layer, the LIW circulates through both basins and eventually exits the Mediterranean through Gibraltar.

The circulation takes place slowly and the turnover time from entry as Atlantic surface water until its return to the ocean is about 80 years [256, 257]. Some deep water bodies may be much older, in the order of 100-300 years, whereas some of the deep water may exit the Mediterranean again after only a few decades. Marine pollution problems in the two basins are therefore to a large extent independent from each other and the long turnover times allow for a rapid accumulation of substances.

A recent report of the European Environment Agency and UNEP identified priority issues in the Mediterranean environment, including land-based pollution by sewage, urban run-off and industrial effluents [258]. The report does not list or discuss desalination activity in any form, however, an earlier report published by the Mediterranean Action Plan addressed sea water desalination in the Mediterranean specifically, including environmental concerns [29]. The document emphasizes that seawater desalination is an industrial process and a growing industry in the Mediterranean, which may have adverse effects on the coastal environment if not well designed and managed.

Hot spots of desalination activity in the Mediterranean Sea are primarily along the coasts of Spain, Algeria, Libya and Israel, and on some larger islands including Mallorca, Malta, Sicily and Cyprus. The main process is SWRO (cf. section 1.4.3, page 17). The concentrate from SWRO plants is primarily characterized by high salinity, and typically contains antiscalants from pretreatment. The daily discharge from desalination plants into the Mediterranean Sea may amount to 23 t of antiscalants, plus 1.9 t of chlorine, and 18 kg of copper from MSF plants[1]. Side streams, such as backwash waters from media filters are discharged into the sea in some of the riparian countries, and the fate of cleaning solutions could be similar [7, updated after 29, Figure 16].

[1] Based on the same assumptions as for the Gulf region.

3.5 Summary and conclusions

Naturally, all enclosed seas have a very limited exchange of water with the open ocean which favors long residence times of pollutants. The Gulf has the world's highest density of desalination plants. If anywhere at all, impacts from desalination activity should be visible in the Gulf. However, a holistic study investigating the cumulative impacts of desalination plants on the Gulf's marine environment is missing. If effects occur, they may not catch the eye of the casual observer for two reasons: First, favorable mixing and flushing may quickly disperse the pollutants, and second, impacts from desalination activities may be overshadowed by other sources of land-based pollution or anthropogenic activity, such as the permanent oil burden or land reclamation.

While no conclusive evidence can be provided concerning the Gulf as a whole, the risk of damage to the ecosystems in close proximity to desalination plants is at hand. It is possible to link environmental effects to desalination activity on some occasions, but even here the scientific data is incomplete. The few available studies are typically short-term, limited in scope, and without ecologic baseline or effects monitoring. They fall short of recognizing the potentially synergetic effects of the single waste components on marine organisms and the complexity of potential responses by the ecosystems.

The Mediterranean and Red Sea have a lower density of desalination plants than the Gulf, although some parts show increased desalination activity. Due to their longer coastlines, greater water depth and lower total desalination capacity, cumulative impacts are less likely and are of secondary importance behind other issues of higher priority, such as sewage and industrial discharges or eutrophication. In the Mediterranean, where SWRO is the dominating process, the problem of chemical discharges is 'reduced' to the antiscalant loads and the possible discharge of untreated backwash sludge or cleaning wastes. Chemical impacts from desalination plants are therefore probably still limited and chemical loads fairly well distributed. Localized impacts, however, may be significant, especially when important ecosystems are affected. For example, salinity increases near the outfall of the Dhekelia SWRO plant on Cyprus were reported to be responsible for a decline of macroalgae forests [171, cf. page 61].

Sensitive ecosystems include the *Posidonia* seagrass meadows in the Mediterranean, which have been classified as a priority habitat by the European Habitats Directive[m], and which have been found to be very sensitive to salinity increases, as well as for instance the coral reefs and mangroves of the Red Sea that are of global importance. The Red Sea is a unique ecological treasure without equal that is vulnerable to ecological damage. With regard to the expected future demand for seawater desalination in both the Mediterranean and Red Sea region, it is important to regulate the development of new desalination plants and to adopt a precautionary approach in the development of new desalination projects. Measures for impact mitigation are discussed in chapter 6.

[m] Directive 92/43/EEC on the conservation of natural habitats and of wild fauna and flora.

Air quality impacts | 4

4.1 Introduction

Air quality impacts of desalination projects are primarily associated with the use of energy during the manufacturing and transportation of materials, the construction of the plant facilities and associated infrastructure – and most importantly – plant operation. The energy use during operation includes the electrical or thermal energy produced on site or taken from external sources, such as the electricity grid. The total energy demand of the facility comprises the energy needed to drive the desalination process, and energy for pumping and pretreatment, for heating and air conditioning, for lighting and office supplies. The *specific* energy demand refers to the energy demand of the desalination process only. The energy demand depends on the choice of the process and pretreatment, as outlined in section 2.3.4, page 46. This chapter puts energy demand of desalination into perspective and discusses the main environmental implications of energy use.

4.2 Energy demand in perspective

SWRO plants require much less energy than distillation plants to produce the same amount of product water if one takes the thermal energy demand of distillation plants into account, but the case can be a close decision when the thermal energy is provided by 'waste' or 'low value' heat (section 2.3.4). Still, the electrical energy demand of the desalination processes is significant: it is typically 3-4 kWh/m^3 for SWRO, 3-5 kWh/m^3 for MSF and 1.5-2.5 kWh/m^3 for MED-TVC distillation (Table 8, page 46).

Parts of this chapter were based on:

S. Lattemann, K. Mancy, H. Khordagui, B. Damitz and G. Leslie. Desalination, resource and guidance manual for environmental impact assessments. United Nations Environment Programme (UNEP), Nairobi, Kenia, 2008.

S. Lattemann, M.D. Kennedy, J.C. Schippers and G. Amy. Seawater reverse osmosis: a sustainable and green solution for water supply in coastal areas? Submitted to Balaban Desalination Publications for a book on seawater desalination to be published in memory of Sydney Loeb.

The electricity demand of a SWRO plant using state of the art technology can be illustrated by a simple arithmetic example. A plant with a capacity of 130,000 m^3/d could supply up to 1 million people with water for domestic use, assuming the German average per capita water consumption of 130 l and not counting any indirect uses (industry, food production etc). At a specific energy demand of 3 kWh/m^3, the plant would increase the electricity demand of every person in the municipality by 10%. In other words, a person consumes on average 0.13 m^3 of water per day, which – if produced by desalination – increases the person's average electricity bill by 10% from 3.8 kWh/d to 4.2 kWh/d[a]. 10% is a considerable increase, however, it could be compensated by reducing the time of a hot shower by 80 seconds every morning[b]. It is estimated that the total saving potential due to the use of energy-efficient household appliances in Germany is about 29% per household [259], without compromising comfort or living standards.

The reference values that we choose, as illustrated by this example, may influence how we perceive and evaluate the significance of energy demand and associated environmental impacts. For desalination, the increase in electricity or energy demand is often calculated on a municipal, regional or national level, for example:

- In the municipality of Carboneras on the Mediterranean coast of Spain, a SWRO plant with a capacity of 120,000 m^3/d accounts for about 33% of the province of Almeria's *electrical* energy demand [260].
- In the two Spanish provinces of the Canary Islands (Las Palmas and Santa Cruz de Tenerife), desalination accounts for 14% of all *energy* demands [261].
- On the Mediterranean coast of Spain, desalination capacity will amount to about 2.7 Mm^3/d in 2010. The electricity demand of 11 GWh/d, assuming 4 kWh/m^3, will cause a 1.4% increase over 2005 national *electricity* generation levels [17, 262].
- The Sydney desalination plant with a capacity of 250,000 m^3/d may result in as much as a 0.6% increase of New South Wales' *electricity* demand and will provide about 15% of the city's water [61, 155].
- The Perth SWRO plant in Western Australia with a capacity of 144,000 m^3/d is responsible for about 0.7% of the peak *electricity* demand in the region in summer time and provides about 17% of the city's water. The value of 0.7% compares to 30% required for air conditioning in summer [70].
- In Kuwait, 10 % of the national *fuel* use is attributed to the production of 443 Mm^3 of desalinated water, which is 90% of the national water supply, and 43 % of the national fuel use is attributed to the generation of 42,257 GWh of electricity per year [252].

As the treatment and distribution of water from conventional sources and by conventional processes also requires energy, the *relative* increase in energy demand should be considered besides the total demand of the process. The electrical energy demand of treating local surface water is typically between 0.2-0.4 kWh/m^3, compared to a specific energy demand of modern SWRO plants of 2.5 kWh/m^3 (Table 8, page 46), resulting in a relative increase of 2.1-2.3 kWh/m^3 for seawater desalination.

[a] Assuming an average electricity of 3.8 kWh per person and day, which is the median of the average per capita electricity consumption of persons living in one, two, three and four person households, which ranges from 3 to 4.9 kWh/m^3 (the average consumption of a person in a single-person household being highest). All figures are German averages in 2006.

[b] Assuming that the water is produced by an electrical continuous-flow water heater with a power output of 18 kW and an efficiency of almost 100%, i.e., it requires 0.3 kWh to take a 1 minute shower.

In locations where the water is furthermore transported over long distances to the consumers, the relative increase of a local desalination plant may be much smaller, or desalination may even be the more energy-efficient solution. In *California*, for instance, water is transferred between water basins by an energy-intensive statewide conveyance system. The total *water*-related energy use in California represented one fifth of the *total* energy use in the state in 2001[c] [263]. If *additional* desalination projects with a capacity of 1.3-1.7 Mm^3/d were to be implemented by 2030 (section 1.4.4) with an average energy use of 3 kWh/m^3, the water-related energy use might further increase by 3-4% over 2001 levels [20, 34]. Taking likely future energy savings in SWRO technology into account, which are estimated to be limited to 15% [5, see also section 2.3.4], the increase will be 2.5-3% assuming an average energy demand of 2.5 kWh/m^3.

Presently, the electricity needed to deliver water to San Diego County is 2.8 kWh/m^3 [264], and 3-3.2 kWh/m^3 if one assumes that the water still has to undergo treatment. San Diego County is the farthest point of delivery in the aqueduct systems [20] and 90% of the county's water supplies are imported from the Colorado River and Sacramento Bay−San Joaquin River via the State Water Project [34]. The City of Carlsbad in San Diego County is planning to switch its entire water supply from imported water to desalinated seawater, with a 200,000 m^3/d SWRO project under development. The plant's electricity demand with present state of the art technology is expected to be 3.6 kWh/m^3 [264], which is 0.8 kWh/m^3 higher than the present electricity demand of imported water, and 0.4-0.6 kWh/m^3 higher if the imported water still needs to be treated. Seawater desalination, in this case, is still the most energy intensive water supply, but the dependency on limited external resources made desalination an attractive option, and it may well become competitive in terms of energy demand in the future.

In the case of *Perth*, Western Australia, the energy demand for diverting water over 1800 km from the Kimberley River System in the North to the city by three pipelines would have an estimated pumping energy demand of 14 kWh/m^3 [70], compared to 'only' 3.8 kWh/m^3 (design value) respectively 3.3-3.5 kWh/m^3 (operational range since start-up) for the Perth SWRO plant [72]. The plant produces 17% of the city's water supply, resulting in a metropolitan bulk water cost of 0.5 kWh/m^3 [70], and equalling about 80 liters of desalinated water per person and day [265][d]. This compares to a per capita water use of 290 l/d for domestic purposes and 420 l/d if one includes all indirect uses [266], but not taking domestic bores into account, which makes water consumption by Perth residents one of the highest in Australia and very high by world standards [267][e].

To conclude, desalination can be a significant energy consumer in some parts of the world, which depend heavily on desalinated water. As seen in the examples above, desalination accounts for 14% of the *energy* demand on the Canary Islands or for 10% of the *energy* demand in Kuwait. On the mainland of Spain, however, desalination accounts for 'only' 1.4% of the national *electricity* use, and this value will even be lower if the electricity use of desalination is compared to the total Spanish *energy* use taking other sectors such as transportation into account. The electricity increase of 1.4% is similar in magnitude to the increases given for the Sydney and Perth plants, but the figures must always be seen in their specific contexts and cannot be generalized.

[c] Water-related energy use of 48,012 GWh in 2001.

[d] Assuming a capacity of 144,000 m^3/d and a population of 1.8 million in the metropolitan area of Perth.

[e] The total Perth scheme water consumption averages 153 m^3/person/year, including households (106 m^3), commerce, agriculture, parks, fire fighting and water treatment (average values 2002-2006) [266]. A high number of the private households in Western Australia [21%, 268] has access to domestic bores which account for water supplies of about 250 m^3/property/year in addition to the scheme water supply [267].

4.3 Emissions into the atmosphere

As both the electrical and thermal energy used for the desalination of seawater is usually produced from fossil energy sources, a main environmental and public health concern of desalination is the release of air pollutants into the atmosphere, primarily greenhouse gas (CO_2), acid rain gases (NO_X, SO_2), and fine particulate matter <10 µm (PM_{10}) and <2.5 µm ($PM_{2.5}$). The emissions can result directly from the process, i.e., when fossil fuels are burnt to provide heat for desalination in cogeneration plants, or indirectly when electricity is produced on site or taken from the grid to be used in the desalination process.

The air pollutant emissions caused by desalination could contribute substantially to other emission sources, conflict with applicable air quality standards, or undermine national and international efforts to curb air pollutant emissions. Relevant policies include for example the Kyoto protocol, which establishes legally binding commitments for the reduction of six climate change gases. For EU member states, the NEC Directive sets binding national emission ceilings for NO_X and SO_2, and the Air Quality Framework Directive defines the policy framework for twelve air pollutants and specifies limit values for NO_X, SO_2 and PM in ambient air.

Climate change gases

Air pollutant emissions generally depend on the fuel type, the technology and the efficiency of the power plant or cogeneration plant as well as the installed exhaust purification equipment. Carbon dioxide emissions can be estimated with a high degree of certainty, as they mainly depend on the carbon content of the fuel. Basic emission factors for carbon dioxide have for instance been established as part of the EU emission trading scheme in order to quantify carbon dioxide emissions from fuel combustion (Table 16).

When electricity is taken from the grid, the energy mix of the respective grid must furthermore be taken into account. The shares of the different energy sources may vary in single countries. For example, 28% of the electricity in the EU-25 is produced by coal, 4% by oil, 21% by gas, 30% by nuclear and 14% by renewable energy sources, which is different from the Spanish energy mix where the shares of oil, gas and renewables are higher but the shares of nuclear energy and coal are lower (Table 17).

Spain has the third largest (8%) seawater desalination capacity on a global scale (cf. section 1.4.3). ENDESA, the leading electric utility company, specifies a CO_2 emission factor of 0.51 kg/kWh for their plants on the Iberian Peninsula. According to the Spanish National Hydrological Plan, SWRO plants consume on average 4 kWh/m^3 in Spain [139]. This results in CO_2 emissions of about 2 kg per m^3 of desalinated water (Table 18). The increase in electricity consumption by 11 GWh/d caused by the Spanish AGUA programme, which targets an estimated production of 2.7 Mm^3 of desalinated water in 2010, will result in additional CO_2 emissions of 5,476 t/d, which represents a 0.6% increase in national CO_2 emissions compared to pre-2005 levels of 326 million tons [269].

It would be more appropriate to estimate the global warming potential, expressed as carbon dioxide equivalents (CO_2-e), by taking *all* relevant climate-change gases into account that arise from the combustion of fossil fuels (namely CO_2, methane, nitrous oxide, perfluorocarbons, hydrofluorocarbons and sulphur hexafluoride as specified in the Kyoto protocol). All climate change gases are expressed as the equivalent amount of CO_2 that would have the same global warming potential as the non-CO_2 emissions. Average emissions factors for CO_2-e of the specific grid and energy mix should be published or available on request from utility companies or national authorities.

Table 16: Carbon dioxide emission factors [270].

Fuel type	Black coal (anthracite)	Brown coal (lignite)	Light fuel oil	Heavy fuel oil	Natural gas	Petrol	Diesel
g CO_2/kWh	338	404	266	281	202	259	266

Table 17: European and Spanish energy mix in 2005 [262].

	Coal	Oil	Gas	Nuclear	Renewables	Other	Total
EU 25 [TWh]	900	136	682	973	440	75	3207
	28%	4%	21%	30%	14%	2%	100%
Spain [TWh]	79	24	80	58	44	9	294
	27%	8%	27%	20%	15%	3%	100%

Table 18: Estimated CO_2 and CO_2-e emissions of selected desalination projects.

	CO_2 [kg/kWh]	CO_2-e [kg/kWh]	electricity [kWh/m^3]	emissions [kg/m^3]	capacity [m^3/d]	emissions [t/d]	
Spain	0.51	no inf.	4.00	2.03	2,700,000	5,476	a
Ashkelon	0.20	no inf.	3.60	0.73	330,000	240	b
Perth		0.98	4.00	3.92			
		0.98	2.30	2.25	144,000	325	c1
		0.98	3.50	3.43	144,000	494	c2
Sydney		1.06	4.00	4.24			
		1.06	3.60	3.82	250,000	954	d1
		1.06	4.00	4.24	250,000	1,060	d2
Queensland		1.16	4.00	4.64			
		1.16	4.10	4.76	125,000	595	e1
		1.16	4.70	5.43	125,000	679	e2
Melbourne		1.31	4.00	5.24			
		1.31	4.20	5.50	434,783	2,392	f1
		n/a	n/a	7.45	434,783	3,239	f2
		n/a	n/a	7.80	434,788	3,391	f3

[a] based on the CO_2 emission factor for the ENDESA plants
[b] gas-fired power plant on-site (without fuel life cycle)
[c1] based on the lowest specific demand / [c2] on all operations
[d1] includes desalination only / [d2] all operations
[e1] includes desalination only / [e2] all operations and water transfer, 345 days of operation per year
[f1] includes desalination only / [f2] all operations, water transfer, life-cycle analysis of operation, 345 days of operation, emissions of 1,117,950 CO_2-e t/a, capacity of 150 GL/a or 434,783 m^3/d
[f3] includes desalination, water transfer, full life-cycle analysis including operation and construction (1,403,140 CO_2-e tons attributed to diesel generators, grid connected power, construction equipment, transportation of workforce and materials, offsite waste decomposition, and embodied emissions of materials for the construction of the power and water grid connections, the desalination plant and the marine structures)

For example, a grid average value of 1.16 kg CO_2-e per kWh was used to calculate the emissions for the Gold Coast desalination project in *Queensland*, Australia, which includes direct emissions of CO_2, methane and nitrous oxide from power generation as well as other factors such as transmission losses [157]. For the Victorian SWRO plant in *Melbourne*, a grid average value of 1.31 kg CO_2-e per kWh applies, which is the highest in the whole of Australia [271] due to a high share of brown coal in the energy mix of Victoria [154] with a relatively high carbon content (Table 16). For *Sydney* and *Perth*, where other large desalination projects are to be located, emission factors for electricity from the grid are 1.06 and 0.98 kg CO_2-e per kWh, respectively [271, Table 18].

For comparison, if one assumes an energy demand of 4 kWh/m^3 for SWRO plants as in the *Spanish* example, CO_2-e emissions would be 4.6 kg/m^3 of desalinated water in the *Queensland* case and 5.2 kg/m^3 in the *Melbourne* case, as compared to 2 kg/m^3 in the Spanish example which refers to CO_2 emissions only. This illustrates that the global warming potential of desalination may be a factor of two higher if one takes other climate change gases in addition to CO_2 into account. It will be even higher if one furthermore includes the distribution of the water, construction activities as well as CO_2 emissions associated with the use of materials and chemicals into the calculation.

The real energy demand of the *Queensland* plant including all operations and pumping within the existing water storage network is estimated to be 24.5 MW for a capacity of 125,000 m^3/d, which equates to 4.7 kWh/m^3 (4.1 kWh/m^3 for desalination alone). The indirect greenhouse gas emissions as a result of electricity use are estimated to be approximately 679 tons of CO_2-e per day, or 5.43 kg of CO_2-e per m^3 of desalinated water, which represents a 2% increase in emissions in the Gold Coast region [157].

The real energy demand arising from the *Melbourne* SWRO project was estimated to be 3,239 tons of CO_2-e per day which equates to 7.45 kg of CO_2-e per m^3 of desalinated water. This value covers electricity used to drive the process and transfer the water, as well as emissions from the transportation of workforce, wastes and chemicals, from offsite waste decomposition, and embodied in the chemicals used during operation. If furthermore the energy used during construction of the project is added, amortized over the project life of 30 years, the energy demand amounts to 7.8 kg of CO_2-e per m^3. The construction process accounts for only 4% of the total project emissions. 75% of the construction-related emissions stem from the desalination plant, 15% from the marine structures, and 10% from the power and water grid connections [154].

The electricity demand arising from the *Sydney* SWRO project with an initial capacity of 250,000 m^3/d may result in emissions of 4.24 kg of CO_2-e per m^3 or 1,060 tons of CO_2-e per day. Similar to the Melbourne project, a LCA found that 5% of the total project emissions are associated with the materials and construction stages, and 95% with operation of the plant, which is mainly electricity use [155]. The electricity demand results in a 1.2% increase in New South Wales' electricity demand and compares with a predicted ongoing annual increase of around 3%.

The Perth project has the lowest grid-specific emission factor and lowest reported energy demand of the Australian projects given in Table 18, resulting in the lowest CO_2-e emissions of 3.43 kg/m^3 for the whole plant. The desalination-specific emissions amount to 325 t/d. The plant provides 17% of Perth's water, or enough water for about 0.3 million people. The emissions of 325 t/d are the same as if all car owners within this share of the population were to drive an additional 13.3 km every day[f].

The above examples all calculate the greenhouse gas emissions for electricity purchased from the grid applying the grid-specific emission factor. It would also be possible to co-locate a desalination plant to an existing power plant, or to build a power plant adjacent to the desalination plant, which would supply electricity 'over-the-fence' to avoid transmission losses. In Ashkelon, the desalination process is driven by a gas-fired power plant on site, which supplies 50 MW of electricity to the desalination plant with a capacity of 330,000 m^3/d (3.6 kWh/m^3). Applying the CO_2 emission factor for natural gas (Table 16) results in a very low emission factor of 0.73 kg CO_2 per m^3. Even if one adds a factor of two to take the full fuel life cycle into account, the Ashkelon project still has the lowest global warming potential from the projects listed in Table 18.

A worst case example are the Gulf countries which depend heavily on desalinated seawater from co-generation plants. In Kuwait, for instance, co-generation plants produce 90% of the national water supply and are almost exclusively fired by heavy crude oils. Kuwait has the fourth largest seawater desalination capacity on a global scale after Spain and accounts for 6% of the worldwide production (section 1.4.1). In 2004, the plants generated 42 million MWh of electricity and 443 million m^3 of water, using 462 million GJ of energy, which is 54% of the national fuel use. The corresponding CO_2 emission factors are 0.7 kg/kWh for electricity and 15.7 kg/m^3 for desalination [252], compared to, e.g., 0.5 kg/kWh for the Spanish electricity mix and 2 kg/m^3 for the Spanish SWRO plants. The total CO_2 emissions are approximately 19.000 t/d for 1.2 million m^3 of water per day, compared to 'only' 5,475 t/d for 2.7 million m^3 in Spain. While desalination accounts for < 1% of the Spanish national CO_2 emissions, it accounts for 10% of the national fuel use and hence the national emissions in Kuwait. The sustainability of energy and water production in Kuwait has therefore been questioned [252].

The Australian Department of Climate Change recommends that the following emissions be included in the estimates for new development projects, which can also be adapted to desalination projects [271]:

- all *direct* emissions from sources within the boundary of the project including:
  - the generation of electricity and heat produced on-site including CO_2 and products of incomplete combustion, i.e., methane and nitrous oxide. This applies to the electricity produced for SWRO plants on-site, as in Ashkelon, or the electricity and heat needed to drive the desalination process in thermal co-generation plants.
  - the manufacturing process itself. This may apply to the release of dissolved gases from seawater during the process of desalination. The oceans store CO_2 in the form of bicarbonate. In thermal plants, the feed seawater is deaerated during pretreatment, and gases evolve from the evaporating water in the flashing chambers. In SWRO plants, acid is often added during pretreatment, which increases calcium carbonate solubility and forces CO_2 out of solution.
  - the transportation of materials, products, waste and people.
  - fugitive emissions from waste management, such as emissions from landfill sites.

[f] Assuming an economic car with a mileage of 5 liters/100 km, a full fuel cycle emission factor of 2.5 tons CO_2-e per m^3 of fuel [fuel combustion of the car of 2.3 tons CO_2-e per m^3 plus fuel extraction of 0.2 tons CO_2-e per m^3, 271], and 650 cars per 1000 Perth residents in 2006 [272].

- all *indirect* emissions including:
  - the consumption of electricity from the grid, taking emissions associated with transmission and distribution losses into account.
  - upstream emissions generated in the extraction and production of fossil fuels.
  - downstream emissions from transport of the product (water) to customers.
  - the manufacturing of materials used during construction, operation, and maintenance, including the membranes, chemicals, steel and other metallic equipment and concrete for construction of the desalination plant.

Other air pollutants
While CO_2 emissions can be estimated with a relatively high degree of certainty, emissions of other air pollutants depend on the fuel type as well as on the technology that is used to minimize pollutant emissions at the source (if any), such as for example SO_2 scrubbers. Non-carbon dioxide emissions are therefore more difficult to quantify.

For example, the leading Spanish electric utility company ENDESA specifies emission factors of 6.21 g/kWh for SO_2, 1.66 g/kWh for NO_X, and 0.12 g/kWh for PM for the company's power generating plants on the Iberian Peninsula. Assuming an average total electricity demand of 4 kWh/m^3 for the Spanish SWRO plants, this equates to emissions of 24.8 g SO_2, 6.6 g NO_X and 0.5 g PM per cubic meter of desalinated water. The targeted seawater desalination capacity of 2.7 Mm^3 in 2010 may hence result in emissions of 67.1 t SO_2, 17.9 t NO_X and 1.3 t PM per day, based on the ENDESA values and not taking the real energy mix of the grid and transmission losses into account[g].

The daily direct and indirect emissions of SO_X, NO_X, PM_{10} and other air pollutants such as carbon monoxide and reactive organic compound (an ozone precursor substance) were estimated for a large SWRO plant in Southern California with a projected capacity of 200,000 m^3/d [132]. The daily direct emissions associated with landscaping, delivery trucks and employee vehicles amounted to 15 kg SO_X, 27 kg NO_X and 29 kg PM_{10}. The indirect daily emissions are caused by electricity production to provide the electrical energy for the facility and amounted to <0.1 kg SO_X, 3 kg NO_X and 0.1 kg PM_{10}. It is interesting to note that the Environmental Impact Report for this project concluded that *operation* activities including direct and indirect emissions will not exceed any established air quality thresholds, but that *construction* activities may result in NO_X-emissions of 176 kg per day that could temporarily and locally exceed established air pollutant emission thresholds. The estimated SO_X and PM_{10} emissions resulting from construction activities amounted to 15 kg and 14 kg per day, respectively.

Construction-related emissions include exhaust generated by construction equipment, trucks and worker vehicles as well as fugitive dust generated by demolition of structures, site grading and trenching. Fugitive dust and diesel exhaust are the main contributing factor to increased levels of particulate material in the construction site. All air pollutant emissions are project-specific, however, the example illustrates the order of magnitude of construction-related air emissions and indicates that construction causes a localized and temporal, but measurable increase in air pollutants, which may violate air quality standards in the worst case. Project-specific emission estimates, based on the specific emission factors of construction vehicles and fuel type, existing background levels and other

[g] It should be noted that the grid average value will likely be different from the values given by the leading utility company, which refer to the company's power generating plants and not to the whole energy mix in the grid, and which do not take other factors such as transmission losses of the grid into account.

emission sources in the vicinity, need to be taken into consideration when evaluating if project-related construction activities may violate any existing air quality standards.

In Kuwait, emissions of SO_X and NO_X are much higher than in the two examples above, which can be attributed to the high shares of heavy crude oil used in the cogeneration plants. Heavy oil accounts for 78%, other crude oil for 20%, and diesel for 2 % of the fuel [252][h]. Heavy oil contains high amounts of nitrogen, sulfur, oxygen, heavy metals, cycloalkanes and aromatics. Emission factors for SO_2 and NO_X have been published for Kuwaiti cogeneration plants, which are 284 g SO_2 per m^3 and 33.6 g NO_X per m^3 of desalinated water, resulting in emissions of 344.7 tons SO_2 per day and 40.7 tons NO_X per day for the production of 1.2 Mm^3 of desalinated water [252]. No explicit emission standards exist for power and water cogeneration plants in most Gulf countries [166].

4.4 Other environmental impacts associated with energy use

When existing power plant capacities are increased or new plants constructed in order to provide electricity for the desalination of seawater, impacts associated with power production will likely be intensified. For coastal power plants using once through cooling water systems, major concerns are the entrainment and impingement of marine organisms (cf. also section 3.2, page 57) and impacts related to the discharge of residual chemicals (e.g., chlorine, cf. also section 3.3.2, page 64) or waste heat. For example, co-generation plants in Kuwait have a fuel use efficiency of 38%, which means that 62% of the energy is rejected as heat. 16% of the heat is dissipated into the atmosphere and 84%, equalling 666 TJ/day, is rejected into the sea as cooling water. 60% of the cooling water discharges are attributed to power and 40% to water production in the cogeneration plants [252].

4.5 Summary and conclusions

The main environmental concerns of fossil energy use are the resulting emissions of greenhouse gases, acid rain gases (SO_X, NO_X), fine particulate matter (PM) and other air pollutants. All desalination processes consume the highest amount of energy during operation, whereas only a low share of the total energy use and resulting emissions, in the range of 5-10%, is attributed to materials use and the construction stage.

Greenhouse gases are relevant in the context of national and international efforts to curb these emissions to minimize climate change impacts, whereas other air pollutants could directly or indirectly conflict with local air quality management plans by exceeding existing limits or by contributing substantially to existing background levels in the area (cumulative impacts). Local air quality impacts may for example arise from a high number of heavy vehicles during construction, or insufficient air purification equipment in the power plant providing the electricity or heat for the desalination process.

Concerns may also arise due to more indirect impacts, such as the cooling water requirements of power plants or the increasing risk for accidents associated with the transport of fuels. When existing power plant capacities are increased in order to provide more electricity or heat for desalination, these indirect impacts will likely be intensified.

[h] The specific gravity of crude oil, as measured on the American Petroleum Institute (API) gravity scale, may range from 10° to more than 60°. Crude oils below 20° API gravity are usually considered to be heavy. Kuwaiti oil fields generally contain medium to light crude oil with gravities in the 30° to 40° API range, and heavy oil in some northern fields and shallow wells [273].

Compared to other activities and amenities of modern lifestyles, such as air conditioning or heated water, desalinated water is not an overly energy-expensive product. However, it is far more energy-intensive than the treatment and distribution of local ground and surface water sources, and is still often more energy-intensive than the transport of water over long distances. The energy use of desalination is therefore a matter of controversial public debate, however, an adequate debate would also take the wider implications of energy and water use into account, which is not always the case. Simply understating the problem by comparing the energy demand of desalination to even more energy-intensive forms of use is not instrumental in the discussion, nor is it to belabor the point if it is clear that conventional water supplies cannot meet the demand.

The wider implications in a debate about a new desalination project are usually how much do consumers value the water as an amenity of modern life-styles, involving swimming pools and irrigation in home gardens as in Southern California or Perth, and how much impact are they willing to accept for it. The price tag does not only include energy use, but other environmental impacts as well. As the problem increases in complexity, double moral standards may be applied in the discussion. The sustainability of desalination is often questioned on the grounds of high energy use and potential marine impacts – under the tacit assumption that the status quo or other alternatives of water supply are more sustainable, which is not necessarily the case.

Desalination is without question an energy-intensive option, but the status quo is that energy is often wasted in other sectors of use, for example by old and inefficient electrical appliances in households. Energy saving in households has not yet been tapped to the full potential, and could possibly make up for the additional energy demand of providing desalinated water to households. The status quo of existing water supplies, which may involve the depletion of ground water resources, or the damming, regulation or diversion of rivers, may also prove unsustainable. According to the World Commission on Dams, a considerable portion of the world's large dams is falling short of their physical and economic targets, that is they deliver less water and electricity than promised whilst significantly overrunning costs, besides having extensive impacts on aquatic ecosystems, which, in many cases, have led to irreversible losses of species and ecosystems [274].

In the end, it depends on the perception and definition of significance and on local circumstances whether or not a community or individual considers energy use of desalination as a *significant* factor. In many parts of the world, energy use is generally recognized as a high priority issue – and hence significant – issue. For example, the Sydney desalination plant received more than 700 formal submissions from the community and stakeholders of which nearly 550 raised concerns about energy use and greenhouse gas emissions [232], which had also been major considerations in the project's EIA [155]. In contrast, CO_2 emissions were not even addressed in the EIA for the Carlsbad SWRO project in Southern California [132], but a Climate Action Plan at an estimated US$ 76 million was nevertheless imposed on the project to mitigate greenhouse gas emissions. Even in Kuwait, which has one of the highest per capita energy and water uses in the world – a luxury it can only afford because it possesses roughly 8% of the world's total oil reserves [273] – it is now realized that the current wasteful use and growth in demand has to be curbed in order to not jeopardize the needs of future generations. Otherwise, the oil production may not be enough to provide the resident population with water and air-conditioned housing in about thirty years time [252].

The perception that energy use is a *significant* aspect is reflected in policy initiatives and stricter standards in many countries to reduce energy consumption and increase energy-use efficiency in all sectors of use. On the project level, the concept of environmental impact assessment stipulates that for all *significant* impacts of a development project, impact mitigation measures have to be identified and implemented, involving a hierarchy of measures from prevention to minimization and compensation (chapter 7, page 156). In all large Australian SWRO projects, CO_2 emissions are a central issue [e.g., Perth, Sydney, Melbourne, 140, 155, 156] and project proponents are encouraged to provide for the use of energy from renewable sources, planting of plantations or rehabilitation of vegetation to offset the emissions [156]. For example, the two SWRO plants in Perth and the Sydney project all compensate their electricity demand by a newly erected wind farm which compensates for the electricity taken from the grid [61, 69, 275].

For the Perth I plant, an associated 82 MW wind farm with 48 turbines was erected 200 km north of the city, which will inject an expected 272 GWh of renewable energy into the grid per year, from which the Perth I SWRO plant purchases 185 GWh (68%) per year [68, 70]. The wind farm that has been purpose-built near Canberra to offset the energy use of the Sydney SWRO project will consist of 67 wind turbines and will provide 132 MW of energy to the grid, versus 42 MW required to operate the plant [61].

Additional measures of mitigating and compensating the environmental impacts of desalination projects are discussed in further detail in chapter 6.

Summary and evaluation of impacts | 5

5.1 Introduction

In the two previous chapters the key environmental concerns of desalination plants were outlined, i.e., the impacts on the marine environment due to seawater intakes and concentrate discharges, and air quality impacts due to energy use. A full-fledged EIA would not be limited to a few key aspects, but would have to consider all possibly relevant impacts related to the construction, commissioning, operation, maintenance and decommissioning of a new project, followed by an evaluation of the significance of the identified impacts. This chapter synthesizes the potential impacts of desalination plants into a series of stressor-response relationships, adopting the approach of an ecological risk assessment, and subsequently evaluates the stressor-response relationships in terms of significance.

5.2 Ecological risk assessment

The main goal of an ecological risk assessment is to systematically identify and evaluate the relationships between all *stressor* sources as caused by anthropogenic activity and subsequent impacts on environmental *receptors*. *Stressors* can be all single characteristics of a project or activity that lead to an ecological effect. They can be of chemical, physical, or biological nature, such as the emissions of residual chemicals from the process, the mechanical impact from construction, or the introduction of an alien species. The *receptors* are the different environmental features, usually operationally defined by an ecological entity (e.g., a single species or its indicator such as population size). The aim is to describe the exposure of each receptor to all stressors in terms of *intensity, space, and time*. To this end, exposure pathways are established, including the source and the spatial and temporal distribution of stressors in the environment, and the extent and pattern of co-occurrence with receptors (exposure analysis). The relationship between stressor levels and resulting responses is then investigated (effects analysis). In essence, the ecological risk assessment approach is based on an analysis of how exposure to stressors is likely to occur, and on an analysis of the *significance* of the associated impacts, which ultimately leads to one overall *risk characterization* [276].

Parts of this chapter were based on:

S. Lattemann and H. El-Habr. UNEP resource and guidance manual for environmental impact assessment of desalination projects. Desalination and Water Treatment, 3: 217–228, 2009.

S. Lattemann, K. Mancy, H. Khordagui, B. Damitz and G. Leslie. Desalination, resource and guidance manual for environmental impact assessments. United Nations Environment Programme (UNEP), Nairobi, Kenia, 2008.

Because of the complexity of large development projects and ecosystems, the result is usually a long-list of stressor-response relationships[a]. These are often interrelated and have a netlike rather than a linear structure, as one stressor may lead to multiple exposures and may also cause secondary, indirect effects. The establishment of single stressor-response relationships should therefore be understood as a simplified conceptual approach which is used to systematically predict and investigate the key relationships between stressors and receptors. The accuracy of this approach depends on how well information on stressor sources, exposure pathways and ecological effects is available.

5.2.1 Risk matrix

The stressor-response relationships are typically summarized in a risk matrix (preference matrix or Leopold matrix), in which the columns represent the various stressors of a project and the rows the various environmental receptors (or media such as water). The potential ecological effects are listed where rows and columns intersect. A risk matrix and risk characterization are often developed at the end of the EIA (cf. also section 7.2).

The stressor sources of desalination projects can be subdivided into the life cycle stages of construction, commissioning, operation, maintenance, and decommissioning/demolition of the project. The project itself usually comprises the intake and outfall systems, the desalination process, and auxiliary infrastructure (Table 19).

An EIA should generally address the effects of a project on fauna, flora, soil, water, air, climate and landscape, including all direct and indirect effects and the interactions between single factors [European EIA directive][b]. Different receptor categories for desalination projects, as given in Table 19, were derived from this definition.

A detailed description of the stressor-response relationships is provided in Lattemann et al. [17]. For lack of space, only the risk matrix will be reproduced here, which has been split into separate tables, one for each receptor category (Tables 21 to 34, page 108ff.), and includes an exposure evaluation in terms of intensity, space, and time.

5.2.2 Risk characterization

The risk matrix provides the basis for risk characterization, in which the stressor-response relationships are integrated into an overall risk estimation and description, which takes the significance and likelihood of effects into account as well as uncertainty in the underlying data. Risk characterization is to be distinguished from decision making, which involves the selection of a course of action in response to the identified risks and other social, legal, political, or economic factors [276].

Evaluation of significance allows project planners and regulators to distinguish between impacts with a high priority for further investigation and impact mitigation and those of lower importance. As the definition of significance depends on many factors, including social and political values, a project- and site-specific evaluation of significance is indispensable. The following assessment is therefore only a general attempt to prioritize impacts. Its primary purpose is to provide a certain level of indicative guidance by identifying evaluation criteria and environmental effects, which will typically have a high priority for project- and site-specific investigations, and which will typically result in the establishment of certain impact mitigation measures.

[a] Also termed cause-effect relationships.

[b] Directive 85/337/EEC on the assessment of the effects of certain public and private projects on the environment, amended by Directive 97/11/EC.

Table 19: Stressors (key features) of desalination projects and environmental receptors.

Stressors	Environmental receptors
▸ Intake system, including: • for surface intakes: inlet with screens • for subsurface intakes: horizontal drains or vertical wells • seawater pipeline(s) to the plant • pumping station ▸ Desalination system, including: • pretreatment line • desalination units • product water storage • pumping / high pressure system • post-treatment line • storage space for consumables • car park, gates, etc. ▸ Outfall system, including: • outfall channel or tunnel • diffuser system • pumping station ▸ Auxiliary infrastructure, including: • water distribution pipeline • energy source and supply line • access roads to the facility	▸ Landscape and natural scenery (Table 21) ▸ Air quality and climate (Table 22) ▸ Soils (Table 23) ▸ Seafloor and sediments (Table 24) ▸ Seawater quality and hydrology (Table 25) ▸ Ground- and surface water quality and hydrology (Table 26) ▸ Terrestrial flora (Table 27) and fauna (Table 28), which can be further subdivided into different functional groups, i.e., plant communities and habitat types such as salt marshes or dune vegetation, or taxonomic groups such as invertebrates, mammals, amphibians, reptiles, and birds including migratory and resting seabirds, if necessary. ▸ Marine flora and fauna, which can be further subdivided into the different functional and taxonomic groups, i.e., benthic macroalgae and seagrasses (Table 29), phyto- and zooplankton (Table 30), benthic invertebrate species (Table 31), fish species (Table 32), marine reptiles and mammals (Table 33) and terrestrial birds and seabirds (Table 34).

Methodology
The stressor sources were rated in terms of *intensity, space and time* on a three-stage ordinal scale (Table 20). Space and time refer to the spatial and temporal distribution of the stressor source. Whether or not an exposure occurs also depends on the spatial and temporal distribution of the receptors in the environment (i.e., the distribution of algae stands or benthic species in the project site). It was therefore generally assumed that the receptor is present in the impacted area. A categorization into far-range, mid-range and localized effects was made with regard to the spatial distribution of the stressor source. The temporal distribution of the stressor source was evaluated by classifying the effects into long-term, medium-term, and short-term effects. The different categories are described in the following.

Far-range effects were defined as those effects which are noticeable beyond 1 km of the point of origin, which could for example be underwater noise emissions during construction. *Mid-range* effects were considered to be those effects which are limited to the project site and nearby areas and typically do not exceed a range of 1 km. A mid-range effect may be caused by the formation of turbidity plumes when marine sediments are suspended into the water column during construction. *Localized* effects would occur only punctually, and are limited in their range to the project site within 100 m distance of origin, such as the construction impact on the local habitat.

Long-term effects were defined as those effects which occur continuously or regularly over the entire project life (e.g., due to the continuous or frequent emission of a waste product, such as the concentrate or filter backwash wastes), including permanent or irreversible effects. *Medium-term* effects would be those effects which last for several years, including periodic events that occur several times per year. *Short-term* effects have a duration of less than one year and are generally reversible.

Concerning the *intensity* rating, a classification was made into *severe, notable and negligible* alterations of natural properties, functions, or processes. It was assumed that the impacts are caused by a large facility, as the intensity of environmental impacts can be assumed to increase with the size and production capacity of a desalination plant. It was furthermore assumed that no impact mitigation measures have been adopted yet. Finally, an attempt was made to include an estimate of the likelihood of the effect (likely, possible, unlikely), taking the likelihood of stressor occurrence (e.g., the likelihood of a chemical spill, which is unlikely) as well as receptor occurrence (e.g., the likelihood that a mobile species may be exposed, which is also unlikely) into account.

The single ratings for intensity, space and time were formally integrated into a single rating following the aggregation logic shown in Figure 17. The overall rating reflects the significance of that effect for project- and site-specific EIA studies and for impact mitigation measures (high, medium, low priority). The probability criterion was not formally integrated but used as an indicator. When a result between two ratings was obtained, the next higher rating was usually selected as a precautionary approach, especially when an effect is likely to occur. The following effects of *high* respectively *low priority* can consequently be distinguished:

Impacts of *high priority* (⊖) for project- and site-specific EIA studies and for impact mitigation are consequently those which fulfill the following criteria:

- *Severe* alterations of natural properties, functions or processes, which are of
  - *long-term* duration and *far range*,
  - *long-term* duration and *mid range*,
  - *medium-term* duration and *far range*.
- *Notable* alterations of natural properties, functions or processes, which are of *long-term* duration and *far-range*.

Impacts of *low priority* (⊗) for project- and site-specific EIA studies and for impact mitigation are consequently those which fulfill the following criteria:

- *Negligible* alterations of natural properties, functions or processes of
  - *short-term* duration and *localized*,
  - *short-term* duration and *mid range*,
  - *medium-term* duration and *localized*.
- *Notable* alterations of natural properties, functions or processes, which are of *short-term* duration and *localized*.

► **Tables 21 to 34** show the stressor-response relationships for each receptor category (as identified in **Table 19**), including their exposure evaluation in terms of intensity, space, and time (as defined in **Table 20**), and their overall aggregation into impacts of high priority (⊖), impacts of medium priority (⊙), and impacts of low priority (⊗) according to **Figure 17**. The impacts of high priority are summarized in **Table 35** on page 120.

Table 20: Criteria and their definition for the evaluation of significance [adapted from 277, 278].

Impact rating		Description	Significance
Intensity	severe	severe alteration of natural properties, functions, processes	high
	notable	notable alteration of natural properties, functions, processes	medium
	negligible	negligible alteration of natural properties, functions, processes	low
Duration	long-term	continuously or regularly (once per day) over project life, permanent or irreversible effects (including aftermath effects)	high
	medium-term	several years (<5) of duration (including aftermath effects), reversible, periodic events (several times per year)	medium
	short-term	less than one year or restricted to construction, reversible	low
Spatial extent	far-range	effects beyond project site and nearby areas, beyond 1,000 m distance of origin	high
	mid-range	within the project site and nearby areas, within 1,000 m distance of origin	medium
	localized	punctual, within the area of the project site, within 100 m distance of origin	low
Probability	definite/likely	highly probable or definite (>80%)	high
	possible	fair chance of occurring	medium
	unlikely	little or no chance of occurring (<20%)	low

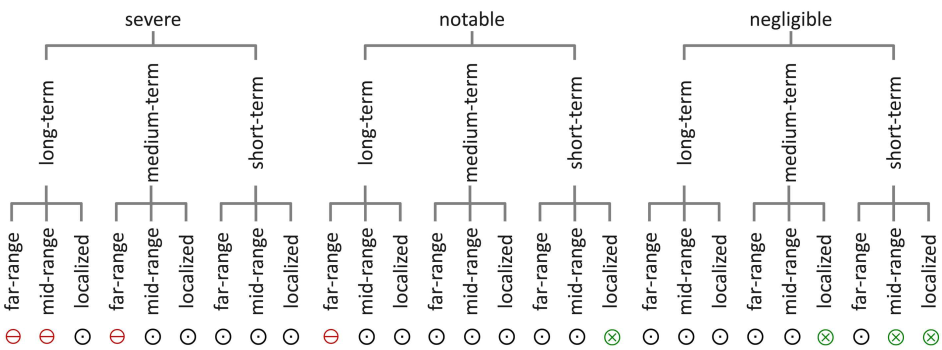

Figure 17: Evaluation of significance of impacts – Aggregation logic for the criteria 'intensity' (severe, notable, negligible), 'duration' (long-term, medium-term, short-term), and 'spatial extent' (far-range, mid-range, localized), and overall rating (⊖ impact of high priority, ⊙ impact of medium priority, ⊗ impact of low priority).

Table 21: Landscape and natural scenery – evaluation of impacts.

Life cycle stage		Effects on landscape and natural scenery	Intensity	Duration	Spatial extent	Probability
Construction						
Offshore facilities	⊙	sediment plume may increase water turbidity	notable	short-term	localized to mid-range	definite
	⊙	underwater noise emissions and machinery positioned on land	notable	short-term	mid-range to far-range	definite
Onshore facilities	⊙	potential visual and acoustic impacts due to movements, dust, exhaust fumes, noise or stockpiles exposed to public views	notable	short-term	mid-range to far-range	definite
	⊙	upon completion, visual appearance of buildings, prominent features, plumbing or power lines, glare, and light sources, noises, etc., which may alter landscape properties	notable	long-term	mid-range to far-range	definite
Operation						
Backwash water	⊖	reddish plume near the outlet and surrounding areas possible when $FeCl_3$ is used and potential discoloration of nearby beaches	notable	long-term / intermittent	mid-range to far-range	likely if discharged
Noise emissions	⊖	may impair landscape properties within acoustic range	notable	long-term	mid-range to far-range	likely

Table 22: Air quality and climate – evaluation of impacts.

Life cycle stage		Effects on air quality and climate	Intensity	Duration	Spatial extent	Probability
Construction						
Offshore facilities	⊙	emissions of air pollutants (NO_X, SO_X, PM_{10}) and greenhouse gas (CO_2) from construction machinery	notable	short-term	mid-range to far-range	definite
Onshore facilities	⊙	emissions of air pollutants (NO_X, SO_X, PM_{10}) and greenhouse gas (CO_2) from construction machinery	notable	short-term	mid-range to far-range	definite
	⊙	fugitive dust from demolition of buildings and site grading	notable	short-term	mid-range to far-range	definite
Operation						
Chemical storage	⊙	accidental spillage or leakage of volatile substances may cause air pollution (e.g., chlorine)	severe	short-term	mid-range to far-range	unlikely
Energy use	⊙	emissions of air pollutants and greenhouse gas from trucks and passenger cars (CO_2, NO_X, SO_X, PM_{10})	negligible	long-term	mid-range to far-range	definite
	⊖	emissions of air pollutants and greenhouse gas from power generation (CO_2, NO_X, SO_X, PM_{10})	notable	long-term	mid-range to far-range	definite

Table 23: Terrestrial soils – evaluation of impacts.

Life cycle stage	Effects on terrestrial soils	Intensity	Duration	Spatial extent	Probability
Construction					
Offshore facilities	⊙ construction in the landing area may affect beachslope stability, dune systems, etc., and may cause erosion by wind and waves where vegetation has been cleared	severe	medium-term to long-term	localized	possible
	⊙ in case of horizontal drilling: debris stockpiles from the borehole may have placement impacts and may require offsite disposal	notable	short-term to medium-term	localized	possible
Onshore facilities	⊙ soil compaction through machinery	notable	short-term to medium-term	localized	definite
	⊙ erosion may occur where vegetation has been cleared	severe	medium-term to long-term	localized	possible
	⊙ stockpiles of excavated material may have placement impacts and may require offsite disposal	notable	short-term to long-term	localized	likely
	⊙ accidental spillage or leakage of fuels, chemicals, or lubricants may cause soil contamination	severe	short-term to medium-term	localized	possible
	⊙ upon completion, surface sealing caused by asphalt and buildings	severe	long-term	localized	definite
Operation					
Backwash water	⊙ backwash sludge may require offsite disposal	notable	long-term	localized	possible
	⊙ spreading on land may affect soil properties	notable	medium-term to long-term	localized to mid-range	possible
Chemical storage	⊙ accidental spillage or leakage may contaminate soils	severe	short-term to medium-term	localized	unlikely
Membrane and cartridge replacement	⊙ disposal may require an appropriate site for landfill	notable	long-term	localized	possible

Table 24: Seafloor and sediments – evaluation of impacts.

Life cycle stage		Effects on seafloor and sediments	Intensity	Duration	Spatial extent	Probability
Construction						
Offshore facilities	⊙	sediment layering and structure may be disturbed	notable	short-term to medium-term	localized	likely (if excavating)
	⊙	sediment compaction from machinery	notable	short-term to medium-term	localized	
	⊙	surface sealing (if structures placed on the seabed)	severe	long-term	localized	definite
	⊖	upon completion, structures may act as breakwaters, changing erosion / sedimentation processes locally / downdrift locations	severe	long-term	localized to far-range	possible
	⊙	accidental spillage or leakage of fuels, chemicals, or lubricants may cause sediment contamination	severe	short-term to medium	localized	possible
Onshore facilities	⊗	loose or contaminated soils and other material washed away by runoff or eroded by wind may affect sediments	notable	short-term	localized	possible
Operation						
Concentrate discharge	⊖	discharge plume may sink to the seafloor and may cause an increase in porewater salinity due to diffusion	severe	long-term	localized to mid-range	likely
Residual chemicals in the concentrate	⊖	heavy metals (if present in the concentrate from corrosion processes) may accumulate in sediments in the discharge site	severe	long-term	localized to mid-range	possible
Backwash water	⊖	sedimentation and accumulation of coagulants in sediments	severe	long-term	localized to far-range	likely

Table 25: Seawater quality and hydrology – evaluation of impacts.

Life cycle stage		Effects on seawater quality and hydrology	Intensity	Duration	Spatial extent	Probability
Construction						
Offshore facilities	⊙	resuspended sediments may increase turbidity, pollutant or nutrient levels or decrease oxygen levels	notable	short-term	mid-range to far-range	likely
	⊙	upon completion, structures may act as breakwaters and change wave patterns and currents	notable	long-term	localized to mid-range	possible
	⊙	accidental spillage or leakage of fuels, chemicals, or lubricants may cause water pollution	severe	short-term	localized to mid-range	possible
Onshore facilities	⊙	loose or contaminated soils and other material washed into the sea by runoff or eroded by wind may affect water quality	notable	short-term	localized to mid-range	possible
Commissioning						
Waste streams	⊙	membrane storage solutions could affect water quality if discharged into the sea	notable	short-term	localized to mid-range	possible
Operation						
Intake	⊙	open intakes may change water circulation in the intake area when large volumes of water are extracted	notable	long-term	localized to mid-range	likely
Concentrate	⊖	increases in salinity in the mixing zone	severe	long-term	localized to mid-range	definite
discharge	⊙	large discharge flows may affect water circulation and mixing processes in the discharge area	notable	long-term	localized to mid-range	likely
	⊖	increased density may cause seafloor spreading of the plume	severe	long-term	localized to mid-range	definite
	⊖	stratification of the water column may be strengthened	severe	long-term	localized to mid-range	possible
	⊖ ⊗	stratification may impede re-oxygenation of bottom waters (whereas turbulent discharge may add oxygen to bottom layers)	severe (positive)	long-term (long-term)	localized to mid-range	possible
	⊙	potential enrichment of nutrients, organic matter, pollutants or trace metals	notable	long-term	localized to mid-range	possible
Residual chemicals in the concentrate	⊙	residual chlorine and chlorination by-products possibly detectable in the mixing zones (if no dechlorination step)	notable	long-term	localized to mid-range	possible
	⊙	sodium bisulfite is a reducing agent and may decrease dissolved oxygen levels if overdosed	notable	long-term	localized	possible

continued on next page

Life cycle stage		Effects on seawater quality and hydrology	Intensity	Duration	Spatial extent	Probability
Residual chemicals in the concentrate	⊙	heavy metals (if present from corrosion processes) may affect dissolved metal concentrations in the mixing zone	notable	long-term	localized to mid-range	possible
	⊙	antiscalants may bind nutrients and ions dissolved in seawater	notable	long-term	localized to mid-range	possible
	⊗	a weak surplus acidity may be discharged which would be neutralized quickly by ambient seawater	negligible	short-term	localized	possible
Backwash water	⊖	increased turbidity / decreased light penetration in discharge area	severe	long-term / intermittent	localized to mid-range	definite
Chemical storage	⊙	accidental spillage or leakage may contaminate seawater	severe	short-term	localized to mid-range	unlikely
Maintenance						
Cleaning solutions	⊙	discharge of acidic or alkaline cleaning solutions may affect the ambient seawater pH in the mixing zone	severe	short-term	localized	likely if discharged
	⊙	detergents or complexing agents may interfere with natural processes of dissolved seawater constituents (e.g., metals)	notable	short-term to medium-term	localized to mid-range	possible if discharge

Table 26: Ground- and surface water quality and hydrology – evaluation of impacts.

Life cycle stage		Effects on ground- and surface water quality and hydrology	Intensity	Duration	Spatial extent	Probability
Construction						
Onshore facilities	⊙	accidental spillage or leakage of fuels, chemicals, or lubricants may cause ground- and surface water pollution	severe	short-term	localized to mid-range	possible
	⊙	loose or contaminated soils and other material washed away by runoff or eroded by wind may affect surface water quality	notable	short-term	localized to mid-range	possible
	⊙	groundwater table may be affected by construction / drainage	severe	short-term	localized to mid-range	possible
Operation						
Intake of feedwater	⊖	intake from aquifers may change flow directions and changes in groundwater salinity	severe	long-term	localized to mid-range	possible
Concentrate discharge	⊖	well injection may cause an increase in groundwater salinity	severe	long-term	localized to mid-range	possible
Backwash water	⊖	potential seepage from landfill disposal into groundwater	severe	long-term	localized to mid-range	possible
Chemical storage	⊙	accidental spillage or leakage may cause contamination	severe	short-term	localized to mid-range	unlikely

Table 27: Terrestrial flora – evaluation of impacts.

Life cycle stage		Effects on terrestrial flora	Intensity	Duration	Spatial extent	Probability
Construction						
Offshore facilities	⊙	construction in the landing area may require clearing of vegetation	severe	short-term to medium-term	localized	likely
Onshore facilities	⊙	clearing or flattening of vegetation in construction site (impact depending on area size or route and site vegetation)	severe	short-term to medium-term	localized to mid-range	likely
	⊙	potential weed infestations in cleared areas	notable	short-term to medium-term	localized	possible
	⊙	potential contamination by spills or leakages	severe	short-term	localized	possible
	⊙	upon completion, permanent loss of land usable by native plants in all areas covered by solid surfaces or landscaped areas	severe	long-term	localized	definite
Operation						
Chemical storage	⊙	potential exposure to harmful substances by accidental spills	severe	short-term	localized	unlikely

Table 28: Terrestrial fauna – evaluation of impacts.

Life cycle stage		Effects on terrestrial fauna	Intensity	Duration	Spatial extent	Probability
Construction						
Offshore facilities	⊙	construction in the landing area may disturb wildlife	notable	short-term	mid-range	likely
Onshore facilities	⊙	construction, e.g., through noise and vibrations, may cause behavioral responses and temporary habitat loss	notable	short-term	mid-range	likely
	⊙	potential contamination by spills or leakages	severe	short-term to medium-term	localized	possible
	⊖	upon completion, habitat alteration / loss for native species	severe	long-term	mid-range	definite
	⊖	prominent features could preclude linkages and movement corridors of wildlife	severe	long-term	mid-range	definite
Operation						
Chemical storage	⊙	potential exposure to harmful substances	severe	short-term	localized	unlikely
Noise emissions	⊖	may scare away sensitive wildlife within acoustic range, causing a potential long-term habitat loss	severe	long-term	mid-range	likely

Table 29: Marine macroflora – evaluation of impacts.

Life cycle stage		Effects on marine macroflora	Intensity	Duration	Spatial extent	Probability
Construction						
Offshore facilities	⊙	habitat destruction by excavation works	severe	short-term to medium-term	localized	likely
	⊙	increased turbidity / reduced light penetration	notable	short-term	localized to mid range	likely
	⊙	increased turbidity / increased sedimentation rates / blanketing	severe	short-term		likely
	⊙	potential impacts from remobilization of nutrients or pollutants from sediments	notable	short-term	localized to mid range	possible
	⊙	potential contamination by spills or leakages	severe	short-term	localized	possible
	⊗	upon completion, macroalgae can attach to subsurface structures (artificial reefs)	notable (positive)	long-term	localized	likely
Onshore facilities	⊙	potential burial by soils or other material washed into the sea	severe	short-term	localized	possible
Commissioning						
Waste streams	⊙	may be exposed to residual chemicals (if discharged)	severe	short-term	localized	possible
Operation						
Intake	⊙	open intakes cause entrainment of spores	notable	long-term	localized to mid range	definite
Concentrate discharge	⊖	increased salinity may cause a decline of algae stands and seagrass meadows (depends on exposure levels / species sensitivity)	severe	long-term	localized to mid range	definite
	⊙	nutrient enrichment may enhance algae growth (eutrophication)	notable	long-term	localized to mid range	possible
Residual chemicals in the concentrate	⊙	antiscalants are non toxic at the concentrations used but they may bind nutrients and ions needed for plant growth	notable	long-term	localized to mid range	possible
	⊖	residual chlorine levels and chlorination by products may have toxic effects on seagrass or algae stands in the mixing zone	severe	long-term	localized to mid range	likely
Backwash water	⊖	coagulants are non-toxic, however, turbidity may impair photosynthesis and could lead to a die-off of seagrass and algae stands	severe	long-term	localized to mid-range	likely
Chemical storage	⊙	potential exposure to harmful substances	severe	short-term	localized	unlikely
Energy use	⊙	when coastal power plant capacity increases: secondary effects from cooling water use (intake and discharge effects)	notable	long-term	localized to mid-range	possible
Maintenance						
Cleaning solutions	⊙	residual cleaning chemicals may be harmful	severe	short-term	localized	possible

Table 30: Marine plankton – evaluation of impacts.

Life cycle stage		Effects on marine plankton	Intensity	Duration	Spatial extent	Probability
Construction						
Offshore facilities	⊙	potential impacts from increased turbidity (reduced light penetration)	notable	short-term	localized to mid range	likely
	⊙	potential impacts from remobilization of nutrients or pollutants from sediments	notable	short-term	localized to mid range	possible
Commissioning						
Waste streams	⊗	may be exposed to residual chemicals (if discharged)	notable	short-term	localized	possible
Operation						
Intake	⊙	open intakes cause entrainment of phyto- and zooplankton	notable	long-term	localized to mid range	definite
Concentrate discharge	⊙	may be harmful or even toxic to organisms (depending on exposure levels and species sensitivity)	notable	long-term	localized to mid range	definite
	⊙	nutrient enrichment may enhance growth (algae blooms possible?)	notable	long-term	localized to mid range	possible
Residual chemicals in the concentrate	⊙	antiscalants are non toxic at the concentrations used but they may bind nutrients and ions needed for plankton growth	notable	long-term	localized to mid range	possible
	⊖	residual chlorine levels and chlorination by products may have toxic effects on plankton organisms in the mixing zone	severe	long-term	localized to mid range	likely
Backwash water	⊙	coagulants are non-toxic, however, they may lower light penetration and primary production in the water column	notable	long-term / intermittent	localized to mid range	likely if discharged
Chemical storage	⊗	potential exposure to harmful substances	notable	short-term	localized	unlikely
Energy use	⊙	when coastal power plant capacity increases: secondary effects from cooling water use (intake and discharge effects)	notable	long-term	localized to mid range	likely
Maintenance						
Cleaning solutions	⊗	high or low pH values and residual cleaning chemicals such as biocides may be harmful	notable	short-term	localized	possible if discharged

Table 31: Marine benthic invertebrate fauna – evaluation of impacts.

Life cycle stage		Effects on marine benthic invertebrate fauna	Intensity	Duration	Spatial extent	Probability
Construction						
Offshore facilities	⊙	habitat destruction (excavation works)	severe	short-term to medium-term	localized	likely
	⊙	increased turbidity may affect filter feeding organisms	notable	short-term	localized to mid range	possible
	⊙	re-sedimentation may blanket sessile epifauna	severe	short-term	localized to mid range	likely
	⊙	potential contamination by spills or leakages	severe	short-term to medium-term	localized	possible
	⊗	upon completion, structures may act as artificial reefs (attachment of sessile hard bottom species, attraction of reef-dwellers)	notable (positive)	long-term	localized	definite
Onshore facilities	⊙	potential burial by soils or other material washed into the sea	severe	short-term	localized	possible
Commissioning						
Waste streams	⊙	may be exposed to residual chemicals (if discharged)	severe	short-term	localized	possible
Operation						
Intake	⊙	open intakes cause entrainment of invertebrate larvae	notable	long-term	localized to mid range	definite
Concentrate discharge	⊖	increased salinity may be harmful or toxic to sessile species	severe	long-term	localized to mid range	definite
	⊙	increased salinity may result in avoidance of mobile species	notable	long-term	localized to mid range	likely
	⊖	toxic effects and avoidance can cause a change in species abundance and diversity in the discharge site (all effects depending on exposure levels and species sensitivity)	severe	long-term	localized to mid range	possible
	⊙	potential enrichment of pollutants in filter feeding organisms	notable	long-term	localized	possible
Residual chemicals in the concentrate	⊖	residual chlorine levels and chlorination by products may have toxic effects on organisms in the mixing zone	notable	long-term	localized to mid range	likely
	⊖	potential for metal bioaccumulation in filter-feeding and deposit-feeding benthic organisms, with the risk of biomagnification	severe	long-term	localized to far-range	possible
Backwash water	⊖	blanketing of sessile animals and ingestion of material by filter- and sediment feeders may occur	severe	long-term	localized to mid-range	likely if discharged
Chemical storage	⊙	potential exposure to harmful substances	severe	short-term	localized	unlikely
Energy use	⊙	secondary effects from cooling water use	notable	long-term	localized to mid-range	likely
Maintenance						
Cleaning solutions	⊙	residual cleaning chemicals may be harmful for sessile animals	severe	short-term	localized	likely

Table 32: Marine nekton – evaluation of impacts.

Life cycle stage		Effects on marine nekton	Intensity	Duration	Spatial extent	Probability
Construction						
Offshore facilities	⊙	construction may cause behavioral responses and temporary habitat loss due to sediment plumes, noise and vibrations, etc.	notable	short-term	localized to mid-range	likely
	⊙	increased turbidity may affect fish gills and re-settling of material may blanket fish spawn	notable	short-term	localized to mid-range	possible
	⊙	potential contamination by spills or leakages	severe	short-term	localized	possible
	⊗	upon completion, structures may attract species (reef effect), e.g., due to increased food supply	notable (positive)	long-term	localized	possible
Commissioning						
Waste streams	⊙	may be exposed to residual chemicals (if discharged)	severe	short-term	localized	possible
Operation						
Intake	⊙	open intakes cause entrainment of eggs, larvae, small juveniles	notable	long-term	localized to mid-range	definite
	⊙	open intakes may cause impingement of nektonic species	severe	long-term	localized	likely
Concentrate discharge	⊙	may avoid the discharge area, loss of potential feeding or breeding grounds	notable	long-term	localized to mid-range	likely
Residual chemicals in the concentrate	⊙	chlorinated-dechlorinated seawater may still have chronic effects due to the presence of chlorination by products	notable	long-term	localized	unlikely
	⊖	residual chlorine levels and chlorination by products may have toxic effects on organisms in the mixing zone	severe	long-term	localized to mid range	likely
Backwash water	⊙	coagulants are non-toxic, however, mobile animals may avoid the high turbidity discharge area and high levels of suspended matter may affect fish gills	notable	long-term / intermittent	localized to mid-range	likely if discharged
Chemical storage	⊙	potential exposure to harmful substances	severe	short-term	localized	unlikely
Energy use	⊙	when coastal power plant capacity increases: secondary effects from cooling water use (intake and discharge effects)	notable	long-term	localized to mid-range	likely
Maintenance						
Cleaning solutions	⊗	high or low pH values and residual cleaning chemicals may be harmful, but animals will probably avoid the discharge site	notable	short-term	localized	possible if discharged

Table 33: Marine mammals and reptiles – evaluation of impacts.

Life cycle stage		Effects on marine mammals and reptiles	Intensity	Duration	Spatial extent	Probability
Construction						
Offshore facilities	⊙	underwater construction may cause behavioral responses and temporary habitat loss due to sediment plumes, noise or vibrations	notable	short-term	localized to mid-range	likely
	⊙	haul-out sites of seals or nesting sites of turtles in the landing area may be affected	severe	short-term	localized	possible
	⊙	potential contamination by spills or leakages	severe	short-term to medium-term	localized	unlikely
	⊗	upon completion, structures may attract species (reef effect), e.g., due to increased food supply	notable (positive)	long-term	localized	possible
Onshore facilities	⊙	construction noise may cause a temporary loss of haul-out sites	severe	short-term	localized	possible
Commissioning						
Waste streams	⊙	may be exposed to residual chemicals (if discharged)	severe	short-term	localized	unlikely
Operation						
Intake	⊙	open intakes may cause impingement, e.g., of sea snakes or turtles	severe	long-term	localized	possible
Concentrate discharge	⊙	may avoid the discharge area, loss of potential feeding or breeding grounds	notable	long-term	localized to mid-range	likely
Backwash water	⊙	coagulants are non-toxic, however, mobile animals may avoid the high turbidity discharge area	notable	long-term / intermittent	localized to mid-range	likely
Chemical storage	⊙	potential exposure to harmful substances	severe	short-term	localized	unlikely
Noise emissions	⊖	may avoid the sites of increased noise levels, potential loss of haul-out sites	severe	long-term	localized to mid-range	likely
Energy use	⊙	when coastal power plant capacity increases: secondary effects from cooling water use (intake and discharge effects)	notable	long-term	localized to mid-range	possible
Maintenance						
Cleaning solutions	⊗	high or low pH values and residual cleaning chemicals such as biocides may be harmful, but animals will probably avoid the discharge site	notable	short-term	localized	unlikely

Table 34: Terrestrial birds and seabirds – evaluation of impacts.

Life cycle stage		Effects on terrestrial birds and seabirds	Intensity	Duration	Spatial extent	Probability
Construction						
Offshore facilities	⊙	construction may cause behavioral responses and temporary habitat loss due to sediment plumes, noise and vibrations, etc.	notable	short-term	localized to mid-range	likely
	⊙	nesting sites of seabirds or penguins in the landing area may be affected	severe	short-term	localized	possible
	⊙	potential contamination by spills or leakages	severe	short-term	localized	unlikely
	⊗	upon completion, structures may attract species (reef effect), e.g., due to increased food supply	notable (positive)	long-term	localized	possible
Onshore facilities	⊙	construction, e.g., through noise and vibrations, may cause behavioral responses and temporary habitat loss	notable	short-term	localized to mid-range	likely
Operation						
Concentrate discharge	⊙	may avoid the discharge area, loss of potential feeding or breeding grounds	severe	long-term	localized	likely
Backwash water	⊙	coagulants are non-toxic, however, mobile animals may avoid the high turbidity discharge area	notable	long-term / intermittent	localized to mid-range	likely
Chemical storage	⊙	potential exposure to harmful substances	severe	short-term	localized	unlikely
Noise emissions	⊖	may avoid the sites of increased noise levels, potential loss of feeding or nesting grounds	severe	long-term	localized to mid-range	likely

An overall significance rating was established for each identified environmental impact by aggregating the single ratings for intensity, space and time into one overall value (the definitions for intensity, space and time and the aggregation procedure are outlined in Table 20, page 107). The effects in the above Tables (Tables 21 to 34) which are highlighted in red (⊖) were found to be of 'high priority' for environmental impact assessment studies and for impact mitigation. The effects highlighted in green (⊗) were found to be of 'low priority'. The majority of potential impacts were rated as being of 'medium priority' (⊙), which means they require a further case-specific evaluation. Based on this evaluation, they may be upgraded into high priority impacts or downgraded into low priority impacts depending on project- and site-specific conditions.

5.3 Results and conclusions

Based on the ratings for intensity, space and time, the effects listed in Table 35 below were identified as being of high priority for environmental impact assessment studies and for impact mitigation.

Table 35: Environmental effects of high priority for impact mitigation.

Receptor	Environmental effects
▸ Landscape properties and natural scenery	▸ visual, aesthetic impacts due to the discharge of reddish-brown backwash water from media filters (specific to the reverse osmosis process) that may cause a discoloration of the water column in the mixing zone or may be transported to nearby beaches ▸ acoustic impacts caused by noise emissions from plant operation
▸ Air quality and climate	▸ any significant impairments of local air quality due to emissions of air pollutants (NO_X, SO_X, PM_{10}) ▸ significant emissions of carbon dioxide (CO_2) and other greenhouse gases
▸ Groundwater quality and hydrology	▸ any changes in flow directions and groundwater salinity ▸ any pollution from spills and seepage
▸ Marine sediments	▸ changed erosion and sedimentation patterns locally and in downdrift locations which may be caused by artificial breakwaters ▸ increases in pore water salinity which may be caused by the concentrate discharge ▸ the accumulation of coagulant material in sediments near the outlet potentially caused by the discharge of media filter backwash water ▸ the risk of heavy metal accumulation in sediments if present in the discharge, e.g., copper from corroding plant materials
▸ Seawater quality and hydrology	▸ significant changes in salinity and temperature in the mixing zone of the effluent plume ▸ sinking of the discharge plume and formation of a dense bottom water layer, which may have a strengthening effect on density stratification of the water column and which may impede reoxygenation of bottom waters ▸ increases in turbidity and decreases in light penetration in the mixing zone potentially caused by the filter backwash plume

continued on next page

Receptor	Environmental effects
▸ Terrestrial fauna and flora	▸ effects that may cause a long-term to permanent loss of habitat ▸ noise emissions that may scare away sensitive wildlife within acoustic range ▸ prominent features that could preclude linkages and movement corridors of wildlife, and which could strengthen the effect of habitat loss
▸ Benthic macrofauna and -flora	▸ salinity or temperature increases in the mixing zone that may cause a decline of algae stands and seagrass meadows, or that may be harmful to benthic invertebrate species, depending on exposure and species sensitivity ▸ any toxic effects of chemicals, e.g., from residual chlorine, chlorination by-products, or heavy metals, alone or in combination with other effects, e.g., synergetic effects between increased temperature and chlorine ▸ avoidance reactions, which may cause a lasting change in species abundance and diversity in the discharge site ▸ a harmful blanketing of sessile species potentially caused by the filter backwash plume
▸ Marine mammals, reptiles or bird species	▸ loss of haul-out sites, nesting grounds or important feeding grounds, for example caused by noise emissions and general disturbance within visible and acoustic range

The evaluation of significance allows project planners and regulators to focus on the most relevant environmental impacts, for which impact mitigation measures need to be developed and implemented. Impacts that are found to be significant have a high priority for impact mitigation and should either be prevented or minimized to levels that are less than significant. If an impact still remains significant after impact mitigation measures have been implemented, compensation is often required (chapter 7, page 156ff.).

The majority of potential environmental impacts of desalination plants as identified in Tables 21 to 34 were rated as being of 'medium priority'. This includes for example all construction-related impacts. Although these are usually severe in terms of intensity, the effects are generally temporary, localized and reversible. The classification as 'medium priority' does not imply that these effects do not need to be considered any further. 'Medium priority' effects might not be decisive for the EIA outcome, but they often require some form of impact mitigation as well. They need to be carefully scrutinized case-specifically, and may be upgraded into high priority impacts or downgraded into low priority impacts depending on project- and site-specific conditions. This underlines the necessity for a case by case evaluation, as part of a project-specific EIA study and using the criteria from this evaluation or other suitable criteria.

Whether or not an impact is rated significant depends on many factors, such as the project size and design, the sensitivity of the environment in the selected site, the availability of impact mitigation measures but also the perception and definition of significance. No universally valid standard for significance exists. It will vary according to different national, regional or local standards and environmental regulations, which depend on a society's or community's social, ideological and cultural values, on economic potentials, and on politics.

The discharge of the concentrate is a unique feature of desalination projects. The potential impacts caused by the discharge of the concentrate and measures to minimize these impacts are also a central issue in the design and development of most new desalination projects. In contrast, other effects, such as impingement and entrainment effects caused by the seawater intakes or the impacts associated with energy use, are similar in nature to other development projects, such as coastal power plants or industrial facilities with seawater cooling systems. Nevertheless, these other effects generally play an important role in the permitting process of new desalination projects, and may even be considered more paramount than the disposal of the concentrate and its potential impacts on the marine environment, depending on a society's values and perceptions.

In California, for example, the "most significant direct adverse environmental impacts of a desalination facility are likely to be caused by its intake" [279] due to entrainment and impingement of marine organisms. However, these impacts can be completely avoided or substantially reduced by using alternative intake designs. Subsurface intakes, such as beachwells or infiltration galleries, are alternatives to open intakes which eliminate impingement and entrainment impacts. Applicants are therefore "encouraged to use subsurface intakes whenever feasible", whereas projects proposing open seawater intakes should expect to provide information about their effects on marine organisms, and projects proposing to co-locate to a power plant should not assume that joint use of the cooling system is the best available alternative but should conduct the necessary study to determine whether subsurface intakes would be feasible [279].

In contrast, carbon dioxide emissions seem to be the paramount issue in all large Australian SWRO projects besides the disposal of the concentrate [e.g., Perth, Sydney, Melbourne, 140, 155, 156]. Different standards in different world regions can also be observed with regard to regulations concerning the backwash waters from the granular media filters or the cleaning solutions used for the SWRO membranes, both of which can either be discharged or treated (cf. section 3.3.3, page 71).

The evaluation method and approach presented here should therefore be understood as an attempt to prioritize impacts based on general criteria in order to provide a first indicative guidance by identifying aspects that will typically be of high priority for project- and site-specific EIA investigations, and that would typically require some form of impact mitigation. Impact mitigation measures and aspects of environmental impact assessment are covered in the following two chapters (chapters 6 and 7, respectively).

Impact mitigation measures

6

6.1 Introduction

As the use of desalination accelerates in many parts of the world (cf. chapter 1), the need for 'green' or 'clean' desalination technologies becomes evident. Both are synonyms for the application of environmental science and technology to conserve the natural environment and its resources and to curb the negative impacts of human involvement. Sustainable development is the core of 'green' technologies [280]. This chapter investigates if desalination can be considered as a sustainable and green technology, as recently claimed for some SWRO projects. It furthermore proposes an approach for developing 'best available technique' (BAT) solutions for seawater desalination projects. BAT, often used synonymously with the terms 'best available technology' or 'treatment', or sometimes also referred to as 'best practicable environmental option', aims at the identification of state of the art technologies which indicate the practical suitability for preventing or reducing pollution and impacts on the environment.

To that end, the main components of desalination plants (i.e., the intake, pretreatment, desalination process, cleaning, and concentrate disposal system as described in chapter 2) are being compared with regard to environmental criteria such as energy, material and chemical use, and resulting emissions and likely environmental impacts. The identified BAT solution can be used as a reference in the determination of individual BAT solutions on a case by case basis, taking site- and project-specific considerations into account. It should be noted here that the environmental impacts of a desalination plant will depend on the *technology and mode of operation* on the one hand, and the *environmental characteristics of the project site* on the other hand. This chapter aims at developing *technology-based* recommendations for mitigating the impacts of desalination projects. For completeness, general criteria for site selection are given in section 6.5.

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S. Lattemann, M.D. Kennedy and G. Amy. Best available techniques for seawater desalination. International Desalination Association (IDA) World Congress on Desalination and Water Reuse, Dubai, UAE, 2009.

S. Lattemann, M.S. Anarna, J.C. Schippers, M.D. Kennedy and G. Amy. Environmental impact assessments (EIA) and best available techniques (BAT) for membrane-based seawater desalination. International Water Association (IWA) Membrane Technology Conference and Exhibition, Beijing, China, 2009.

6.2 The concept of green technology

The Earth Day 2009 on April 22 marked the beginning of a new campaign called Green Generation^{TM}, which seeks to foster the development of green technologies and solutions to urgent global issues such as climate change or the world water crisis [281]. Energy, water and climate change are inseparable global problems. On the one hand, energy is needed to deliver water, and water is needed to generate energy. We are not only living in an era of 'peak oil' but also of 'peak water' [282]. Kuwait, for instance, has to deliberate whether it will sell its oil at record prices or hold more of it to generate freshwater through energy intensive desalination (cf. section 4.5 on page 100). On the other hand, energy use has caused an increase in atmospheric greenhouse gases to 37% above the pre-industrialization level, and research indicates stronger than expected forcing of climate change, which also affects the global water cycle. For instance, the Eastern and South-western parts of Australia have experienced substantial rainfall declines since 1950, which is assumed to be partly due to human-induced climate change, and models predict up to 20% more drought-months over most of Australia by 2030 [283]. The continent currently experiences one of the harshest draughts in its history and turns to desalination in order to alleviate problems of water scarcity in most of the major cities.

Seawater desalination is a technology that can mitigate the problems of water shortage, and analysts agree that capacities will continue to grow rapidly in the coming years [25, 284, cf. also chapter 1]. However, the question is whether desalination is also a green and sustainable solution? The main environmental concerns of desalination have been discussed in detail in the previous chapters and shall only be briefly exemplified and recapitulated here. The world's largest SWRO plant in Ashkelon, Israel, currently has a capacity of 330,000 m^3/d, which is the equivalent of 132 olympic-size swimming pools. It contains 27,000 membrane elements, which need to be replaced every 3 to 7 years, with a total active surface area of about 99 ha, which is the equivalent of 200 football fields. The energy demand attributed to the use of materials and construction, however, is low (in the range of 5%) compared to the energy demand of operation (3.6 kWh/m^3), which is comparable to the energy demand of 330,000 laundry dryer loads every day. Besides materials and energy, all desalination plants use chemicals. Their residues are discharged into the sea along with the concentrate (section 3.3) and may amount to thousands of tons per day in some sea areas (section 3.4). The discharges, and the intakes which may cause impingement and entrainment, are the main reasons why marine protection groups like the 'Surfrider Foundation' in California or the 'Clean Ocean Foundation' in Australia campaign against proposed desalination projects. The other main public concern is pollution of the atmosphere (chapter 4). For example, more than 75% of the formal submissions received in response to the environmental assessment of the proposed Sydney SWRO project expressed concerns about energy use and greenhouse gases [232].

A recent review in *Nature* described desalination as a water treatment technology that is often "chemically, energetically and operationally intensive, focused on large systems, and thus requiring considerable infusion of capital, engineering expertise and infrastructure". The costs as well as the environmental concerns are still an impediment to the widespread use of desalination technologies today [285]. Yet some project developers have recently made headlines with buzzwords such as 'green' [68, 264] or 'sustainability' [61, 69]. This suggests that desalination is a green and sustainable technology after all. In this regard opinions drift apart. As desalination capacities continue to grow, it remains even more necessary to gain an objective understanding of the resource consumption and environmental impact of desalination technologies.

If 'green technology' is the application of environmental science to conserve the natural environment and resources and to curb the negative impacts of human involvement, then 'green desalination' should consequently be:

- the implementation of standards to curb the use of natural resources, i.e., standards for best available techniques (BAT) in desalination (this chapter), and
- the application of environmental science to investigate and minimize impacts on the natural environment, i.e., the conduct of environmental impact assessment (EIA, chapter 7) and environmental monitoring studies (chapter 8).

Both approaches, EIA and BAT, legitimately coexist because the first aims at minimizing impacts at a site- and project-specific level, whereas the other is a technology-based approach. The United Nations Environment Programme released an EIA guidance manual for desalination projects in 2008 [17, cf. chapter 7]. A comparable reference on BAT for desalination, which describes a standard of state of the art desalination technologies that indicate the practical suitability for limiting resource consumption and waste products, is lacking so far.

6.3 The concept of best available techniques (BAT)

The concept of best available techniques (BAT) has been established by different legislative systems, e.g., in Europe and the United States, and has been applied to similar industries and applications, such as power plants and seawater cooling water systems. In Europe and neighboring seas, the concept of BAT is introduced by the EC Directive on Integrated Pollution Prevention and Control (IPPC), the Conventions for the Protection of the Marine Environment of the North-East Atlantic (OSPAR), of the Baltic Sea Area (HELCOM), and the Protocol for the Protection against Pollution from Land-Based Sources of the Mediterranean Action Plan (LBS protocol).

The IPPC Directive imposes a requirement for certain industries with a high pollution potential to obtain a permit, which is issued if certain environmental conditions are met, such as the use of BAT. This applies to industries listed in Annex I of the Directive, such as energy industries, but not water treatment installations. A reference document on BAT in Industrial Cooling Systems has been provided under the IPPC Directive [192], which could at least partly be used as a basis for thermal desalination plants.

The marine conventions, in contrast, clearly indicate the need for a BAT concept for seawater desalination technologies in European and neighboring sea areas. For example, the Mediterranean LBS protocol requires contracting parties to take BAT into account when adopting action plans, programmes and measures. In 2005, the countries of the Mediterranean Action Plan adopted the concept that desalination plants are "industrial facilities", which means that they need to be regulated and assessed through EIAs [286]. In addition, BAT standards for desalination plants should be developed and implemented through the Mediterranean Action Plan. In some countries, general BAT regulations exist already. In Israel for example, a prerequisite for discharge into the sea is the use of BAT, which prohibits discharge directly at the coast with the exception of cooling water outfalls of power plants. Marine outfalls therefore have to be deep offshore outfalls, and the entire length of the pipeline has to be buried using BAT during construction to minimize damage to the coastal area [80]. While the first large SWRO plant in Israel (Ashkelon) is co-located to a power plant and therefore discharges directly at the coastline, tender documents for new projects set a request for an outfall to a depth of 20 m [81].

According to the IPPC Directive and the marine conventions, BAT is defined as state of the art processes, facilities or methods of operation which indicate the practical suitability for limiting discharges, emissions and waste, and for reducing the impact on the environment as a whole. The term ‘technique’ includes both the technology used and the way in which the installation is designed, built, maintained, operated and dismantled. The techniques that are considered BAT should be economically and technically feasible, should be used or should have been tried out on an industrial scale, and should take technological advances in scientific knowledge into account. Special consideration in the development of BAT is typically given to the consumption of raw materials, water, energy, less hazardous substances, and the possibility for recovery and recycling of any resources used or wastes generated. Applying the IPPC principle to an industry through implementation of BAT implies the need to take preventive measures when it is suspected with foundation that an activity may cause harm to the environment even if there is no absolute proof (prevention principle), and to reduce the emissions into the atmosphere, water and soil, as well as waste generation (control principle).

In the United States, BAT terminology is used in the Clean Air Act and Clean Water Act (CWA). Under the CWA, the Environmental Protection Agency (EPA) issues national standards for facilities discharging directly to surface waters. These so called ‘effluent guidelines’ apply to categories of dischargers and are technology-based, and not on the impacts on the receiving waters. The intent of technology-based standards is to establish a basic national standard for all facilities within a category using BAT, which becomes the minimum regulatory requirement in permits that are implemented through the National Pollutant Discharge Elimination System (NPDES) permit programme [287].

Similar to the European IPPC Directive, guidelines have been established for different industrial categories including power plants but not water treatment installations. CWA and NPDES regulations authorize the use of ‘best professional judgment’ to derive technology-based effluent limitations on a case by case basis where standards are absent. Best professional judgment was for example used to derive the NPDES permit for the Carlsbad SWRO plant in Southern California [180]. The EPA has now initiated a new rulemaking on drinking water treatment effluent guidelines to address the direct discharge of drinking water treatment residuals to surface water, such as suspended solids, aluminum or iron salts, organic matter, polymers, *desalination concentrates* or other residuals [287]. Section 316(b) of the CWA furthermore requires that the location, design, construction and capacity of cooling water intake structures reflect the BAT for minimizing adverse environmental impact, that is the impingement and entrainment of fish and other aquatic organisms. The regulations on cooling water intakes could serve as a basis for regulating seawater desalination plant intakes as well.

6.4 BAT approach for seawater desalination plants

The different desalination processes have been sufficiently described in the literature and the state of the technology has recently been reviewed by two expert groups, the Committee on Advancing Desalination Technology of the U.S. National Research Council [5] and a Technical Working Group instituted by the World Health Organization [16]. A short technology overview has also been included in chapter 2 of this thesis, and in the European research project MEDINA [288], on which this chapter and its conclusions are based. Only the main conclusions will be reproduced here, which should be seen as a first approach to identify general BAT solutions for seawater desalination plants.

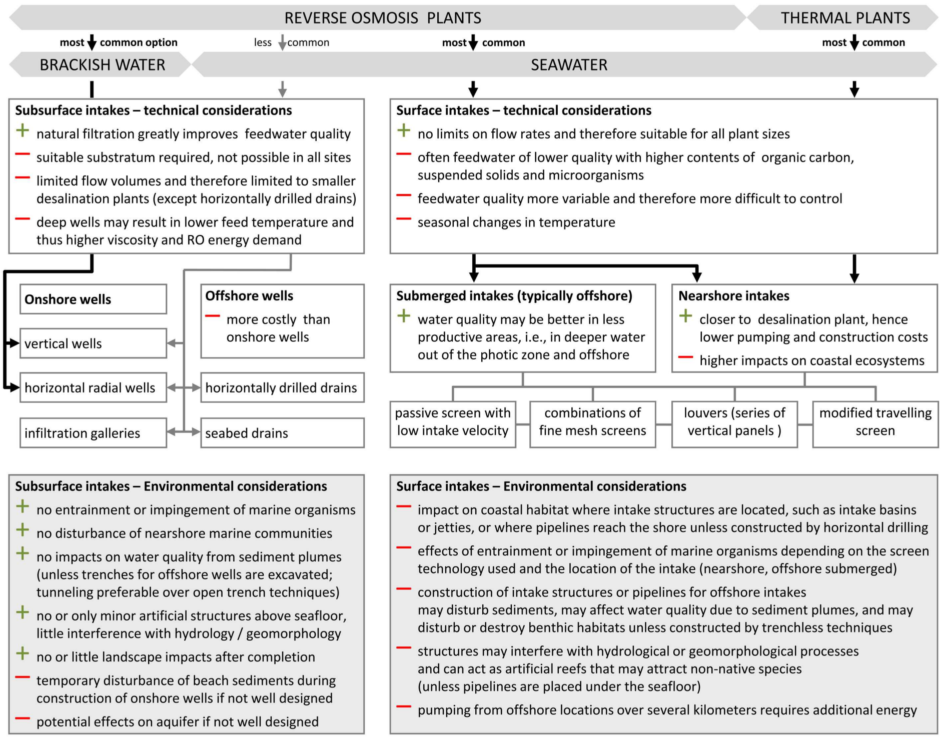

Figure 18: Intakes – considerations for best available techniques.

6.4.1 Intakes

A subsurface intake should be considered where a permeable substratum in the onshore or offshore sediments exists and where the chemical composition and quality of the water allows for its use in a desalination plant (Figure 18). Different types of subsurface intakes have been successfully used or tested in SWRO plants (cf. section 2.2).

A main environmental benefit of subsurface systems is that the impingement of marine organisms against intake screens and entrainment into the plant is avoided. As most or all parts of the subsurface intakes are located below ground, effects on landscape properties, seawater hydrology and sediment morphology are also minimal. These advantages must be balanced against the concerns, which are mainly the temporary disturbance of the seabed during construction and potential impacts on the aquifer if the intake is not well designed. Trenchless construction techniques such as horizontally directed drilling from an onshore location are preferable over excavation works, as trenchless techniques reduce the impact area to a comparatively small site on land and do not disturb nearshore habitats. Hydrogeological studies should ensure that no adverse impacts on the aquifer are to be expected from the subsurface intake. Furthermore, construction activities should be carried out according to best environmental practice. This may include time schedules to minimize disturbance of sensitive species that may be seasonally present in the area, the minimization of construction corridors, or measures to avoid soil and sediment erosion by water, wind and wave action where vegetation has been cleared.

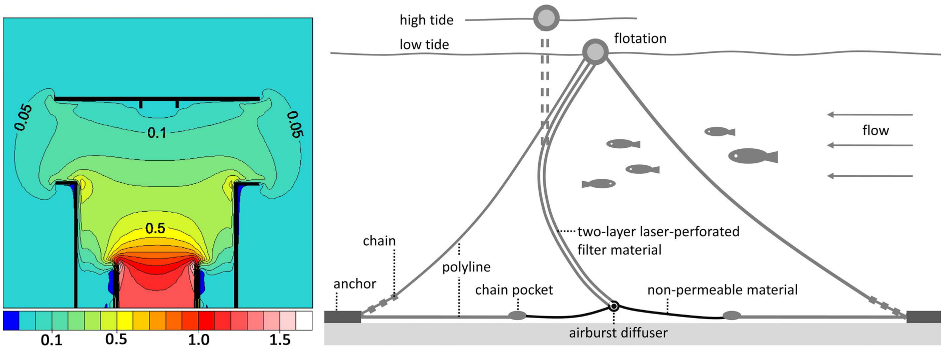

Figure 19: Intake riser of the Gold Coast desalination project in Australia [left, source: 178] with velocities in m/s and marine life exclusion system [right, adapted from 5, 289].

Where a subsurface intake is not feasible, BAT would be a submerged intake in an offshore location and located in deeper water. Mitigating entrainment effects is usually more difficult than mitigating impingement effects. Entrainment effects of smaller plankton organisms, eggs and larvae can mainly be minimized by positioning the intake in such areas where biological activity is low, typically a few hundred meters offshore and in several meters of water depth. Screens with a mesh size of 5 mm or less, which can be backflushed with compressed air, may additionally be used to exclude larger eggs, larvae, and juvenile fish from the intakes. In order to mitigate the effects of impingement, a single or several intake risers with a large surface area can be used in order to reduce the intake velocity, so that mobile animals will be able to swim away from the intake. The California Coastal Commission regards intake velocities <0.5 ft/s (0.15 m/s) to constitute BAT. Velocity caps, which are placed on top of the intake, can furthermore change the flow direction from vertical to horizontal, which enables fishes to better detect changes in currents [279, Figure 19, left]. Marine life exclusion systems (Figure 19, right) are currently being developed and tested. They form a water-permeable ‘curtain’ with a large surface area around the intake that results in a low flow velocity near the barrier that avoids both impingement and entrainment (cf. section 2.2).

Opportunities for co-locating SWRO plants with power plants should be investigated as an alternative to a submerged intake, as co-location may provide several benefits [162]. The total intake water is reduced when the power plant cooling water serves as feedwater to the desalination plant, which minimizes impacts from entrainment and impingement and the usage of certain chemicals such as biocides. It furthermore allows for mixing of the concentrate from the desalination process with the power plant cooling water, and reduces the overall construction and land use impacts. However, a major argument against co-location in California, for example, is that once through cooling (OTC) systems are not considered BAT. Hence, power plants are required to prepare plans to reduce impingement and entrainment, considering measures such as replacing OTC systems with recirculation cooling towers, dry- (air) cooling systems, or hybrid air-water cooling to virtually eliminate water withdrawal. However, these have lower efficiencies and higher energy consumption than wet evaporative cooling. For example, a new dry-cooling system is currently planned for the power plant to which the Carlsbad SWRO plant is going to be co-located, so that the existing power plant intake and outfall would be used solely for the SWRO plant in the future [290]. The desalination plant will therefore possibly require another environmental review when the power plant is shut down to ensure that the intake uses the best available measures to minimize marine life mortality [291].

Locating a SWRO plant in an estuarine site may also provide advantages, mainly because of the lower salinity of brackish water and hence the lower energy demand of the desalination process. This has to be balanced against the main disadvantage, which is the high turbidity and high NOM content of the feedwater, which will typically require a more chemically intensive pretreatment. Bank filtration should therefore be considered where feasible in estuarine sites. The Tampa SWRO plant (90,000 m^3/d), for example, which is located in Florida's largest open-water estuary and receives its water from the cooling water discharges of a co-located power plant, had to be retrofitted with a more robust pretreatment soon after its start-up due to serious fouling problems inside the plant. Another consideration is impingement and entrainment, as it is generally difficult to locate intakes offshore and in deep water within an estuary. The Thames SWRO plant (150,000 m^3/d) in East London, for example, withdraws water during low tide to obtain water with a low maximum salinity of 11. The intake is located on an existing pier, approximately 150 m offshore at a minimum submergence of 0.5 m. To minimize impingement, the intake velocity is limited to 0.15 m/s and an acoustic deterrent is used to scare fish from the area. To minimize entrainment of larger plankton, the intake is fitted with 3 mm mesh size screens [44]. Entrainment of plankton, however, is a minor ecological concern in estuaries, as a high natural mortality of the freshwater and marine plankton occurs at the halocline between the riverine and marine environment. The decay of these organisms, as well as terrestrial runoff, causes the high NOM content in estuaries.

6.4.2 Pretreatment

Reverse osmosis plants

Subsurface or submerged intakes, identified as BAT for intakes, typically produce a better feedwater quality and can therefore reduce the operational intensity of the engineered pretreatment system in the desalination plant (Figure 20, page 130). No or only little chemical pretreatment may be required, which reduces chemical use and waste emissions, and the risks involved in handling and transportation of hazardous chemicals. Moreover, a better feed quality may increase membrane cleaning intervals from several times per year to once every year or less, and may increase the life-time of the membranes which reduces material and energy consumption due to the replacement of membranes over the project life-time.

Where the feedwater is received from an open intake, UF pretreatment can be a suitable alternative to conventional pretreatment. Similar to a subsurface intake, UF can produce a consistently high water quality, as it provides a fixed barrier to particles and also removes some high molecular weight dissolved organics from the water. UF usually requires chemical pretreatment, regular backwashing, CEB and CIP. In order to implement UF successfully, the filtration time, the backwashing and CEB intervals need to be optimized. One option to postpone backwashing and CEB is by having additional pretreatment prior to UF. This may include natural systems such as beachwells, or engineered systems such as media filtration. Another option is by lowering the flux, i.e., by increasing the total membrane area and thus the capital investment. It would also be possible to operate without coagulant pretreatment, but with more frequent cleaning [114]. As it seems, a trade-off between chemical use in pretreatment and in CEB and CIP is necessary. Based on a comprehensive literature review (section 2.3.2), it is concluded that the use of pretreatment chemicals for continuous or intermittent disinfection and for in-line coagulation is common practice in many UF-SWRO systems. The current trend

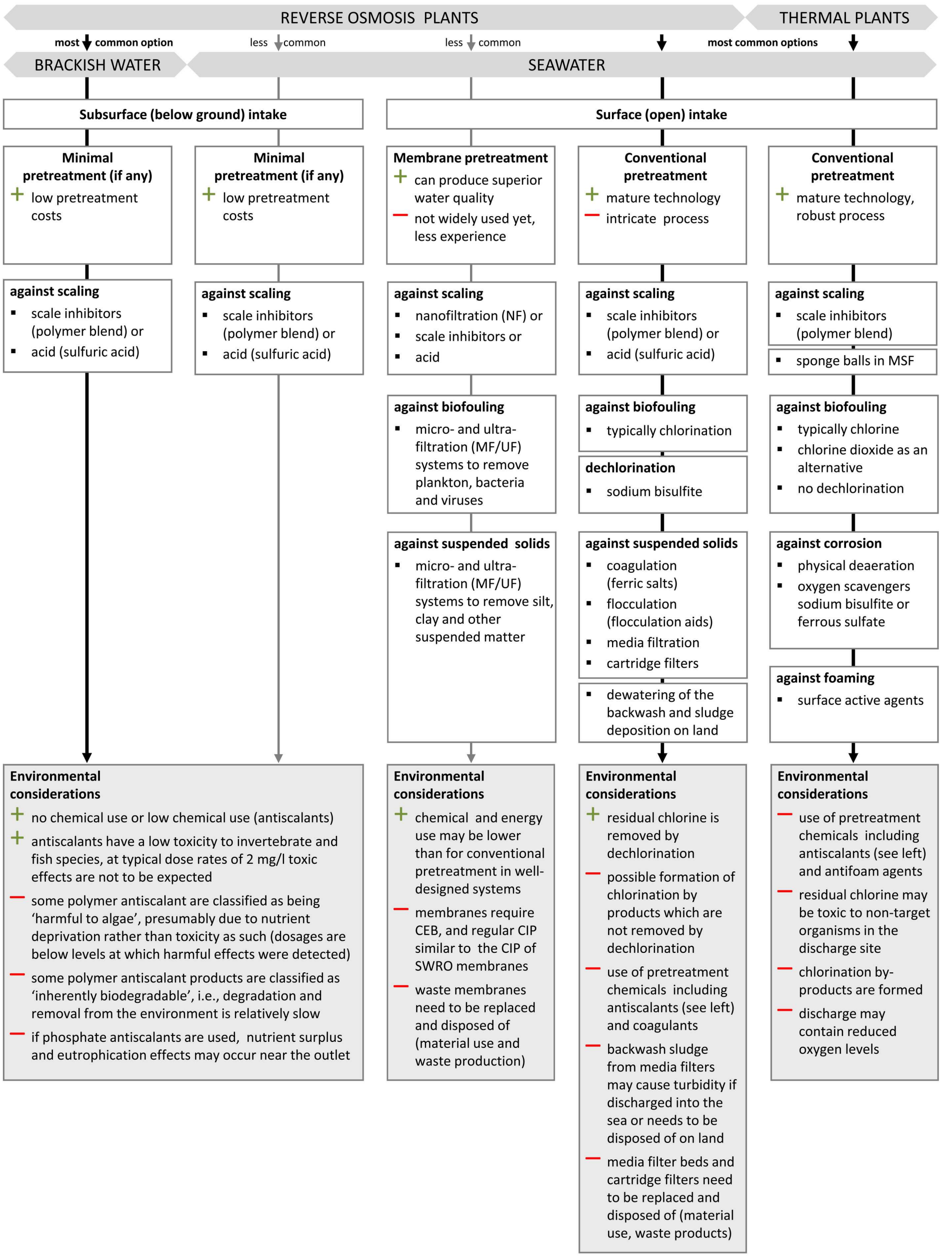

Figure 20: Pretreatment – considerations for best available techniques.

in UF-SWRO systems seems to favor more extensive pretreatment in order to postpone CEB of the UF membranes and CIP of the UF and SWRO membranes.

It has been reported that well-designed UF-SWRO systems operated on good raw water quality may save two RO cleans per year, which would reduce the total water cost of an UF system below that of a conventional system. The "occasional use of commodity chemicals is all that is required" in that case, with "much lower costs than the proprietary chemical cleaning regimes required for RO" [114]. With such a system, the main benefit stems from the reduced CIP frequency of the SWRO membranes, which may cause a reduction in the overall chemical use of the system. However, the UF pretreatment itself does not seem to live up to the expectations that are placed into the technology of being a low chemical process. Differences to conventional pretreatment seem to diminish if in-line coagulation, continuous chlorination and frequent CEB is employed in UF, as may be necessary in difficult waters.

As information on the actual chemical use in membrane pretreatment is partially inconsistent and incomplete, the situation remains inconclusive. Chemical use in UF similar to or higher than in conventional pretreatment has been reported in some occasions, but a low chemical approach is also practicable with UF. For example, only a few papers in the recent literature explicitly stated that coagulants are not needed in their specific UF systems. This was either attributed to operating the plant at a 50% lower UF flux [86], to a beachwell intake [118] or additional media filtration [92, 124] preceding the UF. Two of these papers reported results from pilot studies. More data from operational full-scale plants is needed to actually prove that UF can be used as a low chemical alternative to conventional pretreatment.

A general advantage of periodic chemical use, as in CEB and CIP, over continuous use in conventional pretreatment is that resulting waste streams can be collected and treated, which reduces chemical discharges into the sea. This, however, also holds true for conventional pretreatment designs where backwash waters from media filters are treated by dewatering and sludge deposition in a landfill, and where SWRO cleaning solutions are conveyed to the sewer or treatment facilities on-site. Discharge of side-streams, such as backwash waters or cleaning solutions, can not be considered BAT. Concerning the backwash waters, several levels of treatment may be required depending on the feedwater quality and the volumetric sludge production, including clarification, thickening and sludge dewatering (section 3.3.3). The remaining chemicals, which are discharged into the sea, are the same in both UF and conventional pretreatment. These are antiscalants (if used) and possibly chlorination by-products formed during chlorination. Both conventional and UF pretreatment can therefore be considered as BAT if the backwash waters from media filters, the CEB solutions, and the SWRO membrane cleaning solutions are treated.

Antiscalants are typically polymeric organic substances with a low toxicity to fish and invertebrate species. Some commercial antiscalants have been classified as 'inherently biodegradable' and as 'harmful' to algae (section 3.3.4). If available, antiscalants which are 'readily biodegradable' should be used to ensure a rapid removal from surface waters by degradation processes. However, phosphate-based antiscalants that decompose rapidly to orthophosphate and may thereby cause a nutrient surplus should likewise be avoided. If possible, simple acid treatment may be preferred over polymer antiscalants. A slight acidity of the effluent is effectively neutralized by mixing with ambient seawater following discharge, as seawater is slightly alkaline and has a good buffering capacity.

Where possible, a lower recovery rate of the desalination process might be considered to lower the risk of scale formation. However, this will increase the required intake flow rates of the process and should be carefully balanced against potential negative effects, such as the increased energy use, the increased use of other pretreatment chemicals such as chlorine or coagulants, or more severe entrainment effects. The consideration of a lower recovery rate may be worthwhile where other impacts have already been eliminated, e.g., by using subsurface intakes with no or very little chemical pretreatment and no entrainment and impingement effects, or by compensating energy use by renewable energies. A small-scale but exemplary design is the Enercon Desalination System [292], which uses a low recovery rate to completely avoid the use of antiscalants and any other chemicals. The EDS system can furthermore be coupled with a wind energy converter, and the manufacturer recommends the use of beachwells where feasible.

The question is if polymer antiscalants are actually needed in SWRO systems. Laboratory studies indicate that the induction time of calcium carbonate, the main scalant in SWRO, is about 100 minutes, which suggests that scaling will not occur in SWRO systems with a residence time in the membrane units of only a few minutes [56]. This result needs to be verified by pilot studies and in full scale SWRO plants.

Distillation plants

Similar to SWRO, distillation plants primarily use polymer antiscalants to prevent scale formation. Alternatives to this treatment are limited. Acid use would increase the risk of corrosion of the copper-nickel alloys, which are used as heat exchanger materials inside the plants. As antiscalants have to be stable at the high operating temperatures inside distillation plants, it is questionable if readily biodegradable substances, which might also be less chemically stable, can be used. Due to the lower temperature in MED, the saturation limits of calcium sulphate scales are not exceeded, which might be an advantage of the MED process over MSF plants in terms of antiscalant use. However, the mechanical removal of scale deposits with sponge balls, which are frequently used in MSF systems, cannot be employed in MED because the scales form on the outside (seawater side) of tubes (section 2.3.3, page 44). This places emphasis on the prevention of scales by chemical pretreatment and removal by chemical cleaning in MED. A general BAT recommendation cannot be given. When designing a specific project, antiscalant use should therefore be minimized as far as possible, based on laboratory and/or pilot testing to establish the lowest feasible dosage.

The use of chlorine in distillation plants is generally comparable to chlorination practices in coastal power plants with OTC water systems (section 3.3.2). The environmental assessment of seawater chlorination comprises the free oxidants, which have a high toxicity even at low concentrations, but which decompose quickly and do not bioaccumulate. Their harmful effects will therefore be limited to the mixing zone of the discharge plume. The by-products, which are formed by reactions with dissolved organic seawater constituents, are generally more persistent, some tend to bioaccumulate, and show a chronic mutagenic and carcinogenic toxicity [205, see also page 71]. Substitution of chlorine, minimization of chlorine use and treatment of residual levels have all been proposed and considered in seawater applications.

Where chlorine is necessary to control marine growth, the best approach to minimize adverse side-effects seems to be a low-level or pulse chlorination approach. Chlorine doses should be established based on site-specific and seasonal conditions. Lower doses also result in lower discharge levels, which can be deactivated by dechlorination if necessary. Most modern SWRO plants use intermittent chlorination at the intakes, and free chlorine is removed ahead of the RO membranes with sodium bisulfite to prevent the oxidation of the membranes. Sodium bisulfite, however, may cause oxygen depletion, if overdosed, which may be detrimental to marine life. Sulphur dioxide and hydrogen peroxide have been suggested to treat distillation plant reject waters [185]. Dechlorination before discharge may also reduce the potential for by-product formation. However, laboratory studies investigating the toxicity of chlorinated-dechlorinated seawater observed increased mortality [206, 207] and chronic effects [208] of test species even in dechlorinated seawater, which were assumed to be due to the presence of halogenated organic by-products formed during chlorination (page 67).

In addition to reducing the chlorine load that is discharged into the sea, outfall siting and design should ensure that the extent of the mixing zone is minimized by maximizing dilution, and that sensitive coastal ecosystems are not impacted by the dispersing effluent plume. The whole effluent toxicity of the desalination plant discharge should be analyzed, to take effects from residual chemicals, by-products, and synergetic effects with increased temperature and salinity into account.

Despite the fact that chlorine use has caught the attention of environmental authorities for several decades, it still remains the most common method for the control of marine growth in seawater applications, especially where large water quantities are needed such as in power plant cooling systems or desalination plants. Low-level chlorination is still considered BAT for these systems according to the EU reference document on industrial cooling water systems [192]. A review of the reference document has been announced to start in 2009. The pressure to reduce chlorine use is particularly high because of the large quantities of water and by-products that these systems discharge into water bodies and the environmental and health issues attributed to residual chlorine and disinfection by-products. In some regions, chlorine use has been banned because of its adverse side-effects, as for example in Venice Lagoon [196]. In freshwater cooling systems, continuous chlorinating is also not considered BAT [192] and other biocides, for instance on the basis of peracetic acid or DBNPA (page 70), are in use. In seawater, the viability of alternatives to chlorination remains the subject of continuing review. Several alternative pretreatment methods have been put forward, but none of these has gained acceptance over chlorine use so far (page 68), however, there seems to be a growing interest in the application of chlorine dioxide (page 69). If promising alternatives are more effective and less toxic than chlorine yet needs to be proven in a systematic study.

6.4.3 Cleaning and maintenance

Chemical cleaning is a necessity in all desalination plants. Plant operators have an intrinsic interest in maximizing cleaning intervals in order to reduce costs and increase the life-time of the SWRO membranes, and hence to implement BAT. This generally includes (i) obtaining feedwater of best possible quality, e.g., from a beachwell or an offshore location, and (ii) selecting the best pretreatment technology to reduce fouling and scaling and thereby increase cleaning intervals. Both strategies reduce the amount of cleaning chemicals needed and wastes generated.

If cleaning is required though, less hazardous solutions should be used where these are sufficient to remove deposits, i.e., gentle solutions with pH values of 4 or 10 instead of harsh solutions with pH values of 2 or 12. Solutions with no additional cleaning chemicals are preferable over chemical ‘cocktails’ that often include complexing agents, detergents or biocides (section 3.3.6). For instance, citric acid has complexing properties and is effective against scales, metal oxides and inorganic colloids. It is preferable over EDTA, an organic complexant which is poorly degradable.

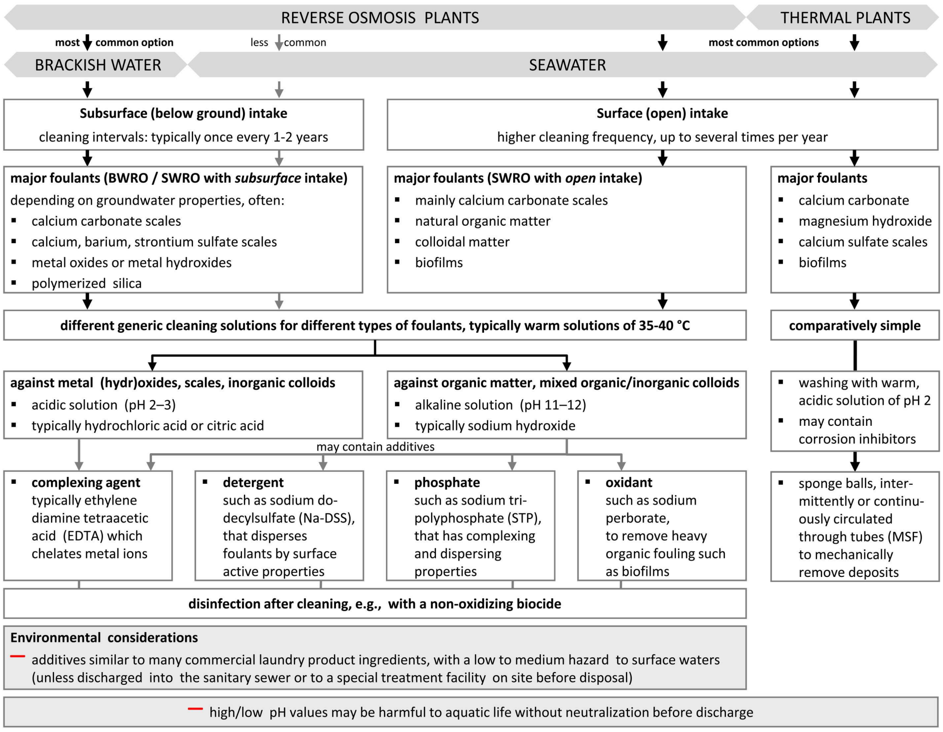

Figure 21: Cleaning and maintenance – considerations for best available techniques.

Finally, spent cleaning solutions should be recovered and discharged into the sewer or into special tanks on site for treatment in order to remove any residual toxicity before discharge into surface waters. Direct blow down or blending with the concentrate cannot be considered as BAT, and was for example prohibited by the NPDES permit for the Carlsbad SWRO project in Southern California [180].

6.4.4 Desalination process

Energy use

Chemical use mainly depends on the pretreatment process, whereas energy use mainly depends on the desalination process. MED and MSF plants have a thermal energy demand of 145-390 MJ/m^3 and 250-330 MJ/m^3, respectively, and an electrical energy demand of 1.5-2.5 kWh/m^3 and 3-5 kWh/m^3 (Table 8, page 46). SWRO plants, on the contrary, only require about 3-4 kWh/m^3 of electrical energy and hence have a significantly lower overall energy demand than distillation plants. In a comparative LCA of different desalination processes [147–149] it was concluded that the environmental 'load' of SWRO is one order of magnitude lower than the 'load' of thermal processes if these are operated with a conventional boiler, but comparable if the thermal processes are entirely driven by low value heat. MED was found to be more energy efficient than MSF and can also be more efficient than SWRO if low value heat is used (section 2.3.4, page 46).

All desalination processes are energy-intensive water supply options if compared to conventional water sources (Table 8, page 46), but desalinated water is not a more energy-intensive product than other amenities of modern life-styles, such as air conditioning

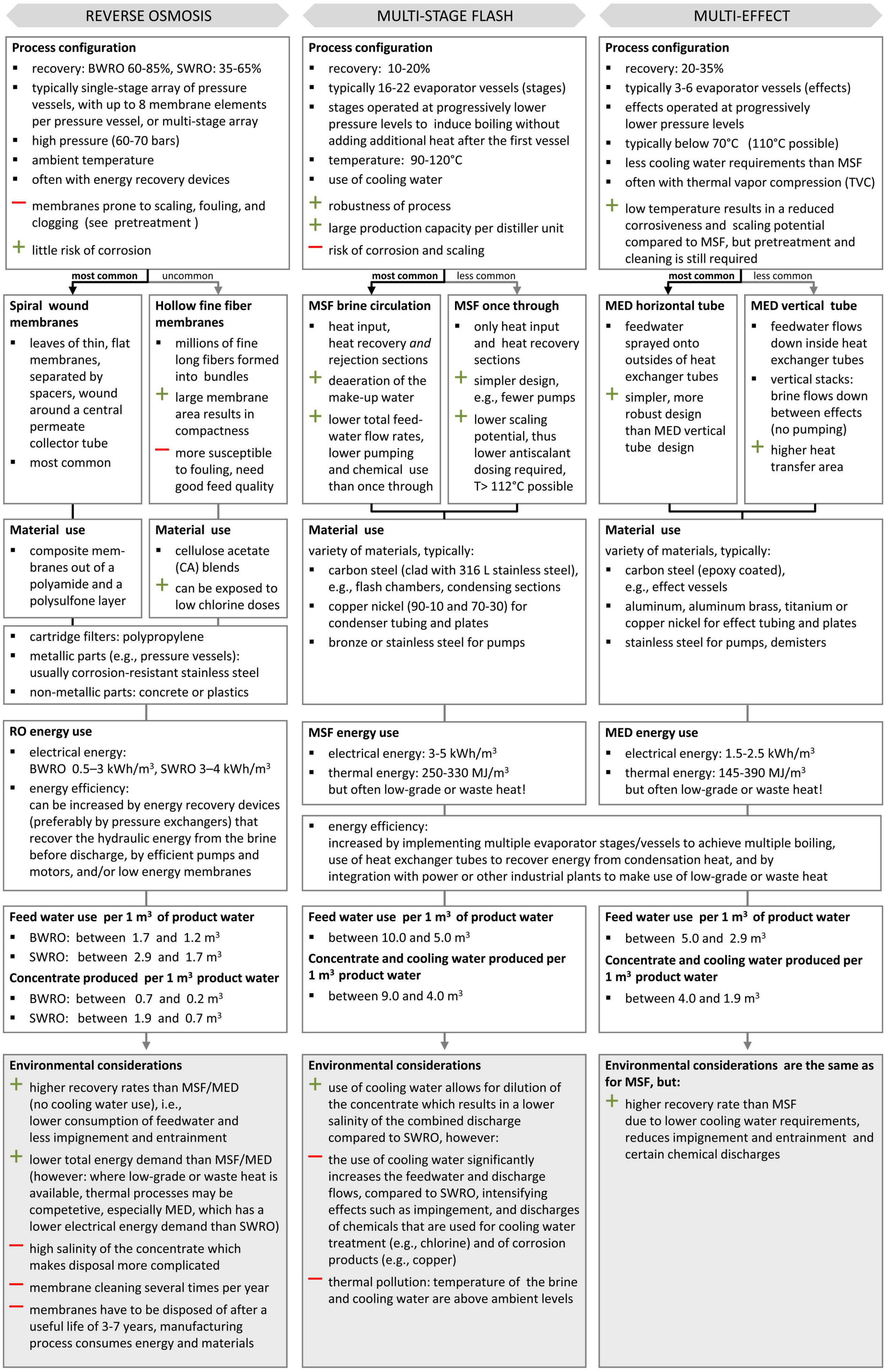

Figure 22: Desalination process – considerations for best available techniques.

or hot water production (section 4.2, page 91). More energy-efficient technologies and more rational energy use is necessary in all sectors of use to mitigate the impacts of climate change. In desalination plants, the challenge is to minimize the overall energy consumption of the process whilst maintaining a good performance of the pretreatment process, meeting the specified product water requirements, and maintaining a high level of protection for the local environment.

Thermal distillation processes, although requiring more energy, are still the first choice for desalination processes in the Middle East. Dual-purpose cogeneration facilities, which integrate MSF or MED distillation with power generation, significantly improve the overall thermodynamic efficiency by using low grade heat. While electricity is produced by high-pressure steam of about 540°C, the desalination process requires steam of maximal 120°C. This can be extracted from the low pressure/temperature end of the steam turbine, after much of the energy has been used to generate electricity. Fuel consumption of co-generation plants is significantly lower than the fuel needs of two separate plants [21]. However, the energy for the desalination process is not a 'waste' product in this case, as the extracted steam could have been used for additional electricity production [138]. Only where low grade heat would be wasted, e.g., heat from diesel generators on ships, thermal processes are energy-competitive with SWRO.

Waste heat is typically released into the atmosphere or into the sea as cooling water discharges. Reducing the amount of heat to be dissipated results in a lower environmental impact of the cooling system. OTC systems are commonly applied to large capacity installations in locations where sufficient cooling water and receiving surface water are available (e.g., coastal based power plants or desalination plants). If a sufficient water source is not available (e.g., in rivers) recirculating systems (cooling towers) are used. A limited water source could also be in a shallow coastal area with low mixing, such as in lagoons or barrier island systems. Replacing OTC systems in power or desalination plants by recirculating cooling systems can reduce the intake of surface water, and hence impingement and entrainment effects, as well as the discharge of large amounts of warm cooling water and chemicals into the sea. However, a change from a OTC to a recirculating system means an increase in energy consumption for auxiliaries, as well as a decrease of efficiency in the thermal cycle [192].

BAT in SWRO plants to minimize energy demand includes pressure exchangers and variable frequency pumps, besides optimizing the process as a whole. Energy savings in a well designed and optimized system are estimated to be about 1 kWh/m^3. Approximately two thirds of the reduction can be achieved by the right equipment, and the rest by optimizing plant operation between best (new membranes, low fouling, etc.) and worst operating conditions (minimum temperature, fouled membranes, etc.) [53]. Another possibility is to identify external opportunities for minimizing energy use, for example, by selecting power plant cooling water with a higher water temperature as feed [162].

A specific energy demand of less than 2.5 kWh/m^3 of a full-scale SWRO plant can be considered as very energy efficient. The Perth SWRO plant achieved the lowest ever reported value of 2.3 kWh/m^3 (3.2-3.5 kWh/m^3 for the total plant) for a large-scale, two-pass SWRO plant, using a 97% efficient energy recovery system [72]. Although the process of SWRO has already been dramatically transformed in the last decade, with the energy demand cut in half by highly efficient energy recovery devices and improved membranes, a further reduction in energy demand seems likely in the future. The practical upper limit of energy savings in SWRO is estimated to be about 15%, with improvements in module design having the greatest potential [5].

The assumption of 15% is based on a system operating at 40% recovery. Most SWRO plants today operate between 40-50% recovery (Table 5, page 30). In section 6.4.2 on pretreatment, it was proposed to lower the recovery in order to avoid or reduce antiscalant use. This has to be balanced against a higher specific energy demand of the process, which increases as the recovery rate decreases. Most SWRO projects are designed to maximize the recovery rate in order to minimize energy use. A second effect of a higher recovery is that a lower feedwater flow is required to produce the same amount of product water, which results in a lower energy use for pumping and pretreatment, and probably also a lower chemical use in pretreatment. Only the risk of scaling inside the plant increases with the recovery rate, so a low recovery rate is beneficial in terms of antiscalant use. This exemplifies the trade-offs that may be necessary for identifying the overall BAT solution for a SWRO plant. A low recovery approach to reduce antiscalant use is ecologically worthwhile where it is combined with other BAT and impact mitigation measures. For example, if the water is taken from a subsurface intake, flow rates can be increased without raising concerns about impingement and entrainment, and little or no pretreatment is usually required. The increased energy use could be offset through climate-protecting measures.

A low energy demand is also in the interest of plant operators, as is a good plant performance and product water quality. However, environmental protection measures, such as sludge treatment facilities or a diffuser system, may increase energy use and hence the water costs. Also, requirements to offset the negative impacts of energy use through climate-protecting measures may add to the costs of a project. These measures may range from the use of BAT to cut emissions and strip pollutants from off-gas at the power plant, to the purchase of renewable energy certificates or the implementation of renewable energy or reforestation projects. The expected increase in water cost is the price for a green, sustainable solution, which some communities have decided to pay. For example, the energy demand of the Perth and Sydney projects is compensated by newly erected wind farms [61, 68, see also section 4.5, page 99].

Water use

The SWRO process is characterized by a lower consumption of source water per cubic meter of product water and consequently a lower volume of concentrate discharges into the sea than distillation processes, which have large cooling water requirements (section 2.3.5). Lower feedwater requirements of SWRO result in a lower consumption of some, but not all pretreatment chemicals, depending on the dosing point inside the plant. Chlorine is usually added at the intake. For distillation plants, this means that the entire intake flow is chlorinated (cooling water and make-up water), whereas other chemicals such as polymer antiscalants and antifoam agents are added to the make-up only. Consequently, a SWRO plant operated at 33% recovery and a MSF distillation plant operated at 10% recovery in the Arabian Gulf region would use similar amounts of antiscalants, but chlorine use would be much higher in the distillation plant. For 1 m^3 of product water, the SWRO plant treats 3 m^3 of feedwater with antiscalant (i.e., the entire flow), while the MSF plant only treats the make-up water with antiscalant, i.e., antiscalant is added to about one third of the total feed water.

Material use

The results of a comparative life cycle assessment of different state of the art desalination process [147–149] indicate that material use and disposal has little influence on the overall environmental burden compared to plant operation due to the high energy demand of

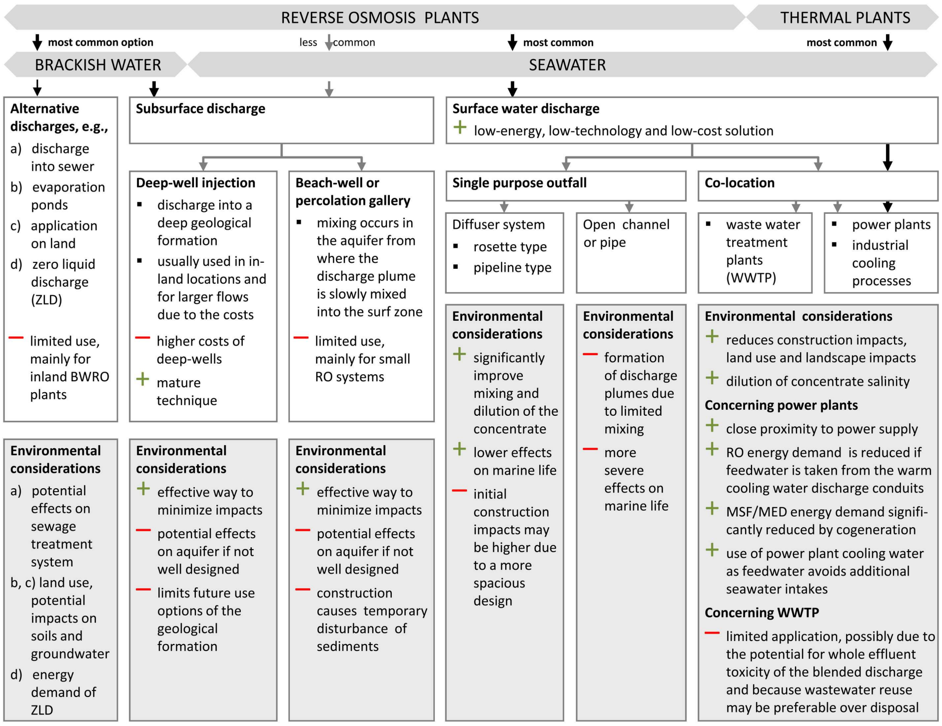

Figure 23: Outfalls – considerations for best available techniques.

all desalination processes. In MSF and MED plants, metallic parts prevail and recyclability of these materials after replacement or demolition can be assumed to be good. A main problem with SWRO membranes is that recycling of materials is not possible due to the composite nature of most modern membranes (section 2.3.6). However, a disadvantage of the copper-nickel alloys frequently used in distillation plants is their liability to corrosion, which may result in increased metal discharges into the sea (section 3.3.5).

6.4.5 Disposal of the concentrate

When considering the potential impacts of the waste discharges onto the marine environment, one has to distinguish between the salt and the chemical additives. Strict salinity thresholds have been established for discharges from desalination plants, e.g., in Australia, because even minor salinity increases may have a harmful though localized effect on marine ecosystems (section 3.3.1). The total salt load, however, is not a concern for sea areas which exchange water with the open ocean (this includes the semi-enclosed Mediterranean Sea, the Red Sea or the Gulf, section 3.4). Moreover, natural evaporation exceeds any salinity increase possibly caused by desalination plants in subtropical and tropical regions by several orders of magnitude. The key to avoiding impacts of salinity is to sufficiently dilute and disperse the salinity load to ambient concentrations. The same argument, however, does not necessarily hold true for the chemical additives. While the salt is of natural origin, the additives are of anthropogenic origin, and some have a tendency to accumulate in the environment. Dilution is therefore a questionable means of impact mitigation for certain chemical additives.

There are several BAT options to mitigate the environmental effects of the concentrate and chemical discharges. Mixing and dispersal of the salinity load can be enhanced by installing a multi-port diffuser system (Figure 10, section 2.4), and by locating the discharge in a favorable oceanographic site which dissipates the salinity load and heat quickly (section 6.5). To avoid impacts from high salinity, the concentrate can also be pre-diluted with power plant cooling water. To avoid impacts from high temperature, the outfall should achieve maximum heat dissipation from the waste stream to the atmosphere before entering the water body and maximum dilution following discharge. The spreading and dispersal of the plume in a given project site should be investigated by hydrodynamic modeling studies, accompanied by monitoring in the mixing area.

Negative impacts from chemicals can be minimized by substitution of hazardous substances by less harmful compounds, by using intake and pretreatment options with lower or no chemical uses, and by treatment of waste waters before discharge. Especially biocides such as chlorine, which may be harmful to non-target organisms in the discharge site, should be avoided or their use minimized, and residual levels treated prior to discharge. Chlorine can be effectively removed by different chemicals, such as sodium bisulfite as practiced in RO plants. Filter backwash water should be treated by dewatering and land-deposition where possible, while cleaning solutions should be treated on site in special treatment facilities or discharged into a sanitary sewer system. The BAT solution for concentrate discharge is closely interrelated with that for pretreatment and cleaning, which should aim at (i) the avoidance or minimization of chemical use, or (ii) the treatment of chemicals where avoidance is not possible.

6.5 Site selection

Identifying a suitable project-site is one efficient way of keeping the impacts of a proposed desalination project on its environment at a minimum. The process to identify sites often includes at least two selection rounds. In a first step, criteria for site selection are established and several preliminary sites are identified. The criteria for site selection can be subdivided into groups of criteria, distinguishing between non-negotiable criteria (e.g., compulsory or exclusion criteria) and criteria which may be balanced against each other (e.g., allowing trade-offs). A group of experts who are capable of making a judgment about the various technical and environmental issues then score the different sites on the various criteria. The whole process can be formally supported by a multi-criteria analysis (MCA) using commercial software packages (cf. chapter 9). For example, MCA has been extensively used for site selection in the planning phase of desalination feasibility studies in Australia [64], and also for two smaller plants in South Africa [293, 294]. A list of potentially relevant criteria for site selection is given in Table 36 on page 140.

The second selection round is the EIA, which is carried out for the most suitable site and possibly one or two alternatives, which have the highest acceptance and no obvious environmental or social constraints or other reason for exclusion. MCA methods can again be used in the EIA to facilitate decision making. MCA is commonly used in EIAs for infrastructure projects in The Netherlands [295] or the UK [296], such as for motor- and railway development projects, waste storage or treatment facilities, river and fresh water basin developments.

Table 36: Criteria for site selection of desalination projects [after 17].

Criteria	If possible, the selected site(s) should:
▸ Geologic and land area requirements	▸ provide stable geologic conditions, with no risk of affecting the stability of soils and sediment, or buildings and pipelines. ▸ be planar or easily allow for initial earthwork activities (site grading, excavation) or the laying of below-ground intakes, outfalls and pipelines. ▸ where relevant, have a permeable substratum that allows for the use of beachwells, infiltration galleries or horizontally drilled drains as intakes. ▸ be sufficiently elevated above sea level with no risk of flooding. ▸ be able to accommodate the intakes and outfalls and all facilities of the plant in terms of area size and geometry. ▸ have no risk of aquifer pollution in the case of spills and seepage.
▸ Biologic resources	▸ be devoid of ecosystems or habitats that are: • unique within a region (e.g., riffs on a mainly sandy shoreline). • worth protecting on a global scale (e.g., coral reefs, mangroves). • important in terms of productivity or biodiversity. • inhabited by protected, endangered, rare species, even if temporarily. • important feeding grounds or reproductive areas for a larger number of species or certain key species within a region. • important for human food production.
▸ Oceano-graphic conditions	▸ provide sufficient capacity to dilute and disperse the salt concentrate, as well as any residual chemicals discharged along with the waste water. In this regard, provide sufficient water circulation and exchange rate as a function of currents, tides, surf, water depth and bottom/shoreline morphology. In general, exposed rocky or sandy shorelines with strong currents and surf may be preferred over shallow, sheltered sites with limited water exchange.
▸ Concentrate discharge area	▸ be close to the concentrate disposal area to avoid pumping and to minimize the risk of land and groundwater contamination from pipelines. ▸ provide a discharge area that is located in sufficient distance from the intake or that is separated from the intake by natural or artificial features (headlands, jetties) in order to avoid recirculation of the waste.
▸ Proximity to consumers	▸ be close to existing distribution networks and consumers to avoid construction and land-use of pipelines and pumping efforts for water distribution. However, impairment of communities by visual effects, noise, air pollution or other environmental health concerns should be avoided.
▸ Proximity to energy supply	▸ be close to the power grid for SWRO plants. ▸ provide access to low-cost heat for distillation plants.

continued on next page

Criteria	If possible, the selected site(s) should:
▸ Other infrastructure	▸ allow for easy connection to other relevant infrastructure, such as access roads or communication networks. ▸ be co-located to power plants to make use of: • the existing intake/outfall structures (no new construction impacts). • the cooling water, resulting in a reduced energy demand of the SWRO process because of a higher membrane permeability at higher water temperature; a lower feedwater intake than for two separate plants with lower impingement/entrainment effects; and a lower discharge salinity if the concentrate is blended into the cooling water.
▸ Raw water quality and proximity	▸ facilitate an intake location that provides a good and reliable water quality, taking seasonal changes into account, with minimum danger of pollution or contamination, in order to avoid performance problems of the plant or impacts on product water quality. ▸ be close to the sea to minimize land use for pipelines and to avoid passage of pipes through agricultural land, settlements, etc. However, in some cases it may be more appropriate to locate the plant further inland, e.g., when construction on the shore is not possible for certain reasons (e.g., use of beaches, nature reserves, geological instability, etc.).
▸ Regional planning	▸ be classified as an industrial area or designed for industrial development in conformity with regional and land area plans. ▸ have the acceptance of neighboring communities and provide as little conflict as possible with other existing or planned uses and activities, especially recreational uses, commercial uses including shipping, nature conservation efforts, or cultural resources. • Recreational conflicts may occur if the project has the potential to reduce the recreational value of the area for residents or tourists by changing the natural scenery through emissions of noise, glare, etc., or by restricting access to beaches, hiking trails, fishing sites, etc. • Commercial conflicts may occur if the project is to be located within existing urban boundaries, where it could reduce the price for land or the value of adjacent residential properties, or if it interferes with maritime structures, navigation, access to harbors or other marine activities like commercial fishing or aquaculture. • Nature conservation conflicts may occur if the project significantly reduces the ecological value of the project site as a habitat for terrestrial and marine species. The decision to protect or open an area for development should therefore consider the presence or absence of rare and endangered species or biological communities. By changing the ecological value of a site, it may loose its present protection status or may no longer be eligible for becoming a protected area in the future. • Archaeological conflicts may occur if archaeological, paleontological or human remains are located in or near the project site, which may be accidentally uncovered or disturbed during construction.

6.6 Summary and conclusions

BAT and EIA are two complementary approaches. BAT identifies suitable processes at the technology level, which can facilitate the identification of individual BAT solutions at a project- and site-specific level through EIA studies. Both approaches have sustainable development at their core. A road map for the identification of individual BAT solutions for desalination projects is outlined in Figure 24, in combination with Figures 18 to 23 which summarize the main process alternatives and environmental considerations for each processing stage. According to the general concept and definition of BAT, it is proposed to consider the following order of measures when determining individual BAT solutions for desalination projects. In practice, the decision for a certain technology and design will also have to take site- and project-specific considerations into account.

- Selection of the desalination process with the highest energy use efficiency.
- Optimization of energy and water use efficiency of that process.
- Lowering the chemical use of that process by
  - reducing the occurrence of fouling and corrosion through process design (i.e., intake design and location) and thus minimizing cleaning and pretreatment requirements,
  - giving preference to no or low chemical respectively no or low waste designs,
  - substitution of harmful substances with less harmful substances,
  - optimizing the application and dosage of pretreatment chemicals based on pilot testing and/or monitoring of the feedwater quality,
  - treatment of wastes before discharge / disposal.
- Selection of manufacturing materials that can be reused or recycled, and identification of appropriate waste disposal options at the end of their useful life.

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If it would be possible to choose freely between the different process options, leaving out technical, economical and site-specific environmental limitations and taking only environmental benefits into account, the most preferred design would be a SWRO plant with a subsurface intake and enhanced multi-port diffuser design in a suitable oceanic site. A subsurface intake completely avoids impingement and entrainment of marine organisms and, as a biofiltration process, can potentially provide a consistently high feed water quality with advantages for (i) pretreatment, (ii) cleaning and (iii) membrane life, hence reducing resource consumption in various ways:

- As beachwells are biofilters which can reduce both organic and biofouling, further pretreatment after a beachwell is often minimal. The conventional steps of chlorination-dechlorination and coagulation-flocculation are often not required, but the presence of iron (II) and manganese (II) in anaerobic well water, which may precipitate when oxidized, may necessitate media filtration. Moreover, energy use, land use and landscape impacts are lower than for plants with an open intake and a conventional pretreatment.
- As cleaning intervals often increase because of the lower fouling potential, chemical use for cleaning is reduced and less cleaning wastes are generated, which would otherwise require treatment on-site or in a municipal wastewater treatment plant.
- Lower fouling potential and less frequent cleaning increases the membrane life-time, which reduces material and energy use in the manufacturing process.

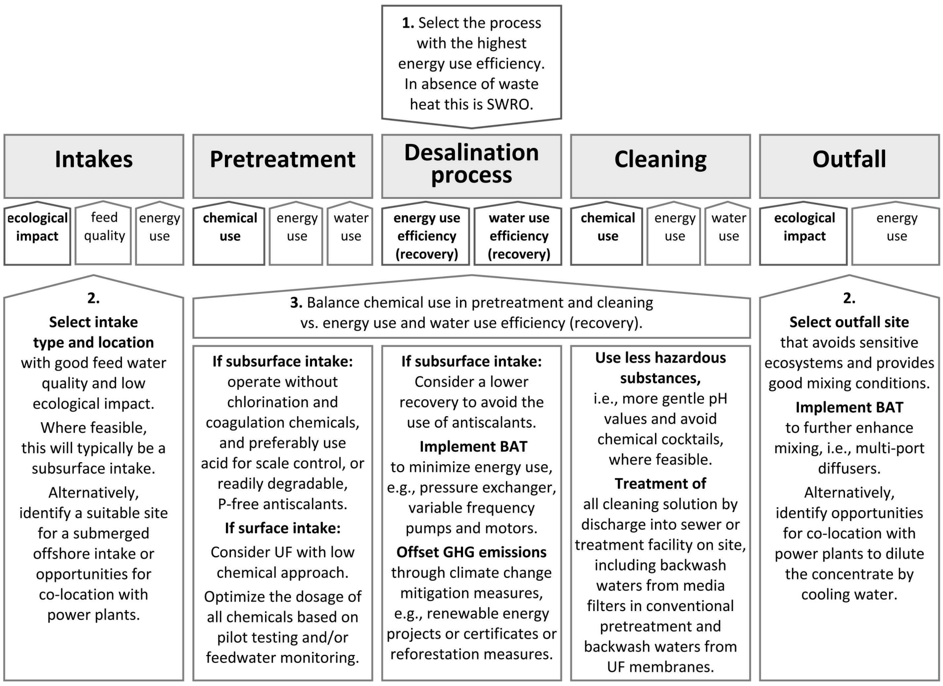

Figure 24: Road map to 'green' or 'sustainable' desalination (see Figures 18 to 23 for details).

The only pretreatment for a subsurface intake is often scale control. However, SWRO plants have been reported to work well without antiscalant additives and the need for antiscalants has to be questioned based on recent laboratory results which indicate that scaling may rarely occur in SWRO systems [56]. For example, it is possible that antiscalants will not be used in the Sydney SWRO plant under construction, which will be one of the largest plants in the world (capacity of 250,000 m^3/d, upgrade capacity of 500,000 m^3/d). However, it is much too early for a final conclusion. More evidence is needed from laboratory studies, pilot testing and operational plants. Until then, the two precautionary approaches to 'lower' the risk of scaling are (i) to lower the recovery and achieve chemical-free operation (if a subsurface intake is used) or (ii) to use antiscalants, which can be simple acid addition to control calcium carbonate scaling in SWRO.

A low recovery rate increases the feed water requirements of the plant, but since the water is taken from a subsurface intake, neither impingement, entrainment nor increased use of other pretreatment chemicals is a concern. However, the specific energy demand of the desalination process and the energy needed for pumping increases with decreasing recovery. In essence, a trade-off between chemical use on the one hand and a chemical-free but more energetically intensive process on the other hand must be made. If a need for antiscalants has been established through pilot testing, readily biodegradable, phosphor-free polymers should be used. A disadvantage of adding readily biodegradable organic material to the system may be an increase in biofouling on the SWRO membranes. The use, dose and type of antiscalants should therefore be carefully deliberated, and preference may be given to the use of acid, especially in single stage SWRO plants. If the recovery is reduced, the increase in energy demand can be compensated by climate change mitigation measures. Another benefit of a lower recovery is that the salinity of the concentrate is lower, and hence dilution to ambient levels can be more easily achieved.

The main concerns of subsurface intakes are the construction-related impacts. These include the disturbance of soils and sediments, the formation of sediment plumes which increase turbidity and may affect water quality, habitat destruction, the disturbance of sensitive wildlife, and possible adverse effects on groundwater processes and flows. As a BAT approach, trenchless techniques and best environmental practice, such as timing of construction activities to seasons when sensitive species may be less abundant or absent, should be implemented to minimize the adverse effects on the coastal ecosystem. The hydrological conditions of the intake area should furthermore be investigated in order to avoid adverse changes in groundwater flows and conditions.

An acceptable alternative where a subsurface intake is not possible due to geological or environmental constraints is a submerged intake in deep water in an offshore location. It should have a large surface area resulting in low flow velocities (passive screen), velocity caps, and fine-mesh screens which can be backwashed with air. A suitable alternative to conventional pretreatment, which is often needed for surface water and even submerged intakes, may be UF with a low chemical approach. Co-location to a power plant with an OTC system, preferably with an offshore submerged intake, may be another alternative where a subsurface intake is not possible. Co-location provides certain environmental benefits over a stand-alone intake [162], but OTC systems are not considered BAT in some locations, especially in restricted water bodies, but also in open coast locations in California, mainly because of impingement and entrainment effects. Since distillation plants are often co-located to power plants and have larger water requirements than SWRO plants, intakes should be located in offshore submerged locations.

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To conclude, seawater desalination could be described as the "epitome" of the global energy and water crisis (cf. section 6.2), as it exemplifies that energy, water, climate change and the world water crisis are inseparable problems. Desalination is the most energy intensive of all water treatment processes, which is usually only implemented as a last resort where conventional freshwater resources have been stretched to the limit. Yet, the production of energy for desalination in conventional power plants with wet cooling systems often depends on freshwater resources in rivers and reservoirs. Desalination is furthermore considered as a draught-proof water source, which does not depend on river flows or reservoir levels or climate change, but produces considerable amounts of GHG emissions if fossil energy sources are used, which contributes to the problem of climate change, which in turn may aggravate draughts in some regions.

A way of overcoming this dilemma is through implementing green, sustainable solutions that really live up to the expectations. For example, the Carlsbad project was recently described as "green SWRO" because of plans to compensate the *net* GHG emissions of the project over water import into San Diego County [264]. In other words, the plans will not change the status quo of the present GHG emissions of water supply in San Diego County. While the company that develops the project may not be responsible for improving the status quo, it is also a misinterpretation to call this a 'green' solution. The GHG emissions of the project are the same as those of 90% water import by long-distance transfer at an energy cost of 2.8 kWh/m^3 – who would call this a green solution? In democratic societies, new desalination projects have to pass the test of public opinion, which is increasingly under the influence of environmental considerations. A green image campaign is therefore only credible if it is well-grounded. Understating the energy requirements of desalination by comparing it to other, more excessive forms of energy use, two common examples being air conditioning and long distance transfer, is not instrumental in convincing the public of a new project.

Sustainable desalination is not a utopia but technically feasible and economically viable. For instance, the Sydney SWRO project with an upgrade capacity of 500,000 m^3/d, which is currently the largest infrastructure project in New South Wales, was developed under a tight schedule of only four years without compromising environmental protection. The project came under much scrutiny from the public regarding its environmental impact, and hence involved various monitoring and other specialist studies, and GHG emissions are being offset by a new wind farm [61].

In contrast, planning of the first large SWRO project in Carlsbad, California, started in 1998 and its commissioning is not expected before 2011 or 2012, mainly because of environmental concerns. Some blame the lack of a well-defined water and desalination policy and the existing regulatory structure, which gives a number of agencies permitting authority over the project rather than nominating one lead agency which coordinates the process. Others assign blame to the applicant for a lack of transparency, a delay in the request for information, and failing to establish an acceptable method of mitigating impingement and entrainment effects at the power plant [297].

One main difference between California and Australia may be that Australia is experiencing a much more severe water shortage, while California is proceeding more slowly because cheaper options are still available. The most important difference, however, may be that Australia has done whatever was required to ensure desalination works environmentally, with price being a secondary consideration [297].

In this respect, it is noteworthy that sustainable solutions may also be economic. Model calculations for a 200,000 m^3/d plant show that cost saving from energy optimization over a 20 year lifetime can amount to or surpass the initial capital expenditures of the complete plant, depending on electricity cost developments in the coming years. Hence, a systematic reduction of the energy consumption of a SWRO plant is not only an environmental protection measure, but also an economical benefit [53].

To conclude, the Australian projects, including Sydney, Perth or the Gold Coast project, set a good example for environmental protection that will hopefully encourage others to follow in their footsteps. The industry, regulators and communities alike, however, have to pave the way by making a commitment to more green and sustainable desalination projects. Environmental protection measures will most certainly increase the cost of water production. For two recent Australian SWRO plants, the advanced seawater intake and concentrate outfalls cost more than the entire capital cost of the Ashkelon plant [298]. Sustainable desalination is not a utopia, but requires a commitment to providing water at a price which does not only include the usual construction and operating costs, but also the costs that are necessary to reduce the environmental footprint through environmental studies, advanced technology, or compensation measures. The best practicable environmental option can best be identified in a project- and site specific environmental impact assessment study. A catalogue of best available techniques (BAT) may be useful in guiding practitioners, consultants and decision makers in their choices.

Part II

Environmental impact assessment (EIA)

7 Concept and methodology

7.1 Introduction

The United Nations Environment Programme (UNEP) developed a guidance document on desalination in cooperation with the World Health Organization (WHO), which intends to assist project designers, consultants, regulators and decision makers to anticipate and address all relevant environmental, socio-economic and public health concerns that may arise when undertaking a desalination project, for obtaining maximum beneficial use of the desalinated water in terms of quality, safety and environmental protection. This chapter gives a short account of the guidance development process and summarizes the main results and recommendations from the environmental working group.

In 2004, WHO identified a clear public health and environmental protection argument to initiate a project on "Desalination for Safe Water Supply". As desalination is applied to non-typical source waters and uses non-typical water treatment techniques, the concern had been raised that the existing WHO guidelines for drinking-water quality [164] might not fully cover the unique factors that can be encountered during the production and distribution of desalinated drinking water. With the worldwide need for desalinated water rapidly increasing, the need for an evaluation of desalination technologies was evident. Environmental considerations, which are normally not a field of WHO activities, were included into the topic areas of the project because the protection of coastal ecosystems and groundwater aquifers were considered key concerns besides public health, which should be addressed during the design, construction and operation of a desalination facility.

Parts of this chapter were based on:

S. Lattemann and H. El-Habr. UNEP resource and guidance manual for environmental impact assessment of desalination projects. Desalination and Water Treatment, 3: 217−228, 2009.

S. Lattemann, K. Mancy, H. Khordagui, B. Damitz and G. Leslie. Desalination, resource and guidance manual for environmental impact assessments. United Nations Environment Programme (UNEP), Nairobi, Kenia, 2008.

S. Lattemann. WHO guidance on desalination: results of the work group on environmental impacts. International Desalination Association (IDA) World Congress on Desalination and Water Reuse, Maspalomas, Gran Canaria, 2007.

A steering committee of renowned experts in the field of desalination and an oversight committee of representatives from different international organizations, including WHO and UNEP, were established to guide the process. Five technical working groups were formed[a], consisting of more than 35 scientists and engineers, which conducted the scientific analyzes and generated results and recommendations over two years time. The draft document underwent an internal WHO review and public commenting process. The results and recommendations from the *environmental* working group were partly included in the WHO publication [16] and reproduced in full by the United Nations Environment Programme (UNEP) as a separate guidance document in 2008 [17].

The UNEP document is divided into three parts. In part A, an introduction to the concept, methodology and practice of environmental impact assessment (EIA) is given and a 10-step EIA approach is proposed. Part B outlines a possible modular structure of an EIA report and gives an overview on a wide range of thematic issues that may be relevant to desalination projects. Part C discusses the potential impacts of desalination plants on the environment, based on a comprehensive literature review, and evaluates the identified impacts in terms of significance and relevance for EIA studies (see chapter 5). In the following, the main results from part A and B are summarized.

7.2 EIA approach

EIA studies are widely recognized and accepted as a suitable approach for identifying, evaluating and mitigating potential impacts of development projects on the environment. The main objectives of an EIA are to provide information on the environmental consequences of a project for decision making, and to promote environmentally sound and sustainable development through the identification of appropriate alternatives and mitigation measures. Based on the EIA results, a decision has to be reached which balances the societal and environmental impacts of a project versus its benefits [299].

Detailed EIA studies involving pre- and post-installation monitoring programmes are often required for major infrastructure projects, such as dams or power plants. In principle, EIAs for large desalination projects will not differ in terms of complexity and level of detail from other infrastructure projects and especially other water supply projects. Depending on the proposed project, it is incumbent on the national authorities to individually define the need, scope and complexity requirements of each EIA study.

EIAs are usually not limited to environmental aspects, but typically address all potential impacts of new projects, plans or activities on ‘man and the environment’. This often requires an interdisciplinary approach, covering different natural and environmental science disciplines. Taken a step further in relating potential impacts to people and communities, it may also be necessary to consider human health and socio-economic aspects where appropriate. Public participation is therefore another fundamental element of EIAs in many legislative systems, particularly for community infrastructure projects.

In other words, EIAs are multi-stage, multi-disciplinary studies, often involving field monitoring, different scientists and experts, government agencies, stakeholders as well as the wider public. With the context so broad, difficulties may be experienced in conducting the EIA and accompanying studies, and in analyzing the large amounts of complex information in a structured and consistent way for decision making.

[a] Technology – engineering and chemistry; Health – toxicology of contaminants and nutritional aspects; Microbiology – sanitary and marine microbiology; Monitoring – microbiological, analytical chemistry, surveillance, regulatory; Environment – environmental effects and impact assessments.

The UNEP guidance document offers a structured 10-step EIA approach to this problem (part A), lists a wide range of thematic issues potentially relevant to desalination projects (part B), which can be used for scoping of the project and structuring the reports, and gives an overview on the main environmental concerns of desalination projects (part C). Not all of the issues listed in the guidance are unique to desalination projects. Some apply similarly to other water treatment or infrastructure projects, while others may not be relevant to a particular desalination project in question. The guidance document intends to raise a wide range of potentially relevant issues, which may help to anticipate the relevant concerns of each desalination project on a case by case basis.

EIA methodology

An EIA is generally marked by three main phases, which were subdivided into 10 steps (Figures 25, 26). The pre- or initial EIA phase includes screening and scoping of the project. In the main EIA phase, the scientific analyses are conducted, which includes the establishment of baseline data, the prediction and evaluation of impacts, and the identification of appropriate alternatives and impact mitigation measures. The final EIA phase involves decision making and a review of the EIA process. An environmental management plan is often established at this stage, which gives further specifications on environmental monitoring requirements during installation and operation of the plant in order to ensure compliance with any obligations that were imposed as part of the project permit. In practice, the proposed 10-step process may deviate from the outlined procedure, as single steps may not always be clearly delimitable, some steps may overlap, or it may be necessary to change the sequence of some steps. The EIA procedure should generally be understood as a continuous and flexible process.

Screening (step 1)

Screening is the process by which a decision is taken on whether or not an EIA is required for a particular project. It shall ensure that a full-fledged EIA is only performed for projects with significant adverse impacts on the environment or where impacts are not sufficiently known. Screening therefore involves a preliminary environmental assessment of the expected impacts of a proposed project and of their relative significance. This requires a certain level of basic information that is readily available about the project and its location, e.g., from literature or other sources [299].

Screening can either be carried out by a standardized or by a customized procedure. In the standardized approach, projects are classified by legislation into categories which are either subject to or exempt from EIA. This may include mandatory (positive) lists for projects that always require an EIA, lists which define thresholds and criteria for EIA, or exclusion (negative) lists. For example, an EIA may be mandatory for large electricity and water co-generation plants, or for desalination facilities with a production capacity above a certain threshold, but not for small systems as used for hotels, small residential communities or recreational areas. A class screening can be undertaken for small-scale projects that are routine and replicable, if there is a reasonably sound knowledge of the environmental effects and if mitigation measures are well established.

In case that project lists and thresholds are not defined by the applicable EIA laws, a customized screening approach is necessary using indicative guidance. Screening checklists are for example provided as part of the European EIA legislative system (including directives 85/337/EEC and 97/11/EC), which were included in the UNEP guidance for easy reference [300]. The lists include a number of questions referring to the project and its environment. Answers should be given based on the information that is readily

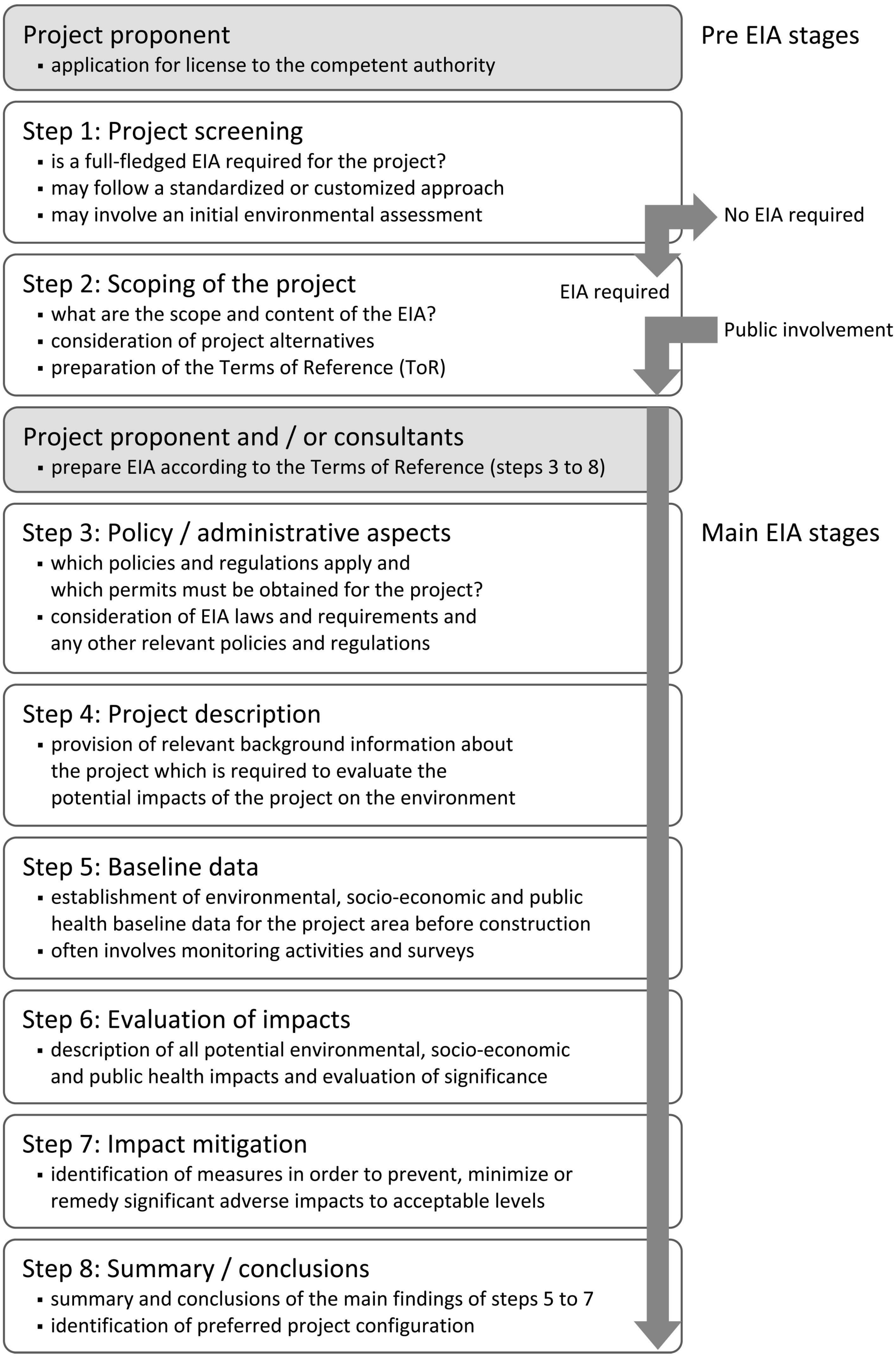

Figure 25: 10-step EIA process – scoping, screening and main EIA phase.

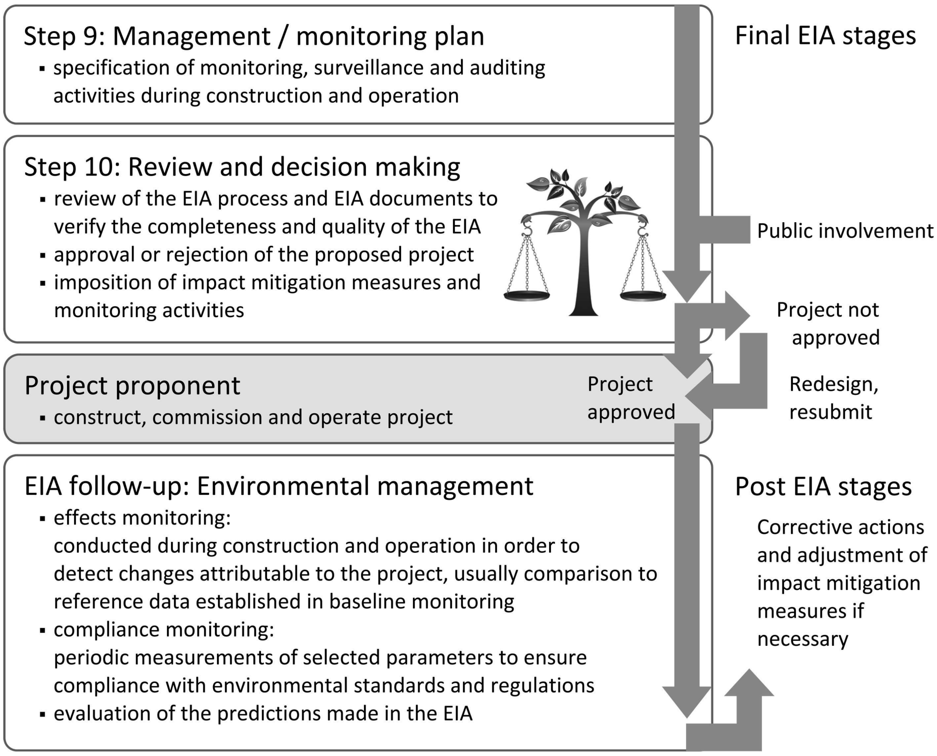

Figure 26: 10-step EIA process – EIA decision phase and follow-up activities.

available at this stage. The lists shall help to provide an answer to the question if the project is likely to have a significant effect on the environment. This is a discretionary decision. As a general rule, the greater the number of potential concerns and the greater the significance of the effects, the more likely an EIA is required. Uncertainty should always point towards an EIA, as the process will help to clarify some of the uncertainty.

After a formal decision has been made whether an EIA is required or not, an official screening document is typically prepared by the competent authority which records the results and underlines the decision. It may be extended into a short screening report, which also includes the results of the preliminary assessment, and can be used for public dissemination in the scoping stage of the EIA.

Scoping (step 2)

Scoping is the process of determining the contents of the EIA study. The terms of reference (ToR), which are elaborated in this process, provide clear instructions to the project proponent on the information that needs to be submitted to the competent authority for EIA, and on the studies to be undertaken to compile that information. Scoping is a crucial step in EIAs because it identifies the issues of importance on which the EIA should focus, and eliminates those of little concern.

Generally, scoping involves four basic steps:

- preparation of a scoping document for public dissemination, including project details and a preliminary environmental analysis,
- organization of scoping meetings, inviting collaborating agencies, stakeholder groups, NGOs, experts and advisers, and announcement of the scoping meetings in public,
- compilation of a complete list of issues during scoping consultations, which are then evaluated in terms of their relative importance and significance,
- preparation of the terms of reference for EIA, defining the scope and information requirements of the EIA, study guidelines and methodologies.

It is recommended that the competent authority takes responsibility at least for monitoring of the process, for preparing the minutes and official transcripts of the scoping meetings, for keeping the records of the scoping outcome, and for preparing the ToR.

An effective way of dealing with a larger number of desalination projects may be to elaborate a standard scoping procedure and standard ToR. The scoping process will often involve the same representatives of government agencies, NGOs, consultants, etc. A guideline, elaborated in a collaborative effort between these groups, may routinize the scoping procedure and may establish standards for the environmental studies to be undertaken and the information to be submitted in EIAs for desalination projects, but would still allow for project-related specifications.

Policy and administrative aspects (step 3)

An EIA usually takes place within the distinctive legislative system established by the individual country, state, or district where the project is to be located, as well as within the legislative frameworks of international institutions. It is therefore recommendable to gain a deeper insight and understanding of any national or international regulations that may apply to the EIA procedure itself.

Moreover, all thematically relevant laws and policies need to be identified, relating for instance to the conservation of nature and biological diversity in the project area, to the control and prevention of pollution of water bodies, to water resources management, or to land-use and regional planning in the area. In many jurisdictions, more than one permit will be required to realize a desalination project. The main approval process, which authorizes the construction and operation of a desalination plant, will not necessarily replace other existing statutory provisions and permits.

It is important to clarify early in project planning which permits must be obtained and to contact the competent authorities. The permitting process may be facilitated by nominating a ‘lead’ agency, which coordinates the process by involving other agencies and by informing the project proponent about permitting requirements.

For example, construction and operations of the Tampa Bay seawater desalination plant and pipeline in Florida required 18 separate permits. The process was described as “lengthy and extensive, particularly the Florida Department of Environmental Protection’s permitting process” [75]. Similarly, a number of different regulatory agencies have permitting authority over the Carlsbad project in Southern California. As the project design evolved, additional reviews by the permitting agencies were required, and the reiterative process took a considerable amount of time. One agency must approve a permit before it can go to the next agency, often causing significant delays. It was therefore noted that a new lead government agency, whose sole responsibility would be to coordinate all permitting activities, would be helpful [297].

Project description (step 4)
A technical project description should be prepared and included in the EIA report, which provides the required background information in order to identify and investigate all potential environmental concerns of the project. The project description should cover the different life-cycle stages of the project including construction, commissioning, operation, maintenance and decommissioning of the plant. It should estimate all resources that are consumed during the different project operations, such as land area requirements during construction, the use of materials or chemicals during plant operation and maintenance, or energy use. It should furthermore include a characterization of all waste products in terms of quantity and composition, including emissions into air, water, and soils, as well as solid and liquid waste products transported to a landfill or discharged into the municipal sewer or stormwater system. The technical description should be succinct and to the point, making a selection between those technical details that are necessary for the impact assessment and those which are irrelevant in this context. For large and complex projects, a more suitable approach might be to split the technical project descriptions and EIA into separate documents. For example, an environmental assessment report and a preferred project report were prepared for the Sydney SWRO plant [155, 232].

Establishment of baseline data (step 5)
This step entails the collection, evaluation and presentation of baseline data of the relevant environmental, socio-economic, cultural and public health characteristics of the project area before construction. This should include any existing levels of degradation or pollution, such as other development activities, noise levels, or sources of emissions. The information requirements of the baseline studies are determined during scoping (step 2, page 153). For more details on marine environmental monitoring, see chapter 8.

A reference area with similar characteristics may be selected, for which baseline data is established in the same way as for the project site. This allows for a comparison between the reference and the project site during project monitoring in order to detect any changes caused by construction and operation of the project. Reference data from a site with similar environmental characteristics is particularly useful to identify natural variations or other anthropogenic effects not related to the desalination project.

Evaluation of impacts (step 6)
This step of the EIA describes and evaluates the potential impacts and benefits of the proposed project on 'man and the environment', covering all relevant socio-economic, public health as well as environmental issues. Socio-economic and cultural considerations include for example the project's effects on the day-to-day lives of the individuals and the community, on the management of natural resources, or on local and regional development. Public health impacts refer to changes in the quality of life and community health, or potential health risks associated directly or indirectly with the desalination project. Impacts on the environment would include all emissions into air, soils and water, impacts on landscape characteristics, or any disturbance of species.

The prediction of impacts is typically based on field and laboratory experimental methods (e.g., whole effluent toxicity tests, cf. page 175), small-scale models to study effects in miniature (e.g., of outfall designs), analogue models which make predictions based on analogies to similar existing projects (e.g., other desalination plants) or mathematical models (e.g., hydrodynamic modeling of the discharges, cf. page 174). As each of these methods covers the range of impacts only partially they are usually used in conjunction with each other, resulting in a range of different specialist studies.

The relative significance of the predicted impacts should be evaluated, using criteria such as: Is the impact direct or indirect, positive or negative? Is the impact temporary, long-term or permanent? What is the extent of the impact, in terms of geographical area, or size of the population affected? How severe is the impact, how likely will it occur, is it reversible or can it be mitigated? If possible and where appropriate, secondary effects, potential cumulative impacts with other development activities on the project site, trans-boundary (far-distance) effects and growth-inducing effects should be identified.

Impact mitigation (step 7)

At this stage, specific recommendations need to be elaborated that mitigate the predicted effects of the project. The step of impact mitigation should identify the most feasible and cost-effective measures to avoid, minimize or remedy significant negative impacts to levels acceptable to the regulatory agencies and the affected community. The definition of acceptable will vary according to different national, regional or local standards, which depend on a society's or community's social, ideological and cultural values, on economic potentials and on politics. Guidance in this process should be provided in the form of standards for BAT of desalination projects (chapter 6).

The elements of impact mitigation are organized into a hierarchy of actions [299]. Impact prevention by adequate measures and alternatives is usually given the highest priority. If prevention is not possible, impacts should be minimized as far as possible. All remaining impacts which are significant but unavoidable, and which cannot be mitigated further, should be remediated and compensated where possible.

Remediation may for example involve habitat enhancement in the project site after construction activities or restoration of the project site to its original state after project decommissioning. Compensation measures may include enhancement of resource values at another location, e.g., by habitat enhancement, reforestation or restocking of a certain species. Impact mitigation measures can generally include structural measures (e.g., design or location changes, technical modifications, waste treatment) and non-structural measures (e.g., purchase of renewable wind energy certificates).

For example, one third (15 hectares) of the project site of the Sydney SWRO plant has been reserved as a conservation area, which will be rehabilitated and maintained to protect endangered ecological communities and habitat for threatened species [61, 155]. The mitigation plan for the Carlsbad SWRO project will restore 22 hectares of wetlands in an off-site location along the Southern California coast to mitigate the reduced productivity of 15 hectares of habitat impacted by the plant [301].

Summary and conclusions (step 8)

In this step, the main findings and recommendations of steps 5 to 7 are summarized. The focus should be on the key information that is needed for decision making. An overview of the main impacts, preferably in the form of a table, should be provided for this purpose, distinguishing between significant impacts which can be prevented or minimized, and those which cannot. The identified mitigation measures or alternatives should be given for all impacts that were found to be significant. In essence, the original project proposal should be systematically compared with alternative project configurations in terms of adverse and beneficial impacts and effectiveness of mitigation measures. Finally, the 'best practicable environmental option' should be identified, which is the preferred project configuration under environmental, social, cultural and public health criteria. It should be ensured that this option is both economically and technologically feasible. The decision should be transparent and backed by arguments. An effective and transparent way may be to analyze and present the results by multi-criteria analysis (chapter 9).

Environmental management plan (step 9)
An environmental management plan should be elaborated to ensure the ongoing assessment and review of the effects of the desalination project during all life-cycle stages. It has the objective to identify the *actual* impacts of the project and to verify that the *observed* impacts are within the levels *predicted* by the EIA. Moreover, environmental management has the objective to determine that the imposed mitigation measures or other conditions attached to the project permit are properly implemented and work effectively. If not or if unanticipated impacts occur, the measures and conditions should be adapted in the light of new information. The management plan should specify any arrangements for planned monitoring activities, including methodologies, schedules, and management protocols in the event of unforeseen events [299]. More details on the scope and design of marine monitoring studies are included in chapter 8.

Effects monitoring is typically based on field measurements, such as surveys of species abundances and diversity in the project site. It has the primary objective to measure the environmental changes attributed to project construction and operation. By comparing the data from baseline and operational monitoring and from the project and reference sites, changes which are attributable to the project can be detected and distinguished from natural variations. For example, two years of baseline and at least three years of operational effects monitoring is conducted for the Sydney SWRO project [61].

Compliance monitoring refers to the periodic or continuous measurement of a certain parameter in order to ensure that regulatory requirements and environmental quality standards are being met, such as for example the measurement of salinity levels in the discharge and mixing zone. For example, real time monitoring buoys were installed in the discharge area of the Perth SWRO plant which measure temperature and dissolved oxygen levels in addition to salinity levels in one-minute intervals [69].

Both types of monitoring activities permit only reactive impact management, since they detect violations or adverse changes after these have taken place. It is therefore important to respond to the outcomes of monitoring by establishing linkages to impact management, for example by establishing protocols to be followed and actions to be taken if a certain threshold or trigger value is exceeded. In the case of the Perth plant, where management responses had been agreed with the regulator, the plant was shut down twice in 2008 due to low dissolved oxygen levels at the seafloor, even though this effect was most likely caused by a natural stratification event [176].

EIA review and decision making (step 10)
The purpose of review is to verify the completeness and quality of the EIA, and to ensure that the information provided in the EIA complies with the terms of reference as defined during scoping (step 2, page 153) and is sufficient for decision making. The review may be undertaken by the responsible authority itself, another governmental institution or an independent body. Participation of collaborating and advisory agencies, the public and major stakeholders in the review process is recommended.

Following review, the EIA report is submitted to the competent authority which will decide on approval or rejection of the proposed project based on the EIA report, the analysis of stakeholder interests, statements from collaborating agencies, etc. In this stage, trade-offs between environmental, social, economic and other criteria usually have to be made, which is a political decision. The decision making process can be facilitated by multi-criteria analysis (chapter 9). The competent authority typically imposes conditions if the project is approved, such as mitigation measures, limits for emissions, or environmental standards which must be observed.

Outline of an EIA report

The EIA report is the primary document for decision making. It should therefore clearly organize and synthesize the results obtained during the studies and consultations of the EIA process. The 'contents' or 'checklist' included in part B of the UNEP document gives an overview on a range of thematic issues potentially relevant for different desalination projects and environments. As such, the list tries to be inclusive rather than exclusive. It can serve both as a reference source in the early stages of the EIA, e.g., during scoping (step 2), as well as for drafting the EIA report at the end of the process. By screening the information provided, it can be decided on a case by case basis which issues may be relevant to a specific desalination project and which are not.

The checklist is subdivided into four sections (front matter, project background information, environmental impact assessment, and back matter to an EIA report) and its structure widely reflects the 10-step process. For example, the project background information comprises four chapters: the introduction, which states the rationale and purpose of the EIA according to the screening decision (step 1), a chapter on the scope and methodology of the EIA as specified in the terms of reference (step 2), a chapter on policy and administrative aspects (step 3), and one chapter detailing technical project aspects (step 4). The section of the EIA report that contains the results from the actual impact assessment comprises all relevant socio-economic, human health as well as environmental considerations (steps 5 to 7). It is proposed to include the following chapters and sub-sections into an EIA report for a desalination project where relevant:

Abiotic environment:
- characteristic landscape and natural scenery
- terrestrial site (soils, ground- and surface water)
- marine site (seafloor, sediments and seawater)
- air quality and climate

Biotic environment:
- terrestrial biological resources
- marine biological resources

Socio-economic and environmental health aspects:
- population, housing and community structure
- economic growth and development activities
- environmental health factors
- water resources use
- land and marine use
- utilities and service systems
- cultural resources

For each of these topic areas, the following information should be included:

- A detailed description of the existing setting (baseline), which describes the present and future state of the environment in the absence of the desalination project (zero alternative), taking into account changes resulting from natural events and from other human activities, and often involving field studies if sufficient literature data about the project site is not available from previous monitoring studies.
- A discussion of the expected impacts in the different life-cycle stages of the project, i.e., during construction, commissioning, operation, maintenance and decommissioning as far as these are predictable at the stage of project planning, including a judgment whether or not these are considered to be significant.
- A description of impact mitigation measures in order to avoid, reduce, remedy or compensate for any significant adverse impact resulting from the project.

The complete 'checklist' spans 30 pages. An excerpt is given in the box on page 159 showing the information included on "Characteristic landscape and natural scenery".

Characteristic landscape and natural scenery

EIAs for desalination projects may include a landscape impact assessment, which is directed towards predicting and evaluating the magnitude and significance of effects that a new facility has on the audio-visual characteristics of the surrounding landscape.

The effects of a desalination project on landscape properties cannot be 'measured' and 'quantified' as precisely and objectively as for other features of the project site. To assess the magnitude and significance of effects, an expert judgment is typically obtained. This should be based on good practice, follow a structured and systematic approach, and provide reasoned arguments, but even so, people will not necessarily subscribe to the expert opinion.

Effects on landscape properties will often be perceived differently by people who judge by their own aesthetics and subjective perception of the project. A landscape impact assessment is typically discussed controversially in the public. The landscape impact assessment is the part of the EIA which will help the public to imagine the potential audio-visual impacts arising from the project, and to form an opinion about the project.

Existing setting

This section depicts the pre-construction setting of the project site with regard to natural features such as islands, cliffs, dunes, river mouths, marshes, scenic views, etc. Typically, photos from different perspectives (e.g., from elevations, in different directions) are taken during good weather and visibility conditions to illustrate the landscape properties as they may be perceived by a human observer.

The description of the scenery would also include an assessment of the ambient noise level. It may distinguish between natural sounds caused by wind, waves, animals, etc., and those caused by human activity in the site or vicinity, such as by docksides, traffic, etc.

This section would include a projection of the anticipated future development without project realization (zero alternative), but taking other development activities into account.

Impacts

It is evaluated how the landscape will change and how an observer may perceive the scenery if the project is realized, including:

- noise generation,
- obstruction/alteration of scenic views,
- production of glare,
- any other audio-visual effect that substantially alters the character of the area.

This section typically includes a visualization of the project from different viewpoints, for example computer generated photomontages or animations, and provides ranges for visibility and audibility of the facility in the form of visibility and audibility maps.

Mitigation and avoidance measures

This section lists the mitigation measures that are proposed for the project, e.g.:

- screens during construction to shield off noise and unsightly views,
- noise reduction measures during operation such as noise barriers,
- landscaping measures such as planting of trees and shrubbery,
- materials of finishes (e.g., reflective or non-reflective materials),
- colors of external appearance,
- lighting of the building complex.

The mitigation and avoidance measures should be designed to blend the facility in with the surrounding natural or artificial landscape features. The different measures such as vegetation and noise barriers should be illustrated by visualizations (photomontages) and their effect on noise levels illustrated in noise mappings.

7.3 General considerations for EIA studies

Consideration of alternatives

A central element of all EIA studies is the comparison of possible alternatives in order to identify the option with the least environmental impact. Alternatives should include project modifications regarding location or process design, but also different water supply or management/conservation options ('zero' alternatives). To facilitate site selection for desalination plants, authorities should designate suitable areas in regional plans. General criteria for site selection are given in Table 36 (page 140). To facilitate process selection, industry standards for BAT should be established (chapter 6). Alternatives can be generated and refined most effectively in the early stages of project development, when the disposition to allow major project modifications is highest among the participants.

Consultation and participation

Another important factor in an EIA is early and extensive coordination with agencies and stakeholders. According to Tom Luster from the California Coastal Commission, there are usually certain key design issues that are likely to result in a review being easier and of shorter duration, or more difficult – as for example in the Carlsbad case. One of these issues is a surface intake versus a subsurface intake. Peter Gleick, president of the Pacific Institute, believes that the applicant's biggest error has been to insist on an open intake, taking water from a power plant OTC system, despite national policies to eliminate OTC. According to Gleick, this has "made the regulatory agencies job much, much more difficult", and hence caused the major project delays [297].

Another example, where early consultation may have prevented conflicts at the end of the EIA is the Olympic Dam SWRO project in Southern Australia. Although the EIA process devoted much attention to site selection and environmental studies, and the outfall was carefully designed as to meet a conservative interpretation of acceptable performance [179], the Parliament of South Australia has just recommended the applicant to reconsider its proposed site for the 250,000 m^3/d plant. It is believed that the proposed location in an inverse estuary experiences slow turnover, and is furthermore recognized as the only known mass spawning aggregation site of Giant Cuttlefish in the world. Further investigations are required into alternative siting of the desalination plant with an emphasis on local, regional, company and governmental collaboration [302].

Public participation is another integral part of EIAs, particularly for desalination projects which will supplement municipal water supplies. Public involvement should seek to inform the public and gather different perceptions about the project, addressing the benefits as well as potential public health, environmental and socio-economic concerns. Involving a broad public will furthermore ensure that important issues are not being overlooked, thus providing for the comprehensiveness, quality and effectiveness of the EIA. Another benefit of public involvement is that a partnership with the community can be developed, which is critical for the success and sustainability of a project.

Precautionary approach

EIAs can only give a prognosis of the expected impacts based on the information that is available before project implementation. It is recommended to deliberate carefully about the accuracy of all predictions made in the EIA, which can only be as valid as the underlying data and information. Information gaps and deficiencies should therefore be clearly identified in the EIA and a precautionary approach applied in the evaluation of potential impacts and in decision making, as established by Principle 15 of the Rio Declaration on Environment and Sustainable Development.

7.4 Summary and conclusions

EIA studies are a widely recognized and accepted approach for identifying, evaluating and mitigating potential impacts of major infrastructure projects on the environment. To this day, however, only a handful of EIA studies have been carried out for desalination plants and made publicly available, most of them from Australia and the United States. EIAs for desalination plants in other parts of the world are scarcely available, although a major effort was made during the WHO project to contact key personalities in the industry and in different countries of the Mediterranean and MENA regions. The reason is probably that EIA studies are considered as intellectual property of the project proponent. If required by law, EIAs are displayed locally for a specified time period or sent directly to the participating organizations. However, this withholds the studies from a wider audience, which does not facilitate the notion of public participation.

In some cases, the EIA investigations were carried out under immense time constraints. For instance, only 4 months were given for an EIA study for a large SWRO plant in Algeria [303]. This shows that environmental concerns can be of secondary importance when a ready supply of freshwater is urgently needed. The opposite is also true: comprehensive and time-consuming environmental studies are currently being carried out for the major SWRO projects in Australia, while environmental concerns are the major hurdle in the permitting process of new projects in California. In Spain, the government has announced plans to speed up the EIA process from the current average of 770 days for infrastructure projects to no more than six months. The reform could benefit vital water projects, however, doubts were raised from within the European Commission that it would be possible to condense the whole EIA procedure including public consultation into a six month period [304].

In the EU, the EIA Directive[b] regulates which project categories have to be made subject to an EIA by member states. It does not list desalination plants, which may be due to the fact that desalination plants were small and only used at a minor scale in Southern Europe at the time when the directive was first introduced in 1985 and later amended in 1997. As EIAs are mandatory for other major water supply projects, such as groundwater abstraction schemes, dams, and works for the transfer of water resources between river basins, it would be consistent to include desalination projects in the directive as well. Moreover, desalination projects should be an integral part of water resources management planning that not only considers the development of new or existing water supplies, but also the economic use and reuse of water where possible. According to EU regulations, a strategic environmental assessment is mandatory for plans and programmes in the field of water management (SEA Directive[c]).

In 2004, Manuel Schiffler from The World Bank stated that an "internationally agreed environmental assessment methodology for desalination plants does not exist so far and its development would be desirable" [4]. The UNEP guidance document partially fills this gap. It offers guidance for designers of desalination projects, consultants, regulators and decision makers on the methodology, scope and contents of EIA studies and specifically for desalination projects. Still missing, however, are long-term monitoring studies that improve the basic understanding of the actual environmental impacts of desalination plants. Although an increasing number of EIA studies is being published, these are mainly based on conceptual models and laboratory experimental methods, including

[b] Directive 85/337/EEC on the assessment of the effects of certain public and private projects on the environment, amended by Directive 97/11/EC.

[c] Directive 2001/42/EC on the assessment of the effects of certain plans and programmes on the environment.

hydrodynamic modeling of the discharges and effluent toxicity testing, carried out *before project start-up.* The results need to be verified for the majority of these projects in effects monitoring studies *during plant operation.*

Chapter 8 deals with the scope and design of environmental monitoring studies for desalination projects. Problems in designing adequate monitoring programmes are discussed, the scopes of the studies are outlined, including baseline and operational monitoring, compliance monitoring, toxicity testing and hydrodynamic modeling, and criteria for assessing the sensitivity of species and habitats are proposed.

Marine environmental monitoring 8

8.1 Introduction

As outlined in chapter 6, BAT and EIA are complementary approaches. BAT aims at identifying suitable processes at the technology level, which can facilitate the identification of individual BAT solutions with a low environmental footprint. The environmental impact furthermore depends on the *site-specific* characteristics of the project site, which are investigated in EIAs. EIAs usually comprise a predictive process, aimed at detailing the likely impacts that would arise from a proposed activity on a given site (ecological risk assessment), and the postdictive process, aimed at quantifying the actual impacts after they have taken place [305]. In order to evaluate the sensitivity of the project site and to quantify the actual impacts, EIA studies usually involve extensive field monitoring programmes before and after project implementation (section 7.2, page 150).

Although a few EIAs of desalination projects have recently become available, these reflect the state of knowledge from the predictive process, while results from the postdictive process are only now beginning to be investigated. The longest monitoring programme, of which results have been regularly published, only looks back on two years of operational monitoring [176]. In 2008, the U.S. National Research Council attested a "surprising paucity of useful experimental data, either from laboratory tests or from field monitoring". Among the long-term research needs identified were site-specific assessments of the impacts of source water withdrawals and concentrate management, and the development of monitoring and assessment protocols for evaluating the potential ecological impacts of concentrate discharge [5]. In other words, still missing are the results from systematic monitoring studies and methodological frameworks for conducting these studies. The core of the problem is to design a monitoring programme that can adequately distinguish the effects of the desalination project from natural processes.

The existing studies, as reviewed in Lattemann et al. [17], used a wide range of approaches and methods to investigate the environmental impacts of desalination plant discharges. They were either limited in *scope* – addressing only one effect, such as elevated salinity on a specific species, *short-term* – without a continuous baseline and effects monitoring, and *localized* – not taking effects over a wider area into account which may

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S. Lattemann, M.D. Kennedy, J.C. Schippers and G. Amy. Seawater reverse osmosis: a sustainable and green solution for water supply in coastal areas? Submitted to Balaban Desalination Publications for a book on seawater desalination to be published in memory of Sydney Loeb.

arise from the dispersal of pollutants. In a nutshell, most studies fell short of recognizing the potentially synergetic effects of the single waste components of the discharges on marine organisms and the complexity of potential responses by the ecosystem. While the possible risk of damage arising from the concentrate discharge to the marine environment in close proximity to the outfall is at hand, no conclusive evidence can yet be provided concerning the long-term impacts or cumulative impacts in certain sea areas.

One of the most comprehensive environmental monitoring programmes to date is carried out for the Perth SWRO project in Western Australia, which started operation in 2006. The initial EIA studies covered potential contaminant releases, hydrodynamic modeling, and ecological effects of the discharge. A peer review of the pre-construction studies by the National Institute of Water and Atmospheric Research (NIWA Australia) in 2005 concluded that the studies have in general been carried out to a high standard, but that they were constrained to using mostly existing data due to significant time pressure. The reviewers were thus not convinced that the studies addressed all concerns adequately, and did not believe that the conclusions of the EIA reports can be accepted with a high degree of confidence [306]. In response to the review, more extensive studies were initiated, including marine baseline studies, a real time monitoring system before and during operations, and laboratory tests on toxicity [69].

The Perth example illustrates the difficulties that may arise in deciding upon adequate monitoring studies and stresses the need for a holistic monitoring and assessment framework. Comprehensive environmental monitoring studies, involving baseline and operational monitoring and laboratory toxicity tests, are now also being carried out for other Australian projects, including the Sydney, Gold Coast and Olympic Dam SWRO projects, which will provide valuable results in the near future. This chapter discusses aspects relevant to the design of monitoring programmes for desalination projects, including the scope of the studies as well as their scientific underpinnings.

8.2 The principles of environmental monitoring

A holistic monitoring framework, as part of an EIA, should integrate stressor-based and effects-based approaches. The *stressor-based* approach consists of identifying potential stressors associated with a project over its life-time, potentially affected receptors in the environment, and pathways for interaction. The approach assumes that all stressors associated with a project are known, and falls short of recognizing that cumulative stressor sources may exist within an aquatic ecosystem. It should therefore be combined with an *effects-based approach* which measures the ‘accumulated environmental state’ of the ecosystem by comparing environmental indicators between developed and undeveloped sites in order to identify effects that may occur as a result of unidentified stressors, or as a result of stressor interaction. This requires more intensive field monitoring than would be required for a project under a stressor-based approach only [307].

The stressor-based approach usually involves baseline and operational monitoring in the project site. The effects-based approach additionally requires monitoring in an undeveloped reference site. This design is known in its simplest form as the ‘before and after’ and ‘control and impact’ (BACI) approach. Monitoring programmes based on the BACI design have the objective to isolate the impact from the ‘noise’ introduced by natural temporal and spatial variability [308]. However, there are many practical problems with the BACI approach in its simplest form which need to be overcome by more sophisticated designs in order to be able to actually detect impacts.

One practical problem is the large temporal variance of many populations, which is reflected in very 'noisy' abundances [309]. To capture this temporal variance, the BACI design can be extended to have several simultaneous 'paired sampling' dates before and after the perturbation in both the control and impact site (BACI*PS* design). The difference (Δ) in a parameter value between both sites is assessed on each sampling date. The *average* delta in the *'before' period* (Δ_B) is an estimate of the present and expected future difference between the two sites in the absence of an impact. The difference between the average *'before'* and *'after'* deltas ($\Delta_B - \Delta_A$) provides an estimate of the magnitude of the environmental impact. Parameters with a large impact and small natural variability will yield more powerful assessments with fewer sampling dates than parameters with a small impact and large natural variability, for which it will be difficult to detect the impact with any degree of confidence [308]. In the latter case, an ecologically realistic interpretation is that the fluctuation in the impacted area is within the boundaries of what occurs naturally, and that it is therefore not a cause for concern [310].

Another problem is that the 'control and impact' design is based on the unrealistic assumption that the two sites would be identical over time in the absence of the activity [305]. However, ecosystems exhibit considerable spatial variability and most natural populations oscillate in ways that are not concordant from one place to another. The BACIPS design ensures that chance temporal fluctuations in either location do not confound the detection of an impact. However, any site-specific temporal fluctuation that occurs between the two sites will be interpreted as an impact, even if it has nothing to do with the disturbance. Alternatively, a parameter in the control may change in the same direction by some other factor, making it impossible to detect the impact. The study would only demonstrate that there are temporal patterns between the control and impact site, but the patterns are not necessarily indicative of an impact [310].

For example, if the abundance of a species is significantly higher at the control site, this may be taken as evidence that the discharge of concentrate from a desalination plant outfall diffuser may have adversely affected the abundance of that species in the impact area. Due to a lack of spatial replication, however, it is uncertain if the observed effect is actually caused by the discharge or some other type of natural fluctuation or anthropogenic perturbation that occurs at one site but not at the other. A decrease in oxygen levels, for instance, might naturally occur in bottom waters due to density stratification in sheltered areas during autumn, and might be responsible for the decline in abundance in the project site. In this case, the change is falsely interpreted as an impact. Alternatively, if the discharge actually causes a decline in species abundance in the project site, and a similar decline is observed in the control site due to naturally decreasing oxygen levels, the impact is masked. This illustrates the problems associated with a lack of spatial replication. Similarly, temporal replication may have detected that the decline in abundances caused by oxygen levels does not coincide with the project start-up.

The problem of confounding (or 'pseudoreplication') can be overcome by having several replicated impact and control sites. While it is difficult to have replicated impact sites (i.e., several desalination plants in randomly chosen locations on a coastline), there is no reason not to have multiple control sites. These do not have to have identical characteristics and abundances as the impact site, but should adequately represent the range of habitats of the site that might be disturbed [309].

For example, if the outfall of a desalination plant is to be placed onto a marine headlands with mostly rocky areas, a few sandy patches and strong currents, the controls must be placed at random in similar locations. It is usually assumed that an outfall has only a local effect on the surrounding few hundred meters, so that controls would typically be sites at the same headlands but outside the impact area. In order to detect an impact, the temporal pattern of a parameter in the impact site must differ from the range of patterns in the set of control sites from 'before' to 'after' the start of the perturbation. If the estimated scale of the impact is wrong, and the outfall causes a change in a parameter over the entire headland, the sampling design would not detect it as all controls would be affected. To overcome this possibility, sampling at two scales, i.e. sites at the headland and other headlands along the coast, would be necessary.

For illustration, baseline monitoring for the Gold Coast SWRO project was carried out over 18 months at four impact sites around the diffuser at the edge of the designated mixing zone, at four reference sites 500 m to the north and at four sites 500 m to the south of the diffuser [178]. Baseline monitoring for the Sydney SWRO project was carried out over 24 months at two impact sites within the designated mixing zone, at two sites located just outside the mixing zone (80 m), at two nearby references possibly still within the zone of influence from the plume, and at one far reference [61].

To conclude, sufficient temporal and spatial replication is required to increase the statistical power of the monitoring studies and achieve a given level of confidence in the estimate of the impact size. However, a study must often be planned in the absence of sufficient preliminary data that would permit an estimation of the number of sampling dates (temporal replicates) and control sites (spatial replicates).

Monitoring in the 'before' period entails assembling, evaluating and presenting data of the relevant environmental properties of the project area before construction, including any other existing levels of degradation or pollution (cf. page 155). One objective of the pre-impact studies is to provide a characterization of the abiotic properties and the biotic resources in the area. For the biotic resources, the minimal objective is to describe:

- *what* marine life can be found in the environment by providing an *inventory list* of species highlighting the dominant, rare and endangered species, and by providing an estimate of the *biodiversity* in the area,
- *where* the main species and habitats can be found by providing *habitat maps*, and
- *how* the structure of assemblages changes over space and time by *univariate and multivariate analysis* of primary (abundance, biomass) and derived variables (biodiversity indices) between impact and control sites, and 'before' and 'after' periods.

The descriptive data need to be converted into a judgment about the sensitivity of the flora and fauna and the relative importance of different regions on the seafloor [311, section 8.4]. The second objective of the pre-impact studies is to serve as a baseline for estimating the magnitude of the impact in the period after the perturbation has begun.

Monitoring in the 'after' period (operational monitoring) is the continuation of baseline monitoring during construction, commissioning and operation of the project in order to assess the accuracy of predictions, to detect new impacts (effects monitoring, section 8.3.2), and to ensure that regulatory requirements and quality standards are being met (compliance monitoring, section 8.3.3). In general, the same survey techniques, sampling sites and schedules as established during baseline monitoring should be used to allow for a comparability of the results, unless modifications are necessary because of methodological or technical problems or in the light of new information.

Obtaining an adequate number of sampling dates in the 'before' period is crucial since additional samples can no longer be obtained once the perturbation begins. However, in many situations, the baseline studies will be rather abbreviated for a variety of reasons [308]. Baseline studies for the Perth project, for example, were constrained to using mostly existing data due to significant time pressure (cf. section 8.1). Generally, the greater the number of replicates, the greater the probability of distinguishing putative impacts from natural variation. Temporal replication should preferably involve a non-regular frequency of sampling to avoid coincidences with natural cycles [312].

Baseline studies typically require one or two years of monitoring before project implementation. If project implementation and accompanying operational monitoring studies are delayed for some reasons, additional baseline studies may be needed to ensure that the baseline data still represent an adequate estimation of the environmental state in the impact and control area, to which the operational monitoring data can be compared. Operational monitoring is typically carried out in similar time periods as baseline monitoring or longer. For example, two years of baseline and at least three years of operational effects monitoring is conducted for the Sydney SWRO project [61].

For discharges from desalination plants, it may be desirable to estimate the spatial extent of effects from the point source. This is typically achieved by sampling along a gradient of distance away from the outfall. Knowledge of the exact location where the structure will be situated is crucial for the correct placement of the sampling grid in the impact area and may require some preliminary studies.

8.3 Marine monitoring framework for desalination plants

The information requirements of EIA studies, and the scope of the accompanying monitoring studies, will depend on the size, nature and *location* of the desalination project. Because of the diversity and complexity of marine ecosystems, there is no standardized, universally applicable technique for monitoring ecological impacts [313]. The scope of the EIA and specialist studies should have been determined during scoping (page 153). The main components and general scope of a marine monitoring programme for desalination projects are described in this section (Figure 27)[a].

8.3.1 Preliminary studies

The initial input usually comes from exiting information sources (literature, maps, databases, etc.) or information provided by locally interested parties (recreational divers, fishermen, etc.). Existing information is often limited or covers a much coarser area. It can however provide useful general information on the environmental setting in a certain sea region, like water mass characteristics or prevailing habitat types that will likely also occur in the project site. A pollution source survey should be carried out to collate information on discharges from existing sources in the vicinity of the plant [61]. This is relevant to identify potential cumulative impacts on the environment, but also environmental considerations for the desalination process and the pretreatment design.

The second and more important input comes from preliminary studies in the broader project area involving first 'broad brush' inspections of intertidal areas; or divers, underwater cameras or side scan sonars in subtidal areas. The objective of preliminary studies is to identify characteristic features within the broader area which will help to identify suitable locations for the plant's intake and outfall, which will become the 'impact' sites

[a] Monitoring in this specific context refers to the living and non-living *environmental* resources. It should be noted that an EIA typically addresses all potential impacts of projects or activities on '*man* and environment', also including *socio-economic* implications and *public health* implications where necessary (cf. page 150). Except for projects with major public health risks and socio-economic perturbations, an EIA will typically rely on existing and readily available data, as it is time consuming and expensive to generate new socio-economic and health data.

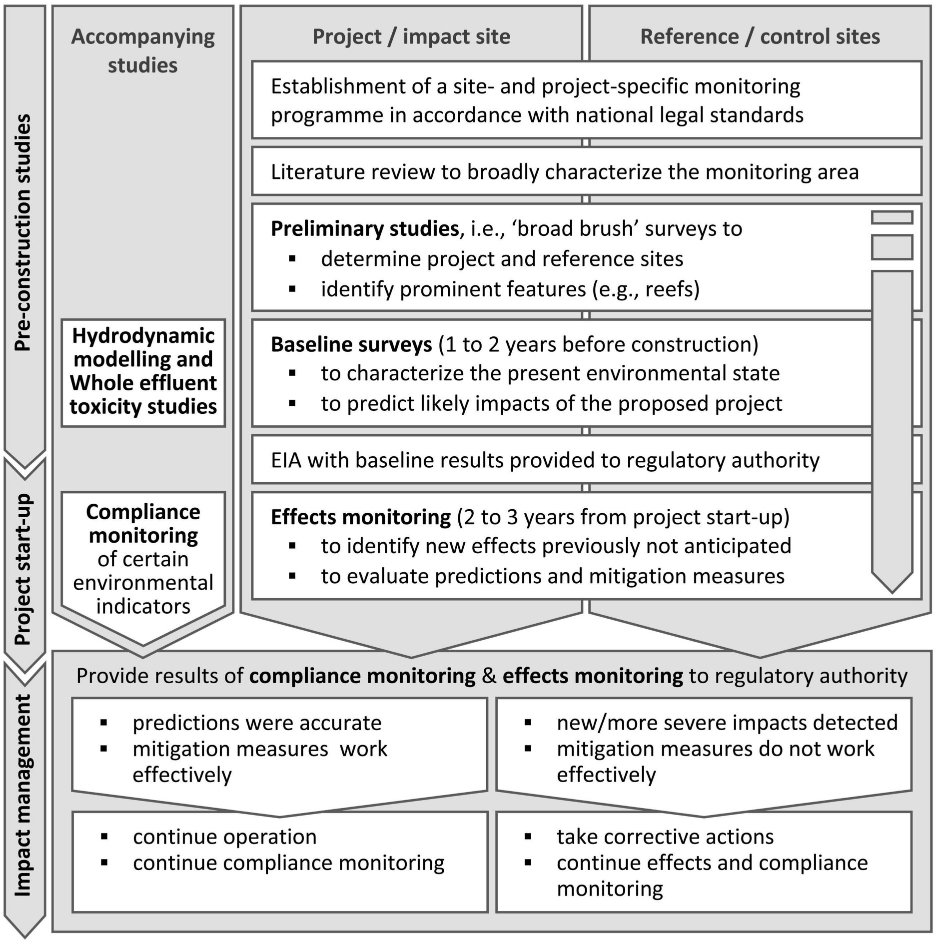

Figure 27: Outline of a monitoring programme for desalination projects.

in the following detailed surveys. The preliminary studies will also facilitate the identification of suitable 'control' sites and planning of effective monitoring designs with regard to the number of temporal and spatial replicates, transect or grid stations, and the best combination of qualitative and quantitative sampling techniques.

8.3.2 Baseline and effects monitoring studies

Seawater

Seawater quality monitoring has the objective to characterize the intake water quality including seasonal variability with regard to *oceanographical, chemical and biological* parameters. The information serves as a baseline for effects monitoring and can provide desalination plant engineers with information on water quality conditions to determine a robust pretreatment process [61] and effective outfall design.

The relevant *oceanographical* parameters are salinity, temperature, density, pH, turbidity, dissolved oxygen (DO), current direction and velocity, water depths and tidal patterns [314, Table 37]. They are typically measured in-situ by shipborne sensors, or alternatively by stationary buoys or autonomous underwater vehicles that provide continuous data over an extended period of time. For instance, two solar-powered buoys provided in-situ measurements of salinity, temperature, turbidity, DO, nitrogen, phosphate and chlorophyll-a for a desalination project in California [315], and three buoys were deployed to measure salinity, temperature and DO for the first Perth project [69]. For the second Perth project, stationary measurements also included pressure sensors for tidal variations, and acoustic doppler current profilers for current profiles at water depth intervals through the water column. A gliding autonomous underwater vehicle was additionally deployed to continuously monitor the sea region over a wider area, and rhodamine dye tracer studies were carried out to track water mass movements in the discharge location [316]. Turbidity monitoring to detect short-term construction impacts on water quality relating to sediment disturbance can involve in-situ optical or acoustic backscatter sensors. The method cannot differentiate between a change in concentration and a change in particle size, and particles from organic or inorganic origins, which can only be achieved through direct sampling [317].

Chemical analysis typically includes the major nutrients (phosphate, silicate, nitrate, nitrite, ammonia), DOC, TOC and chlorophyll a. Nutrient studies in tropical waters often require the detection of compounds at extremely low levels and consequently a greater vigilance than might be needed for water samples from temperate regions [318]. Depending on the results from the pollution source survey (section 8.3.1), a chemical analysis of priority pollutants and trace elements may be conducted. For instance, a full chemical analysis was carried out for the Tampa Bay SWRO plant in Florida, including 200 compounds which may be present in the feed water and would be enriched in the concentrate [75]. Seawater quality monitoring for the Ashkelon project in Israel included a metal analysis of water and sediment samples (Cu, Cr, Cd, Pb, Ni, Zn, Hg, Fe, Pb, V) besides a comprehensive nutrient analysis [80]. For a chemical analysis, representative water samples must be taken, preferably collected at the same sampling stations where oceanographical measurements are carried by a research vessel.

Seawater monitoring should also entail a survey of the *biological* resources which may be potentially entrained within the seawater intake, such as bacteria (microbial parameters such as heterotrophic plate counts), eggs and larvae, phyto- and zooplankton, and smaller nektonic species, such as small fish or invertebrates. As an indicator of phytoplankton, chlorophyll a can be measured in situ. Representative water samples for phyto- and zooplankton can be derived by plankton net tows with a research vessel. The data can be used to estimate entrainment impacts caused by the intake.

Seafloor

Seafloor monitoring has the objective to classify and map the marine landscape with regard to *bathymetry and topography, sediment types and composition, and distribution of species and habitats* in the area. The information serves as a baseline for effects monitoring and can be used to identify the intake and outfall locations and pipeline routes. If the intake and outfall pipelines are to be drilled from an onshore site, information on the substratum sediments is additionally required. Surveys usually combine acoustic remote devices, underwater cameras and sampling (Table 38).

Table 37: Monitoring techniques for seawater parameters.

Method	Parameters measured	Description of monitoring device
CTD profiler	conductivity (salinity), temperature, depth, and density (calculated), often also carries an array of other sensors measuring, e.g., light transmission, fluorescence, dissolved oxygen content, optical backscatter	seasoar (or Batfish): towed platforms with wings, which give it the ability to 'fly' up and down in the water column while being towed behind a ship, or attached to neutrally buoyant devices (usually for large scale currents): drifting along a specified pressure surface, periodically surfacing to transmit CTD and position data
ADC profiler	acoustic doppler shifts to measure current velocity over a range of depths through time	typically moored at the seafloor or attached to the bottom of a ship
Current meters	current velocity through time	moored instruments (mechanical, electromagnetic, or acoustic measurements)
Dye tracer studies	water mass movements	e.g., 20% rhodamin dye solution over 30 minutes at a rate of 0.3 l/min
Echo sounder	wave height and period	typically moored, upward looking
Gauges	water level data	moored instruments
Water samples	major nutrients (phosphate, silicate, nitrate, nitrite, ammonia), DOC, TOC, chlorophyll a, chemical analysis (trace elements, priority pollutants), microbial parameters	discrete collection of water from a specific time, position and depth, using a single bottle attached to a rosette or water drawn from another non-contaminated sample
Plankton net tows	phytoplankton and zooplankton data diversity and abundance	horizontal tow of a plankton net, which consists of a circular ring opening with a conical shaped net (mesh sizes <1000 µm) and a jar at the end

Table 38: Monitoring techniques for seafloor parameters.

Method	Parameters measured	Description of monitoring device
Multibeam echo-sounder (swath)	high-resolution mapping of bathymetry	autonomous underwater vehicle or attached to ship hull, measures the time it takes for a pulse of sound emitted from the device to travel to the ocean floor and back
Side-scan sonar	detailed mapping of bottom topography, objects and texture (mud, sand, gravel)	fan-shaped pulses in a frequency range of 100-500 kHz or higher are emitted towards the seafloor at a wide angle perpendicular to the path of the sensor, which is towed by a surface or submarine vessel, or mounted to a ship hull, and creates an 'image' of the seafloor by which topographical features and objects can be identified
Sub-bottom profiler	profiles and textures of the shallow structure of the seabed	low frequency echo-sounder, towed or mounted to a ship hull
Visual surveys	habitat type and coverage (e.g., seagrass or macroalgae), sediment type (e.g., shell banks, rocks, etc.), identification of larger benthic species on the seafloor (epifauna) and demersal fish species, visible disturbances of the habitat or sediment	underwater video along transects and/or photography at individual sampling stations, by divers or remotely-controlled devices in greater water depth
Sediment samples	infauna and smaller epifauna: total number of species, biodiversity indices, species abundances and biomass, dominance structure and community analysis, sediment analysis: grain size distribution, organic carbon content, pollutants	core or grap samples, from a research vessel, or manually in intertidal areas, or by divers in subtidal areas
Trawls	epifauna: total number of species, biodiversity indices, species abundances and biomass, dominance structure and community analysis	in sandy areas devoid of reefs, larger rocks, seagrass or macroalgae stands, and where the impact by the sampling method can be justified, e.g., in areas already impacted by fisheries, and water depth permitting

Bathymetric and topographic surveys in shallow waters are usually carried out by remote acoustic sensing devices mounted to or towed by a research vessel, typically multibeam echosounders (swath), side-scan sonars, and/or sub-bottom profilers. An image of the seafloor is thus created by which topographical features, underwater objects and the texture of the seafloor surface (mud, sand, gravel, rock) can be identified. These methods can be used to classify the different habitats on the seafloor which usually have clear demarcations, such as between reefs or seagrass beds. For assessing near-field changes in seabed morphology such as scour and deposition around installations, as for example caused by structures placed on the seafloor, a high-resolution bathymetry survey of the spread of sediment types across the site can be carried out. The footprint of scours mainly remains local to the installations. An initial survey should be carried out immediately after construction and in longer intervals thereafter [317].

The main *sediment* parameters are grain size distribution, geochemical properties and organic carbon content. If the pollution source survey indicates that pollutant levels may be increased in the project area, a chemical analysis for pollutants with a tendency for accumulation in sediments should be included (e.g., metals as in Ashkelon). Sediment sampling is often carried out simultaneously with biological sampling. *Benthic species* are subdivided into infauna and epifauna, i.e., those living in the seafloor sediments and those occupying the surface of soft and hard bottom substrates. As a third group, demersal fish assemblages may be included which inhabit the bottom waters at the sediment interface, such as flounders in soft bottom habitats or reef-dwelling fish.

The marine floor surveys often move from 'broad-brush' preliminary surveys, typically involving remote acoustic sensing devices in conjunction with underwater video transects or habitat characterization by divers (section 8.3.1), to more detailed and focused studies. The latter involve small-scale sampling in the project area, i.e., near the intake and outfall and along pipeline routes, as well as within control sites. For example, transects with sampling points were established for the Sydney project in the intake and outlet areas, covering approximately 150 m x 200 m [61]. Duplication of sampling by three replicates is recommended to confirm the representativeness of the samples at each station. Sampling should be repeated at different times of the year to take seasonal variability of species, especially migratory species into account, such as for example the migration of fish into coastal areas for spawning [319].

In areas with soft bottom habitats, samples can be taken by grab or core samplers from research vessels to provide information on the grain size distribution, on pollutants in sediments, or on the species abundance and biodiversity of the infauna. Where grab sampling is not possible, surveys need to be conducted by underwater video or photography, or by diver observations and manual sampling. This may pertain to hard substrates such as very coarse or rocky terrain, reefs or artificial structures. The epifauna in soft bottom habitats can be sampled by means of a trawl or dredge, however, this method is rather invasive and may not be suitable for areas where habitats and species of high nature conservation importance are present, such as seagrass beds.

If the desalination plant is to be co-located to an existing facility, such as a power plant, and will make use of existing intakes and outfalls, pre-existing monitoring data for the power plant may be available and could be used to establish the baseline for the desalination plant. The monitoring should include sea walls and channels [319].

Quantitative samples of biological resources result in species densities per volume or sample area, either giving numbers for individuals or percent coverage, e.g., for plant growth or barnacles. Larger organisms can often be identified and counted on site and left in situ. A photographic record may be obtained from unidentified larger organisms for taxonomic identification. Smaller or unidentified organisms are typically retained in formalin for laboratory identification and counting [319]. The outputs of these inventories are species lists and distribution maps. Distribution maps requires a multivariate analysis of species distributions at the habitat and species level (e.g., seagrass meadows, macroalgae stands, sandflats with macrofauna, etc.), and a univariate analysis of spatial and temporal patterns in density and biomass for the most common species (key species such as *Posidonia* and *Zostera* seaweeds). The spatial distribution data is often integrated into a geographic information system (GIS).

Nekton

Nekton refers to the aggregate of actively swimming organisms in the sea, which includes certain invertebrates such as squid or larger shrimps, fish, or reptiles (i.e., sea snakes and turtles in tropical waters). The construction and operation of a desalination plant may adversely affect these species through impingement of organisms or entrainment of larvae at the intake, or loss of habitat (e.g., spawning and feeding grounds).

Depending on the project and the information requirements of the EIA, it may be necessary to monitor the nektonic species in the broader project area (impact and control sites) in different levels of detail. Existing sources of data should be used if possible. The first step would be to identify if important recruitment, feeding and overwintering areas or migration routes exist within the project area, with particular emphasis on species that are of conservation importance. However, for some sites, there may be either a lack of information or a local concern, so that a monitoring programme may be required.

A *quantitative* fish survey requires careful design, usually a combination of different sampling methods appropriate for the site and species in question, and sufficient replication and coverage to take account of the mobile nature of nektonic populations. Otter trawls are commonly used for demersal fish assemblages and may also catch some pelagic fish (e.g., herring). If flatfish (e.g., plaice, sole) are the primary target species, a larger beam trawl would be more appropriate. Juvenile or small demersal fish are best sampled by a small beam trawl or shrimp trawl. Tows of commercial gear should be of 30-60 minutes duration, while sampling with small trawls should be 5-15 minutes duration, depending on the quantities of fish in the area. In general, useful data may be collected during the spring spawning season for most species, although seasonal fisheries may also necessitate additional sampling in summer and/or winter [320].

Trawling data are given as relative abundance (i.e., catch per unit effort, typically number of fish per hour) and are highly variable by nature. Therefore, any statistical data analysis and interpretation in terms of abundance and spatial distribution of species must use extreme caution. Variance can be reduced by increasing the number of trawls before and after project implementation. Even if the number of spatial and temporal replicates is increased (cf. section 8.2), it may not be possible to actually quantify the impacts (if any) of a desalination project on nektonic species in the project area.

The marine structures of a desalination plant may affect an area that is small compared to the area that is covered by trawling if the study design accounts for sufficient replication. Moreover, most fish species are broadcast spawners and opportunistic predators without well defined feeding areas, so that small-scale habitat losses will unlikely have severe implications at the population level. Some species may nevertheless congregate in certain areas at given times of the year to spawn or feed on particular prey. A disruption to these areas or during these particular times should be avoided.

For example, concentrate from a proposed desalination plant in Spencer Gulf, Southern Australia, could be discharged in the vicinity of an area that is known to exhibit a unique annual spawning aggregation of the giant Australian cuttlefish. While the EIA concluded that impacts on cuttlefish within the reef habitat at the location of the outfall would not be detectable (i.e., negligible) [179], others fear that the discharge poses a potential threat to the unique spawning aggregation and suggest that knowledge of the key egg-laying sites within the breeding aggregation will enable more cautious decision making with regard to large-scale industry of any kind [321].

To conclude, the expenditure and impact of a quantitative fish survey has to be carefully balanced against the knowledge gain of such a study. For some nektonic species, which are of conservation interest, it will also be difficult to establish quantitative data by non-invasive measures. In most cases, a reasonable approach will be to carry out a literature survey in order to assemble *existing quantitative* data where possible and to identify species or habitats of special conservation interest. If existing data are scarce, a *qualitative* survey using trawls or underwater video should be carried out with the objective to produce a list of species that may occur in the target area.

Birds and mammals
Monitoring of birds and mammals has the objective to establish a list of species that may occur in the project area, and to ascertain if a special conservation interest exists for that area. This typically involves a literature review and qualitative survey in the target area during different seasons of the year. The list should include terrestrial birds in the project site on land, seabirds and marine mammals including onshore and coastal habitats up to a seaward distance of 1 km from the outfall by ship-based observations.

8.3.3 Compliance monitoring (indicator approach)

While it is desirable to examine as much as possible in an EIA, it is certainly not possible to investigate all species in all habitats all the time. An EIA is therefore to some extent always implicitly employing an indicator approach [322], for example by focusing monitoring efforts on the abundant macrobenthic species in the area. EIAs also explicitly make use of indicators in compliance monitoring, which refers to the regular measurements of a limited number of indicators which summarize a significant aspect of the state of the environment in order to ensure that regulatory requirements are being met. For example, microbiological indicators are used to summarize the status of bathing waters.

For desalination plants, suitable physical indicators are salinity and dissolved oxygen levels (or temperature for distillation plants). Measuring these parameters at the point of discharge has the objective to ensure compliance with effluent standards, while measurements at the edge of a regulatory mixing zone (e.g., by a moored buoy) ensures compliance with ambient water quality standards. When selecting a bioindicator, relevant criteria are the relative abundance, ecological importance (e.g., sea urchins in kelp beds, polychaetes and bivalves in soft-bottom habitats) and socio-economic importance (in terms of fisheries and public health) of a species. Jones and Kaly [322] advise against a rigid set of criteria build into regulatory frameworks for selecting an indicator and stress the need to consider a variety of taxa from different trophic levels.

Developing a bioindicator approach for assessing the impacts caused by the discharges from a desalination plant would involve the following steps:

- establishment of a quantitative baseline survey to obtain information on the relative abundance and ecological importance of the species in the area (section 8.3.2),

- characterization of the desalination concentrate through whole effluent toxicity testing (section 8.3.5), using selected local species,
- selection of an indicator which is abundant in the area, as determined during the baseline studies, and which is sensitive to changes in environmental conditions caused by the concentrate discharge, as determined during toxicity testing,
- determination of the spatial and temporal evolution of the concentrate plume in the discharge area through hydrodynamic models (section 8.3.4), and
- development of a monitoring approach and identification of monitoring stations for the selected indicators based on the range of the discharge plume.

For example, the monitoring of indicators can take place as part of the effects monitoring surveys (tracking the distribution of indicator species over time in their natural habitat) or in defined locations and experiments (e.g., using buoys with settlement panels). The use of indicator species may be particularly useful to monitor the environmental state at regular intervals throughout the life-time of the project after the end of the effects monitoring studies, which are usually limited to 2-3 years after project implementation.

8.3.4 Modeling studies

Hydrodynamic modeling studies are usually part of the baseline investigations (Figure 27). They have the objective to predict changes to currents and flows caused by the intake of large quantities of seawater, and to predict the mixing behavior of the reject water plume and any residual chemicals in the receiving water body. By estimating the spatial and temporal extent of the plume, potentially affected habitats in the vicinity of the outfall can be identified, and the outfall location and design modified if necessary.

The mixing behavior of an effluent mainly depends on (i) the oceanographic conditions in the receiving water body, (ii) the discharge practice and (iii) the properties of the reject stream. Therefore, hydrodynamic models usually have to integrate a large number of variable parameters and require detailed information on the prevailing oceanographic conditions in the discharge site and on the planned outfall design. By using different variations of these parameters, dilution scenarios can be developed under a number of theoretical conditions, including worst case (quiescent) and a range of normal conditions [61], such as tidal cycles or seasonal currents. Brine discharge modeling should adequately cover the near field and far field processes (section 2.4, page 52), which may require the coupling of two separate models. While near-field mixing is dominated by the outfall design, far-field mixing is dominated by ambient processes [153].

It should be demonstrated that the models can accurately reproduce all key features known to affect the temporal and spatial evolution of the brine in the study area. First and foremost, model results should be validated against key oceanographic processes and parameters relevant to the study area using representative field data from baseline monitoring. For example, the modeled salinity and temperature values should adequately reflect horizontal changes in the project area, and the existing depth-profiles and density stratification in the water column. Another option is to run different models separately and compare the results which, if similar, will increase the confidence in the results. A third option, which is particularly useful to model near field processes, are miniature models in the laboratory [153]. Alternatively, tracer experiments with dye can be carried out in the project area to evaluate the model's ability to reproduce advection and dispersion [316]. Finally, it is important that the quality of the models and the modeling results are reviewed and accredited by an independent expert group or institution [61].

8.3.5 Bioassay studies

Before additional bioassay studies are carried out, the risks associated with the discharge of the concentrate and residual chemicals should be evaluated using existing data. Risk characterization is basically a three step process involving [323]:

- Exposure information: Prediction of environmental concentrations (PEC) of all chemical residuals and salinity levels at the edge of the mixing zone based on the project-specific hydrodynamic modeling studies.
- Effects assessment: Establishment of the predicted no effects concentration (PNEC) of all substances based on existing ecotoxicity data sources and additional bioassay studies where necessary. The data set should at least comprise short-term acute toxicity tests with organisms from three trophic levels, and preferably also at least one long-term chronic test with the most sensitive species[b].
- Risk characterization: If the PEC exceeds the PNEC, a potential risk to the environment must be anticipated. If the PEC and PNEC are associated with a high degree of uncertainty, further studies may be necessary to refine the PEC/PNEC ratio. If the PEC still exceeds the PNEC after that, impact mitigation measures are necessary.

Whole effluent toxicity

If further testing is necessary, tests should preferably be whole effluent toxicity (WET) tests using a range of marine indicator species with different sensitivities, some of which should be known to be present in the desalination plant location [61]. The advantages of WET tests are that the testing effort is considerably reduced and that synergetic effects between salinity and different chemicals are taken into account. However, as bioassay studies are typically part of the baseline studies, representative solutions must be obtained from a pilot plant or created by mixing and dilution of the single components.

WET testing was for example undertaken for the Perth, Sydney, Gold Coast and Olympic Dam SWRO projects in Australia following the Australian and New Zealand Guidelines for Fresh and Marine Water Quality [324]. Based on the WET tests, a species protection trigger value (SPTV) is calculated, which is the safe dilution ratio for the concentrate that protects a certain percentage of the species from adverse impacts (Table 39). A species protection level of 95% is usually adopted for slightly to moderately disturbed ecosystems, and 99% for ecosystems of high conservation value.

The most extensive WET tests were carried out for the Olympic Dam SWRO project. Basted on WET tests with 15 species from four trophic levels, it is predicted that a dilution of 45:1 will protect 99% of the marine species in the area, corresponding to a salinity increase of 0.7 units above ambient. Based on the hydrodynamic modeling studies, this dilution will be achieved within 300 m from the outfall in 90% of all times. 100% species protection at all times would be achieved within 3.9 km from the outfall [85:1 dilution or salinity increase of 0.4 units above ambient, 179]. The studies for the Sydney project showed that salinity was the key source of toxicity of the whole effluent [63].

A similar methodology for testing the long-term salinity tolerance of marine species was applied for two SWRO projects in California [325]. Based on hydrodynamic modeling, the salinity level in the middle of the zone of initial dilution (ZID, defined as the area within 330 m from the point of discharge) in 95% of the time was predicted. A long-term

[b] For most substances, the pool of data is very limited. In these circumstances, empirically derived assessment factors must be used to establish a PNEC [for further details see 323]. For some common substances, such as chlorine, a PNEC may have already been established by risk assessments carried out by regulatory authorities or independent expert groups [e.g. 191, see also page 66].

Table 39: Whole effluent toxicity test data: The species protection trigger value (SPTV) is calculated from a range of test species and gives the minimum dilution ratio that should be achieved at the edge of the mixing zone for a given species protecting level (SPL). The SPTV is compared to the actual dilution ratio that has been predicted for or is actually achieved by the diffuser.

Plant	SPL	SPTV	Diffuser dilution ratio	WET test species
Perth [176]	95% 99%	12.3:1 15.1:1	45:1	Tests at commissioning and after 12 months of operation 72-h macroalgae germination (*Ecklonia radiata*) 72-h macroalgae growth test (*Isochrysis galbana*) 48-h mussel larval development (*Mytilis edulis*) 28-d copepod reproduction test (*Gladioferens imparipes*) 7-d larval fish growth test (*Pagrus auratus*)
Sydney [177]	95%	30:1	30:1 dilution ratio at the edge of the near field (50-75 m) equal to salinity variations of 1 unit above ambient as determined by modeling	five target organisms: algae, crustaceans (prawn), molluscs (oysters) echinoderms (sea urchin fertilization and larval development), chordates (fish)
Gold Coast [178]	95%	9:1	47:1 minimum dilution in 60 m distance from the diffuser (edge of mixing zone) determined by modeling; validation during start-up confirmed a dilution in excess of 9:1 at the edge of the mixing zone	6 species from more than 3 trophic levels representative of the local ecosystem, targeting sensitive early life cycle stages (fertilization, germination, larval development and growth): Acute microtox (bacterium *Vibrio fischeri*) 72-h microalgae growth inhibition (*Nitzschia closterium*) 72-h macroalgae germination (*Ecklonia radiata*) 48-h rock oyster larval development (*Saccostrea commercialis*) 72-h sea urchin larval development (*Heliocidaris tuberculata*) 7-d larval fish imbalance (*Pagrus auratus*)
Olympic Dam [179]	99% 100%	45:1 85:1	45:1 dilution within: 0.3 km (90% of time) 1.1 km (99% of time) 2.2 km (100% of time) 85:1 dilution within: 1.1 km (90% of time) 2.8 km (99% of time) 3.9 km (100% of time) 45:1 dilution would be achieved in 30% of the time at the edge of the near field mixing zone (100 m); the salinity increases for the dilution ratios of 45:1 and 85:1 would be 0.7 and 0.4 units above ambient, respectively	15 species from more than 4 trophic levels representative of the local ecosystem, including acute and chronic tests with early life cycle stages, juveniles and adults: 72-h microalgae chronic growth rate inhibition test (*Nitzschia closterium* and *Isochrysis galbana*) 72-h macroalgae chronic germination success (*Ecklonia radiata* and *Hormosira banksii*) 48-h chronic copepod reproduction (*Gladioferens imparipes*) 96-h acute prawn post-larval toxicity test (*Penaeus monodon*) 21/28-d juvenile/adult prawn growth (*Melicertus latisulcatus*) 7-d sub-chronic crab larval growth test (*Portunus pelagicus*) 48-h sub-chronic oyster larval development (*Crassostrea gigas* and *Saccostrea commercialis*) 72-h sea urchin sub-chronic fertilization success (*Heliocidaris tuberculata*) 96-h acute fish larval imbalance and mortality (*Seriola lalandi*) 7-d sub-chronic fish larval growth test (*Seriola lalandi, Pagrus auratus, Argyrosomus japonicus*) chronic developmental and hatching tests (*Sepia apama*)

biometric test with 18 species in a single aquarium over a period of 5 months was carried out to investigate chronic effects at this salinity. In addition, salinity tolerance tests were carried out over a range of salinities to investigate if marine organisms will be able to survive periodic extreme (worst case) salinity conditions. Three local species which are known to have the highest susceptibility to salinity stress were used (purple sea urchin *Strongylocentrotus purpuratus*, sand dollar *Dendraster excentricus*, and the red abalone *Haliotis rufescens*). The tests produced no indication of potential negative effects of the proposed discharge. Methods for measuring the acute and chronic toxicity of effluents to marine organisms have also been established by the U.S. EPA [326, 327].

8.4 Assessment of data

The assessment of environmental impacts is usually a two tier approach (Figure 28). Tier one is the evaluation of stressor sources in terms of *significance*, using criteria such as *intensity, space and time* (chapter 5). Tier two is the assessment of environmental features (receptors) in the project site, based on the descriptive monitoring data, in terms of *sensitivity*, using criteria such as *tolerance*, *importance* or *recoverability* of species or habitats. An evaluation of what is significant or sensitive is not entirely possible without taking the counterpart into account. Impact mitigation measures can either aim at reducing the stressor level (e.g., technical modifications), or at separating stressors and receptors (e.g., location changes), or both (chapter 6). For example, the diffuser can be modified to reduce the area around the outfall where salinity levels are increased, or the siting of the outfall can avoid sensitive species or habitats. This section deals with defining criteria for the evaluation of sensitivity of species and habitats.

Sensitivity partly depends on the *tolerance* of a species or habitat to adverse external factors (stressor sources) that may cause damage or death. However, a species or habitat only becomes *vulnerable* when the external factor is likely to happen [328]. For instance, most benthic species will have a high sensitivity to physical impact such as construction impacts but are only vulnerable if activities such as the trenching of pipeline routes are being undertaken where the species are present, i.e., in the case of co-occurrence of species and stressor source. It also has to be taken into account if a species is mobile (such as demersal fish species), semi-sessile (such as sea urchins) or sessile (such as seagrass), i.e., if it will likely be able to escape the stressor source or not.

Sensitivity also partly depends on the *recoverability*, i.e., the ability of a species or habitat to return to a state close to that which existed before the development activity. Recovery may occur through re-growth, re-colonization by migration or larval settlement from undamaged populations, or re-establishment of viability [328]. In many cases, a recovery is possible, either partially or completely, but the question is how long does it take for a species, community or habitat to recover. The lower and the slower the recoverability, the higher the sensitivity. For example, a seagrass meadow may only slowly recover after being impacted by turbidity and increased salinity, and recoverability may even be incomplete if re-colonization from undamaged populations is limited.

Another indicator for sensitivity is *importance* for nature conservation. Species or habitats have a high importance for nature conservation if they are listed as protected, if they are endangered, rare or very restricted in their distribution, if a high proportion of the regional (or world) population is defined to a certain area (locally high abundance), if the area has a high species richness (locally high biodiversity) or if keystone species or habitats are present. For example, *Posidonia* seagrass meadows are a priority habitat in the Mediterranean region according to the European Habitats Directive.

A main purpose of assessing species and habitat sensitivity is to identify suitable locations for the intake and outfall structures, i.e., habitats with a low sensitivity to adverse effects. Low sensitivity in the best case means a low *importance* for nature conservation in conjunction with a high *tolerance* and high *recoverability*. However, ecosystems near the coast will often have a moderate to very high sensitivity to external factors arising from human activities, so that intakes and outfalls may have to be placed further offshore and pipelines may be constructed by trenchless techniques from a site on land.

A methodology for assessing species and habitat sensitivity has been developed by the Marine Life Information Network for Britain and Ireland [328, MarLIN]. The MarLIN approach proposes an ordinal scale with seven categories for evaluating the sensi-

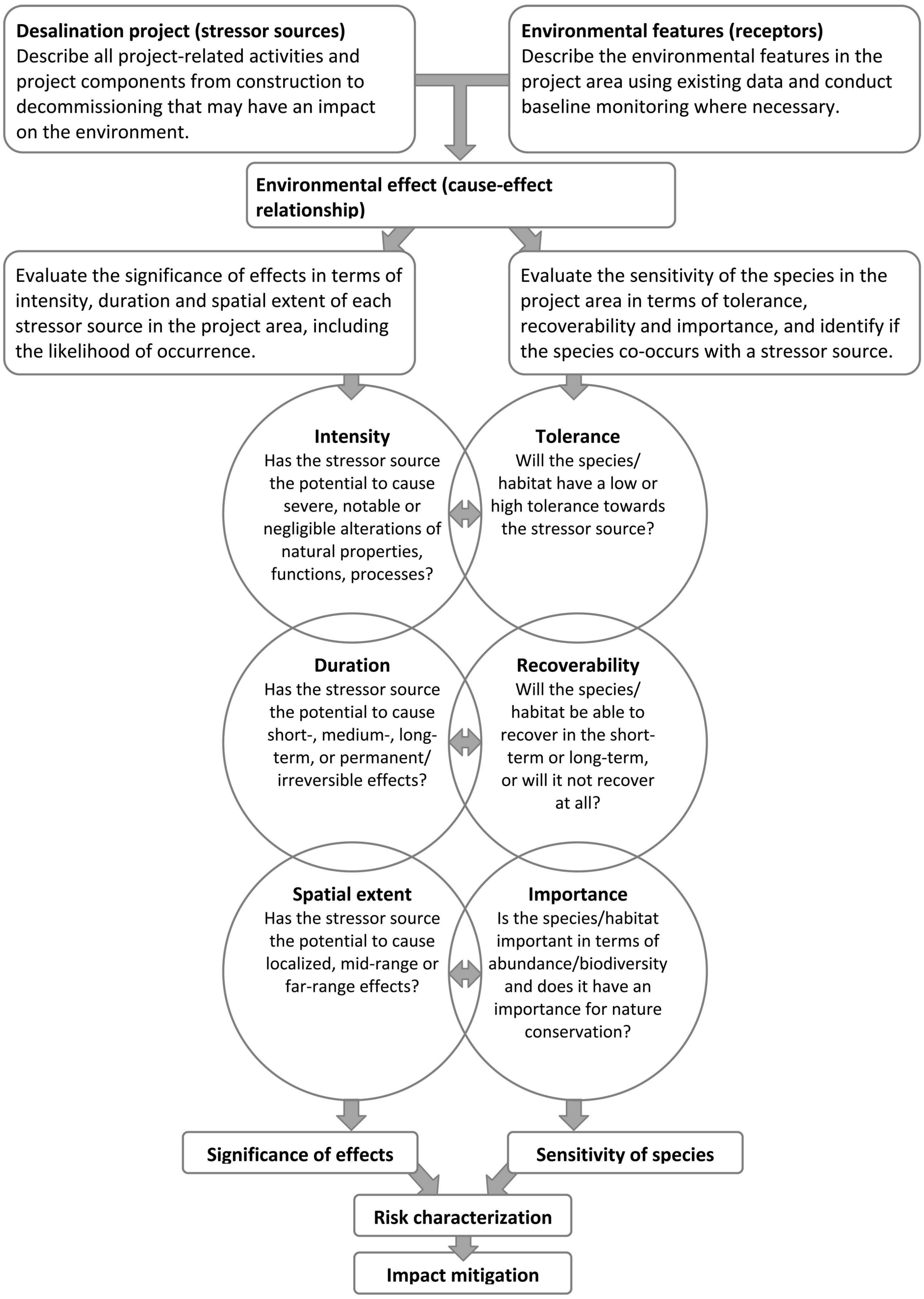

Figure 28: Two-tier assessment approach for impacts and monitoring data.

tivity of species/habitats (not relevant, not sensitive, very low, low, moderate, high and very high sensitivity) towards a disturbance, based on the *tolerance* and *recoverability* of the species/habitats in question. The definition of each sensitivity category is logical and coherent. However, the decision problem of selecting an outfall location has a discrete decision space in which the answers to the question can only be ‘no’, ‘yes’ or a conditional ‘yes’ with impact mitigation. A distinction between ‘very high’ and ‘high’ or ‘low’ and ‘moderate’ sensitivity is thus not very useful for decision making. It allows for a ranking of habitats, but in the end a division line must be drawn between what is acceptable and what is not. The 7-step ordinal scale was therefore reduced to a binary scale (‘yes’ and ‘no’) for both criteria and a third criterion (*importance*) was introduced (Table 40). The three criteria were then formally integrated into an overall rating for sensitivity (Figure 29).

Table 40: Criteria and their definition for the evaluation of sensitivity [adapted from 328].

Sensitivity rating		Description
Tolerance	low	adversely affected (death of species, partial or complete destruction of habitats)
	high	mildly affected (physiological stress, reduced fecundity or growth, but no deaths or destruction of habitat expected)
Recoverability	low	not expected to fully recover over the project life-time or thereafter
	high	expected to recover rapidly or over a limited period of time (<5 years)
Importance	high	above average abundance of rare, endangered or listed species, and/or extraordinary aggregation (locally high abundance) of other species, and/or a high species richness (locally high biodiversity), and/or presence of keystone species or habitats, and/or general ‘pristine’ status of the environment, which results in a high conservation interest of the area
	low	species/habitat without conservation interest, i.e., with average or below average abundances, or habitats which have already been moderately to strongly impaired

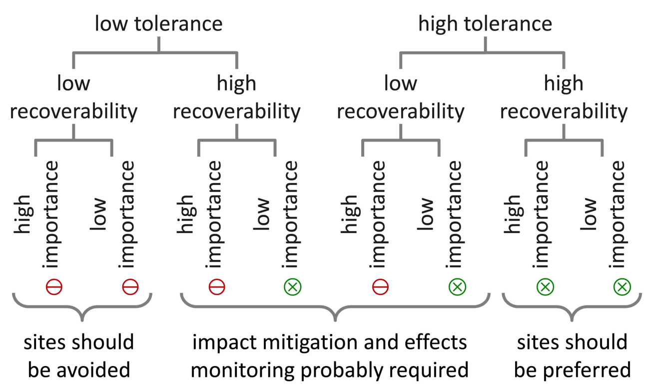

Figure 29: Evaluation of sensitivity of marine ecosystems and organisms: Aggregation logic for the criteria ‘tolerance’, ‘recoverability’, and ‘importance’, and overall rating (⊖ high sensitivity, ⊗ generally low sensitivity).

Species/habitats which are *sensitive* to impacts are consequently those which fulfill the following criteria:

- All species/habitats which are adversely affected (death of species or habitat destruction) *and* which are not expected to fully recover over the project life-time or thereafter.
- Species/habitats with a particular conservation interest,
  - which are adversely affected, even if they have a high recoverability.
  - which are only mildly affected, but may have a low recoverability.

Species/habitats which are *not sensitive* to impacts are consequently those which fulfill the following criteria:

- All species/habitats which are 'only' affected through physiological stress, reduced fecundity or growth *and* usually recover rapidly, or within a few years of the impact.
- Species/habitats without a particular conservation interest
  - which are adversely affected, but have a high recoverability.
  - which are only mildly affected, and have a low recoverability.

8.5 Summary and conclusions

Although distillation plants have been operating for some decades in certain sea areas, like the Arabian Gulf, the process of SWRO is comparatively young (chapter 1). Large projects and accompanying monitoring programmes are only now being implemented. The longest reported monitoring programme of a SWRO project, implemented in 2006, just looks back on two years of operation. Although a few EIAs of desalination projects have recently become available, these reflect the state of knowledge from the predictive process, while results from the postdictive process, both over time and including other inputs in a particular region, are only now beginning to be investigated.

The importance of operational monitoring cannot be overemphasized, which is also illustrated by the following case study. Ambrose et al. [313] compared the actual impacts of the cooling water discharges from a nuclear power plant in Southern California, established by a 15-year monitoring programme, to predictions made in the EIA which had been generated in three different ways. The comparison showed that (i) almost all of the testimonies of scientists before the permitting agency, which were based on professional judgment with little scientific analyzes, were wrong. (ii) The accuracy of the final environmental statement, based on standard assessment methods at that time, was mixed but generally not too high. (iii) The predictions of the marine review committee, based on a comprehensive baseline study over several years, were the most accurate but also showed inaccuracies. Although a clear correlation between effort and accuracy of the predictions seems to exist, the lesson learnt here is that EIAs cannot predict with complete confidence what will happen in the environment, even if considerable resources are dedicated to monitoring [313]. EIAs, like other observational studies, are likely to be 'messy' even after a conscientious effort to apply the appropriate techniques and mathematical statistics [329]. The impacts predicted in EIAs are not always the actual impacts, although they become the *de facto* impacts if there is a lack of follow-up studies [313].

As mentioned in the preamble of this thesis, the number of publications discussing the *potential* for environmental impacts of desalination facilities has been steadily increasing over the last few years. These remain no more than *de facto* impacts in the absence of more rigorous follow-up studies than is presently the case. The National Research Council attested a surprising paucity of useful *experimental data*, either from laboratory tests or from field monitoring in 2008 [5]. This has to change in order to prove whether or not the predicted potential impacts of desalination plants are accurate. It will only be possible to detect impacts with the adequate monitoring designs. Reputable journals would reject results which were derived with less than good scientific practice in 'academic' field experiments. The same standard must apply for EIAs, although they are 'applied' science, unless the entire assessment should become a random process [310].

Regulatory agencies may still be reluctant to require rigorous operational monitoring studies and project developers are understandably opposed to funding it [313]. However, there is an increasing tendency to regulate new developments worldwide under the requirement that predictions will be tested by measuring the real impacts by scientific means, and by imposing project modifications if impacts are found to be different from those predicted. Section 8.2 outlined that spatial and temporal variability could falsely be interpreted as an impact without sufficient replication. Adequate monitoring could therefore also be understood as an 'insurance' against unwarranted claims [310].

In this context, it is also noteworthy that both effects monitoring and compliance monitoring only allow for reactive impact management. It should therefore be in the interest of all parties (and of the environment) that management responses are established in case that unexpected or more severe impacts are detected during effects monitoring, or in case that trigger values are exceeded during compliance monitoring.

Effects monitoring also serves to produce much relevant fundamental research, which is of particular value to industries which are not involved in one-off developments [319]. The desalination industry can thus learn from each experience to minimize impacts for the next development. As mentioned in the introduction of this chapter, comprehensive environmental monitoring programmes are underway for several large Australian SWRO projects, which will provide valuable results in the near future.

Multi-criteria analysis as a decision support system in EIA

9

9.1 Introduction

EIA studies, as outlined in chapter 7, are part of the permitting process of large infrastructure projects, including new desalination projects. Depending on the size and complexity of the project, the EIA may have to consider different site and process alternatives, and investigate a long list of potential environmental concerns. In chapter 5 (Tables 21 to 34, page 108ff.), about 150 concerns of desalination projects were identified, of which about 20 were classified as being of high priority for EIAs (Table 35, page 120).

To cover the wide range of concerns, different specialist studies are usually carried out and summarized in the EIA. A vast body of quantitative data and qualitative information is thus compiled, which results in lengthy reports with numerous appendices, and extensive evaluation tables, which are usually unsuitable for direct evaluation [295]. For example, the recently completed EIA of the Victorian desalination project in Australia covered 1600 pages and the volumes stack nearly 1.5 m high.

Moreover, different government agencies, stakeholders and the wider public usually participate in the permitting process of a new desalination project. It is therefore necessary to communicate the results of the EIA to the decision makers, and conflicting preferences about the project need to be balanced in decision making. This requires a structured and transparent approach. As a single and objectively best solution often does not exist, the process of environmental decision making has been described as a conflict analysis characterized by environmental, social, economic and political value judgments, which is essentially a search for an acceptable compromise solution [330].

The decision making process in an EIA can be facilitated by a formalized decision support tool. One such tool is multi-criteria analysis (MCA). In contrast to traditional decision support tools such as cost-benefit analysis (CBA) often used in economics, MCA allows for a comparison of alternatives by using non-monetary and non-metric (i.e., qualitative) criteria, which are usually more appropriate in environmental contexts. MCA has been successfully used in a wide range of environmental planning and management contexts, including allocation of water resources, coastal development, or the management of coastal resources, [330–332], such as fisheries [333]. Moreover, MCA has become part of the standard decision aid frameworks used in EIAs [295, 334, 335].

Parts of this chapter were based on:

S. Lattemann, M.S. Anarna, J.C. Schippers, M.D. Kennedy and G. Amy. Multi-criteria decision support system for seawater reverse osmosis plants. European Desalination Society (EDS) Conference and Exhibition on Desalination for the Environment, Baden Baden, Germany, 2009.

This chapter has the objective to develop a decision support system for desalination plants using multi-criteria analysis, which can be used in the planning and permitting process of new projects. MCA is a powerful tool to compare *site* and *process* design alternatives for the desalination plant, or *route* alternatives for the water supply pipeline. For example, it is known that MCA has been used to facilitate site selection for one large desalination plant in Australia and two small plants in South Africa [64, 293, 294]. General criteria for site selection have been summarized in section 6.5, and have also been proposed in the recent literature [17, 336, 337]. However, as environmental data is highly site-specific and will largely depend on the project sites in question, it is hardly possible to conduct an MCA without having real *site* data at hand. The present study will therefore focus on *process* selection instead of site selection.

Although the process design may still vary between different desalination projects, the data is generally more uniform, and it is therefore possible to compare the principal design and operation options. As outlined in chapter 1, SWRO is the preferred process for most new desalination projects in the Mediterranean region, in Australia, in South-East Asia and in the Americas, and it is also gaining market shares in the Middle East where distillation processes have been traditionally preferred. The most important consideration in a SWRO system is the intake and pretreatment, as a good and reliable water quality must be obtained with a low fouling potential to the SWRO membranes. The design of the outfall is another important considerations in terms of environmental impacts. However, concentrate disposal is more straightforward, i.e., it basically requires an effective diffuser in a suitable oceanographic site, and the alternatives are therefore rather limited. It was therefore decided to apply the MCA to the main intake and pretreatment options for SWRO plants. The information needed for this approach was obtained from literature sources and, to a limited extent, directly from plant operators. An introduction to MCA is given in section 9.2, the methodological approach and input data are described in section 9.3, and results and conclusions are presented in sections 9.4 and 9.5.

9.2 Multi-criteria analysis

9.2.1 Pros and cons

Proponents of the method claim that MCA provides a systematic and transparent approach that increases objectivity and generates results that can be reproduced, whereas opponents claim that the method is prone to manipulation, is very technocratic, and provides a false sense of accuracy [295]. For example, the Dutch Ministry of Transportation and Waterways has been promoting the use of MCA for a long time, and the Dutch Commission for EIA published a manual on MCA [295], as did the UK government [296]. In contrast, the German Institute of Hydrology, which advises the Federal Ministry of Transportation, concluded in a critical review of assessment methodologies that the disadvantages of MCA outweigh its advantages, primarily because the formalized step of data aggregation is *intransparent*, gives a *false sense of accuracy*, and is above all highly *questionable*. However, the critical review also acknowledged that no single assessment procedure exists that is objective and universally accepted [338]. The main points of criticism are laid out in the three following paragraphs, from which a catalogue of basic requirements for an MCA can be derived.

Point 1: Aggregation in an MCA is a four step process that transforms each alternative into a single dimensionless value. Each step requires a decision to be made which is to some degree subjective and can influence the results. First, a single representative value is selected from a pool of often variable data. These values are then standardized, choosing a standardization function, and multiplied by a subjective weight factor. Even the choice of the MCA model may influence the result. It can be difficult to understand and retrace these single steps, especially for a layman or due to poor documentation, which explains why the method is often perceived as *intransparent*.

Point 2: It may also be difficult to understand and interpret the results of the MCA themselves. Aggregation of the criteria is an essential element of MCA and a prerequisite for decision making, but it reduces each alternative to a single abstract value, and relevant information is thereby eclipsed. For most people, it is easier and more intuitive to deal with real figures and units, such as energy use in kWh, than with an abstract value. What does it actually mean if one alternative is by a value of 0.05 better or worse than another alternative? The values are often calculated to one or two decimal places, which gives a *false sense of accuracy* if one considers the variability in the underlying input data and the multiple choices to be made with regard to standardization or modeling.

Point 3: Another main point of criticism is that poor performance with regard to one criterion can be compensated by good performance in another, for example habitat loss could be compensated by a low energy use. Aggregation is therefore *questionable*, and probably not permissible for all criteria, from a regulatory point of view. Furthermore, the method may be prone to manipulation that way, as poor performance in one or more aspects could be concealed. Another way of manipulating the results is by omitting relevant alternatives and criteria during problem definition. Adding an irrelevant alternative with extreme scores may also influence the ranking of alternatives, as extreme scores may influence the standardization of scores, and therefore all other scores [339].

To play its role in the process, the MCA must therefore be well-documented and it must include all relevant alternatives and criteria. By making each step and each choice explicit, the MCA becomes transparent and its results can be scrutinized. An assessment always depends on the subjective decisions and value judgments of the persons involved, but if decisions are made explicit they are also open for discussion. Another benefit of MCA is that it highlights factual differences between alternatives (e.g., alternative A requires more energy than B) as well as subjective preferences of stakeholders (e.g., alternative A is preferred by the project proponent, B by environmental groups). Highlighting differences raises awareness on trade-offs which are inherent to decision making [332].

A screening step may be included to eliminate non-feasible alternatives which do not comply with certain ‘non-compensatory’ or ‘non-negotiable’ criteria [332, 334]. For example, screening criteria could be the overall cost of an alternative if a certain budget may not be exceeded, its technical feasibility (e.g., a subsurface intake will not be feasible in all locations), or legally binding environmental standards or thresholds, such as protected species or habitats which may not be affected, or discharge levels for pollutants.

Finally, it has to be kept in mind that MCA is a tool that can facilitate but not replace decision making. It was noted that the attitude towards MCA often changes in the process [295]: in the beginning, it is often perceived as a ‘black box’ which is easily manipulated, whereas, ironically, the confidence in the results is often too high in the end. To that end, the MCA method should not only be well documented and transparent, but the limitations of MCA and decision support systems in general should be clear to all participants.

To conclude, the purpose of MCA is not always to single out the correct or best decision but to dynamically evaluate a set of alternatives in order to gain information about the effects of different courses of action [332]. No MCA technique can eliminate the need to rely heavily on sound knowledge, data, and judgments, or the need for a critical appraisal of the results [334]. The final selection of an alternative should therefore be supported by a weight of evidence discussions and qualitative considerations.

9.2.2 The role of MCA in EIAs

EIAs are complex multi-stage, multi-disciplinary and multi-participatory processes, in which different site and process alternatives and a wide range of environmental impacts are being considered. When confronted with complex situations, most individuals will attempt to use intuitive judgment to simplify the problem, which may result in biased decision making without taking the full complexity of a problem into account [334]. The increasing volume of complex and controversial information generated by EIAs and the limited capacities of individuals to integrate and process this information emphasizes the need for a formalized method that aggregates the information in a transparent and consistent way. MCA is such a method, which can facilitate the EIA in different ways and in different stages (Table 41).

In the simplest case, the EIA process includes three design rounds: 1. preliminary design $\hookrightarrow$ analysis/evaluation $\hookrightarrow$ bargaining/choice $\hookrightarrow$ 2. revised design $\hookrightarrow$ analysis/evaluation, $\hookrightarrow$ bargaining/choice $\hookrightarrow$ 3. final design [295]. The first round corresponds to the step of scoping in which all possible alternatives are considered and a few are selected for further design in the second round. In the second round, a more detailed evaluation of the environmental impacts of these alternatives is then provided (section 7.2). The iterative design rounds usually move from broad brush to more detailed information, and alternatives are often eliminated, refined or added in the process. The process is also known as ‘adaptive management’ [332]. At the end of a design round, a feedback loop is added that allows the re-design and re-ranking of alternatives in the light of new information. The information established in each round is typically summarized in an evaluation table, and aggregation may be supported by MCA. The EIA process ends after the final design round, when decision making begins. It requires a final evaluation of the alternatives, based on the EIA results and stakeholder interests, and often bargaining and trade-offs in order to select one single alternative [295].

Considering site and process design alternatives in separate design rounds may result in sub-optimal solutions. For example, the best location is often selected in a first round, followed by the best technological design in a second round. This approach may be tempting in order to split the decision problem into smaller, more manageable units, and to limit the number of alternatives and criteria for each design round. However, this approach is only permissible if both aspects are truly independent, which may not always be the case [295]. For example, a subsurface intake, as identified as BAT in chapter 6, is only feasible in locations with a permeable substratum, and the design of a diffuser system has to take local currents and distribution of habitats into account.

9.2.3 MCA procedure

An MCA typically involves two phases (Figure 30). In the first phase, the decision problem is defined, input data is generated, and the alternatives can be ranked based on the input data by means of a graphical evaluation. In the second phase, the alternatives are ranked using MCA, which involves the selection of an MCA model and standardization functions, applying weights to the criteria which reflect value judgments, and a sensitivity analysis of the ranking.

Table 41: Requirements of EIA and capabilities of MCA studies.

EIAs are...	MCA can...
...multi-stage processes: ▸ in a first selection round (typically scoping), all possible alternatives and their potential environmental concerns are identified, of which a limited number are selected for the second round, ▸ in the second design round, more detailed investigations are carried out for the selected alternatives, ▸ in the last step of decision-making, the preferred alternative is selected.	...be used in different stages (design rounds) of the EIA to eliminate, refine or add alternatives in the light of new information, which is summarized in an evaluation table and aggregated by MCA in each stage/round.
...multi-disciplinary processes: ▸ which cover different natural and environmental science disciplines as well as human health and socio-economic aspects where appropriate and, ▸ which involve various specialist studies.	...be used to integrate a large number of criteria, including non-commensurable, qualitative criteria, into decision making processes.
...multi-participatory processes: ▸ involving different decision makers, such as different government agencies, politicians, and stakeholders, ▸ involving the wider public, particularly for projects that arouse public interest such as water supply.	...raise awareness of different value judgments of decision makers and stakeholders, and highlight trade-offs between alternatives.
...based on predictions: ▸ EIAs can only be as accurate as the information that is available at the time of project planning, ▸ information gaps need to be clearly identified in EIAs ▸ a precautionary approach should be applied in decision making.	...include a sensitivity analysis to evaluate whether the ranking changes if variations occur in the input data, in case of uncertain data or unforeseen events.

A prerequisite to MCA is problem definition. It refers to the establishment of an overarching, primary objective that clearly states what the decision seeks to achieve, and the identification of a complete set of *alternatives*. The objective can be further subdivided into sub-objectives, which are then translated into operational *criteria* for decision making. Depending on the number of criteria, these can be grouped into clusters or categories (e.g., economic, ecologic). Structuring the defined objectives and criteria is an important prerequisite of MCA and results in the establishment of an *objective tree*.

The input data to an MCA includes information on all criteria and alternatives, i.e., *scores*, as well as information on personal preferences within defined stakeholder groups, i.e., *weights*. In a first step, the alternatives are scored against the criteria, which produces

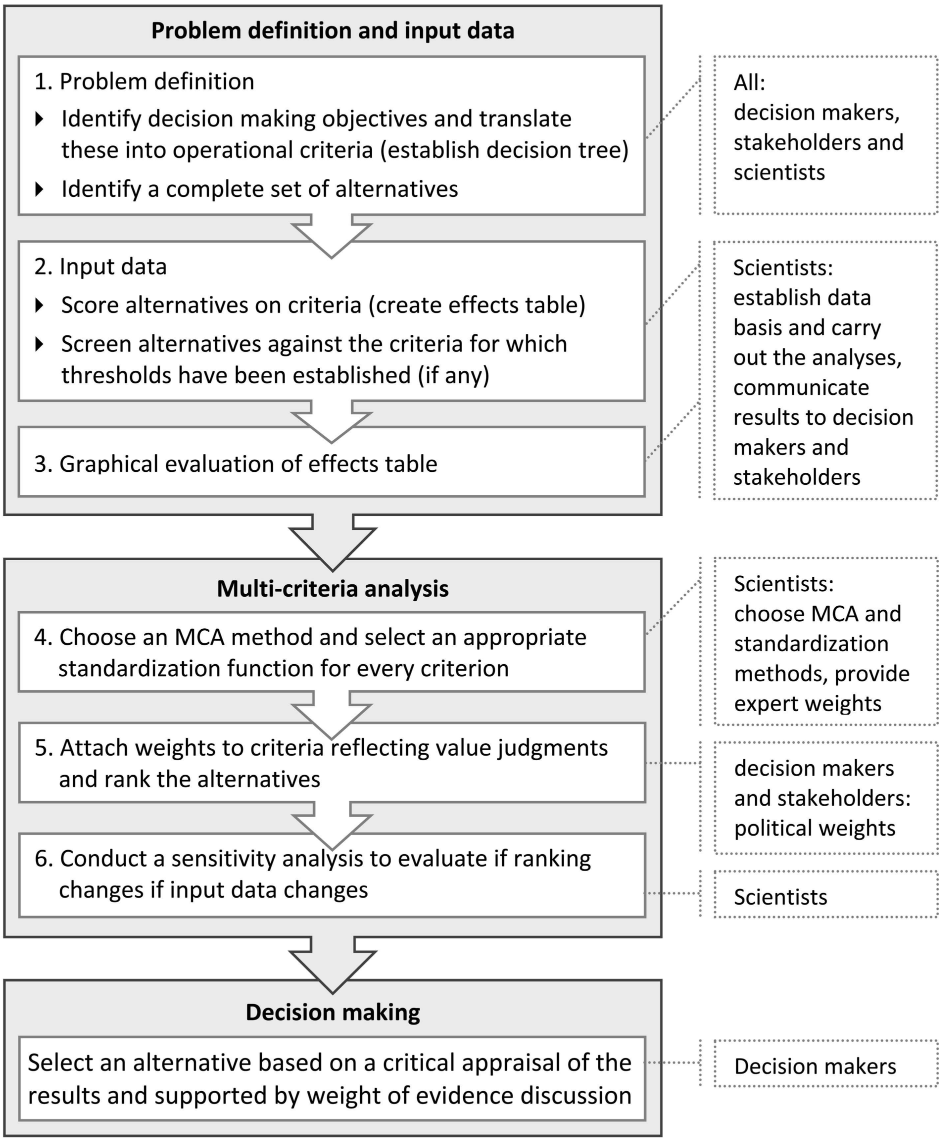

Figure 30: Roadmap to decision making using MCA.

an *effects table*. It is an intermediate product, which can be graphically evaluated without attaching weights to the criteria. Subsequently, MCA is carried out. This involves the selection of a suitable *MCA model* and *standardization functions* for the different criteria to transform the incompatible units of measurement of scores (e.g., chemical use in kg or energy use in kWh) into a dimensionless scale (usually from 0 to 1). Furthermore, *weights* that represent value judgments of decision makers and stakeholders, need to be generated and allocated to the different criteria in order to rank the alternatives. Finally, ranking involves a *sensitivity analysis* to investigate whether the ranking is robust, i.e., whether or not the ranking changes if a variation in the input values occur.

Criteria
The final set of criteria should meet the following requirements: completeness, non-redundancy, operationality and minimality [330, after 340]. This means the selected criteria should completely cover all aspects of the objectives and 'tell' something meaningful about the objectives. They should be independent, without duplication, and measurable. Data should be available for the criteria, and the number of criteria should be kept as low as is consistent with making a well-founded decision [341, 342].

Although up to a hundred criteria have been reported for complex infrastructure projects, a more manageable range would be from six to twenty, which is sufficient for well-founded decision making in many cases [295]. The crux of the matter is to identify those criteria which are truly relevant to decision making and to avoid redundancy.

Alternatives
An EIA may include project alternatives (e.g., building a dam instead of a desalination plant), site or route alternatives within a proposed project (e.g., different sites for the desalination plant or routes for water transfer pipeline), and technical alternatives (such as different pretreatment alternatives, as considered in this study). The decision on project alternatives is usually taken at the more strategic planning level, and should involve a strategic environmental assessment (strategic EIA or SEA), whereas site/route and technical alternatives are often evaluated within the project-specific EIAs.

The set of alternatives to be compared in an EIA should be complete [295], i.e., it should include the alternatives favored by the project proponent and regulatory bodies, as well as a 'zero' ('do nothing', 'no project') alternative and possibly a 'zero plus' alternative with small adjustments to the current situation (e.g., further increase of groundwater abstraction rates in combination with water restrictions). The number of alternatives, for instance, in locating a desalination plant, is theoretically infinite. In practice, however, decision making requires a finite set of alternatives, which should be allowed to shrink or grow during the planning process. The initial set of alternatives is usually reduced in the first round of scoping, which leads to a second and third design round [295, 330].

In EIAs, alternatives often have to be evaluated against non-commensurable criteria (e.g., landscape impacts). This makes MCA an appropriate choice for EIAs, however, MCA assumes that criteria are independent. It was pointed out that this assumption may be incorrect in real-life EIAs [343]. For example, common evaluation criteria such as visual landscape impacts and land use impacts (in terms of area size) are interdependent, and hence not independent. Separating alternatives may similarly result in sub-optimal solutions. For example, the best location is often selected in a first round, followed by the best design in a second round. However, this is permissible only if both aspects are independent which is often not the case, as illustrated by the above example, i.e., the visual damage to a location cannot be estimated if the size of a facility is not known [295]. In this study, the selection of a certain type of intake (open intake or beachwell) will also depend on the site-specific geological conditions in the project sites.

Scores and weights
Scores measure the performance of every alternative against all criteria. Scores can be assessed by experimental methods (e.g., field or laboratory measurements, simulation models), by expert judgment (e.g., landscape impacts) or can be taken from the literature (e.g., energy demand of a certain process). Scores can be measured on quantitative scales (e.g., ratio scales such as costs in €, or interval scales such as temperature in °C), qualitative scales (ranking on an ordinal scale from highest impact to lowest impact) or binary

scales (yes/no). The scores are then transformed into a dimensionless scale (usually from 0 to 1) using a suitable standardization function [341, 342].

The next step is to allocate weights to the criteria, which can either be established directly, or indirectly following a formal procedure. Having the right combination of people is the first essential element in eliciting weights. The three basic groups of people are decision makers, stakeholders/community representatives, and scientists/engineers [Figure 30, 332]. The opinions of the different groups can be formally integrated into the decision process through surveys, workshops or other techniques suitable for eliciting value judgments [332, 335]. How weights are chosen (the method used) can be as important as who chooses the weights, because different methods translate criteria importance into different operational meanings [344].

In the *pairwise comparison* method, all individual criteria are paired against all others, usually by answering questions such as, "With respect to the selection of an alternative, which is more important: water use or energy use, and by which degree?" resulting in answers such as "energy use is much more, equally, less, etc., important than water use". The technique assumes that humans are more capable of making relative judgments than absolute judgments. On the basis of all pairwise comparisons, the quantitative weights are calculated, often facilitated by a software that moves through the pairwise comparisons and then conducts the calculation. In the *expected value method*, quantitative weights are derived by directly ranking all criteria in a consistent order of importance. Criteria can also be given equal importance [341, 342].

In the *swing weight method*, the decision maker assesses which 'swing' from the worst score to the best score of a criterion gives the highest increase in overall value. If the value tree is small, all bottom-level criteria are assessed simultaneously. The criterion with the swing that gives the greatest increase in overall value is given the highest weight. The process is repeated on the remaining set of criteria. To assign values to the weights, the decision maker must also assess the relative value of the swings. For example, if a swing from worst to best on the most highly weighted criterion is assigned a value of 100, what is the relative value of a swing from worst to best on the second ranked criterion, and so on [345]. The weight on a criterion thus reflects both the range of difference of the alternatives, and how much that difference matters [296].

Depending on the number of criteria, these may have been grouped into categories (e.g., economic, ecologic, etc.). In most cases, weights within a category are given by experts and weights between categories by decision makers. Expert weights reflect the relative importance of an effect in scientific terms [295]. For example, an expert weight would be to say that a potential loss of seagrass beds due to salinity increase caused by the outfall of a desalination plant is more severe than the placement impact of the outfall structures on a sandy seafloor inhabited by motile macrofauna species. Expert weights reflect the opinion of one or more experts, and do not create much controversy in the best case, although they also have to balance (sacrifice) one effect against another [295].

In contrast, political weights often create much controversy, as they reflect the trade-offs between categories [295], such as economic versus ecologic aspects. For example, an offshore outfall with a diffuser has a potentially lower environmental impact than a nearshore single purpose outfall but is more expensive to construct. Political weights can be specified for technical and economic perspectives (which are most likely the point of view of the project proponent) and social, health or environmental perspectives (which most likely represent regulatory agencies and different stakeholder views). Separate ranking results can then be produced that reflect the different perspectives.

MCA models
Different MCA models with different strengths and weaknesses have been developed, which synthesize the input data and rank the alternatives by different means [332]. A main difference lies in the compensation between criteria, which can lead to different results with the same data. The two main categories of MCA models are [332, 334]:

- Value or utility function based methods, such as multi-attribute utility theory (MAUT) or the analytic hierarchy process (AHP), are compensatory models. A linear value (utility) model provides full compensation between the criteria, i.e., a poor score on any criterion can be compensated by a sufficiently good value on another criterion. Compensation can be decreased by using nonlinear utility models, however, this leads to the difficult problem of determining the correct shape of the utility function.
- Outranking methods, such as Electre or Promethee, are partially compensatory. They typically do not provide full compensation. They also allow inferior performance on some criteria to be compensated by superior performance on others, however, they do not necessarily take the *magnitude* of relative under-performance in a criterion versus the magnitude of over-performance in another into account.

Some of the criteria in an EIA may not be compensatory or 'negotiable'. For example, good water quality is typically defined by several criteria (e.g., salinity, dissolved oxygen, pollutants, etc.). A poor performance in one of these parameters cannot be compensated by a good performance in another, i.e., low oxygen levels compromise water quality even if pollutant levels are low. This requires either the use of partially compensatory methods such as outranking, or the MCA approach has to be combined with thresholds for criteria which are not negotiable, such as regulatory standards for water quality, or targets which need to be achieved such as conservation of a species or habitat.

There are some requirements for MCA models to be used in environmental problems: The method should be well defined and easy to understand, it should be able to manage the necessary number of alternatives and criteria, and support different decision makers, and the method should be able to handle inaccurate or uncertain criteria information, as uncertainty is inherent to many environmental decision contexts. There is usually no means to objectively identify the best MCA model. Therefore, the chosen model should be justified in real applications, although this is rarely done [330].

The choice of model is hardly an issue for the average MCA users, despite the intensive debate in the scientific community on the different methods. The main methodological challenge is not the development of more sophisticated MCA methods, but to support problem definition. Weighted summation, one of the simplest MCA methods, performs well in most cases. It was found to be the most popular MCA model in Dutch EIAs, and because the model is methodologically sound, easy to explain and transparent, it is also recommended by the Dutch Commission for EIA [295].

Sensitivity analysis
EIAs can only give a prognosis of the expected impacts, based on the information that is available for a specific project and its location at a certain time (Table 41). The costs of a project are often estimated incorrectly, and the same must be anticipated for more complex environmental criteria [330]. A key problem, which ecologists face, is the interconnectedness of ecological cause-effect relationships. This makes it difficult to give a prognosis of what is going to happen, let alone understand what is going on [346]. A decision must be made despite the fact that some of the assumptions may be incorrect, or some of the data may be variable. The accuracy of predictions in an EIA therefore needs

to be scrutinized, information gaps and deficiencies clearly identified, and a precautionary approach applied in decision making [17, 347].

In MCAs, the ranking results, which are the basis for decision making, can be systematically analyzed by a sensitivity analysis. Its objective is to analyze how the ranking changes if the input data change, in case that the input data are incorrect or variable by nature [348]. For example, the energy demand of a desalination plant will vary depending on the intake seawater temperature and salinity, which may vary between intake location sites (e.g., estuarine, surface or offshore submerged intakes) and seasons. As energy demand will likely have a significant influence on the overall ranking of the alternatives, it is necessary to investigate how statistical variability in the input data will influence the ranking. Sensitivity analysis furthermore investigates which changes in the scores or weights are necessary to bring about a change, particularly if two alternatives have only a small difference in their overall utility value. A ranking is considered as robust when it is not sensitive to minor variations in the scores or weights [348].

9.3 Methodology and data input

The scope of the study is to compare different intake and pretreatment systems for SWRO using MCA (cf. section 9.1), which are usually optimized to control fouling by bacteria, organic compounds, suspended and colloidal matter, and scaling (section 2.3.1). Depending on local circumstances, solutions range from minimalist to operationally intensive, the latter consuming considerable amounts of natural resources including chemicals, materials, energy, land, and water. The overall environmental footprint of a desalination process can therefore by minimized by selecting the best pretreatment option in a given site. The methodology follows the MCA approach described in Figure 30, using the software tool DEFINITE 3.1 [342], which facilitates ranking and comparing a finite set of alternatives. It requires that the decision problem has been structured, i.e., cause-effect relationships are known, evaluation criteria are specified, and alternatives under evaluation are well described [341].

9.3.1 Problem definition

In the first step, the problem was defined by identifying the main objective and sub-objectives of the MCA, the alternatives (Figure 31) and evaluation criteria (Figure 32).

Criteria

The primary objective of the MCA is to optimize the pretreatment of SWRO plants, which is subdivided into eight sub-objectives and 15 criteria (Figure 32). Objectives 1 to 6 reflect different aspects of resource consumption and environmental impacts, and objectives 7 and 8 refer to economic and operational aspects. For MCA, one should distinguish between 'cost' criteria, which have a negative correlation between score and effect (the higher the score, the worse the effect), and 'benefit' criteria with a positive correlation. The criteria of water quality and material recyclability were defined as benefits, while the remaining criteria were all costs. Furthermore, the measurement scales of the criteria have to be specified. Chemicals, land, energy and water use as well as costs are ratio scales, i.e., the importance of a criterion is proportional to its value, e.g., a two times higher energy demand is two times as bad (or two times as good for benefits). Ecological impact, material recyclability and SDI are binary scales, which only indicate whether an effect is fulfilled (yes) or not fulfilled (no). A 'yes' means a benefit in the case of material recyclability and SDI, and a cost in case of ecological impact.

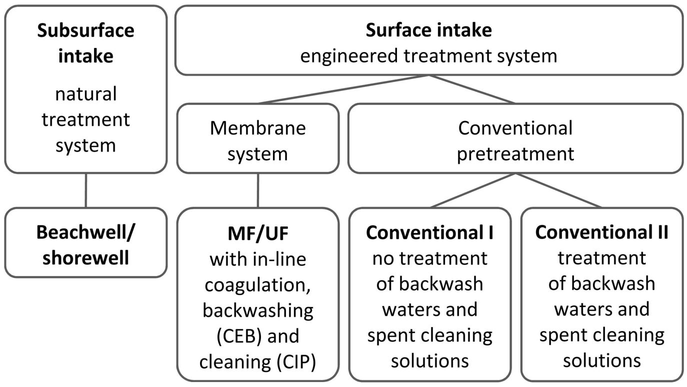

Figure 31: Pretreatment system alternatives considered in the MCA.

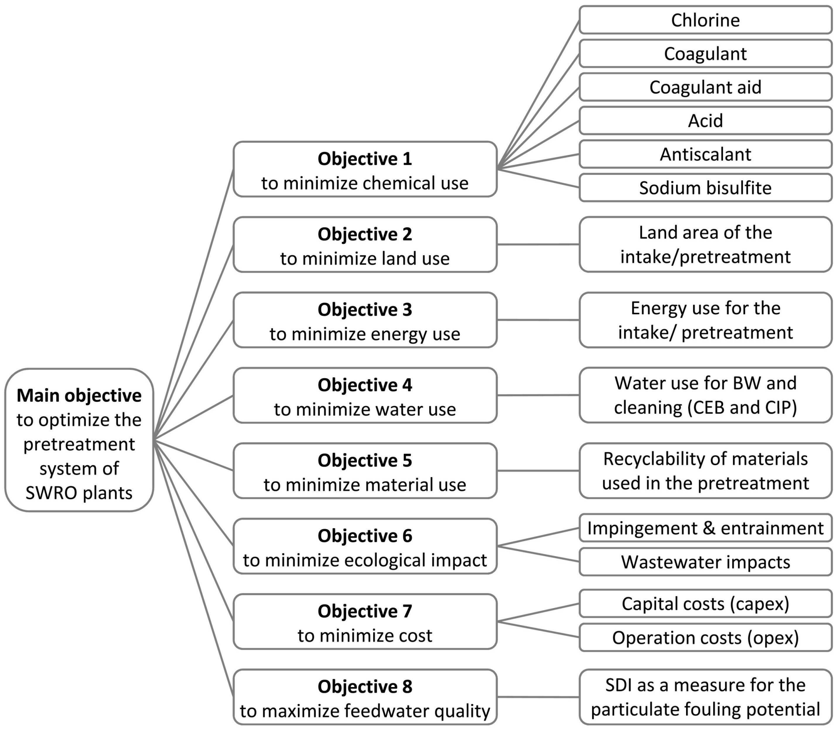

Figure 32: Objective-tree (value tree) hierarchy of the MCA with main objective (left), sub-objectives (middle) and evaluation criteria (right). BW: backwashing, CEB: chemical enhanced backwashing, CIP: cleaning in place. Material use: Recyclable means the materials can be reused or reprocessed into new products in order to prevent waste. Ecology: refers to the ecological impact due to seawater intakes, causing impingement and entrainment, and potential effects from the discharge of untreated backwash waters and cleaning solutions. SDI: surrogate parameter that estimates the potential of SWRO fouling caused by fine suspended organic or inorganic colloids.

Alternatives
The considered alternatives as shown in Figure 31 included natural subsurface systems, i.e., beachwells (see page 24), and engineered pretreatment systems, distinguishing between conventional pretreatment (page 25ff.) and membrane pretreatment (page 39ff.).

It was assumed that the intake and pretreatment systems pretreat between 40,400 and 42,000 m^3 of seawater per day, depending on the water losses in the pretreatment system for backwashing (from 1% to 5%), and provide 40,000 m^3 of feedwater for a single pass SWRO plant operated at 50% recovery and with a capacity of 20,000 m^3/d (Figure 33). A nominal plant capacity of 20,000 m^3/d was selected because beachwells have a limited intake capacity and are therefore mostly used for smaller SWRO plants. Similarly, UF/MF pretreatment has so far mostly been applied to smaller SWRO plants, with the largest operational UF-SWRO plant having a capacity of 50,000 m^3/d (Table 6, page 33). Most large SWRO plants still use a conventional pretreatment with one granular media filtration stage. Some plants have two media filtration stages, and a few plants have a third pretreatment stage in addition to two stage media filtration (section 2.3.1). The conventional pretreatment design considered in this study also consists of one stage, similar to most SWRO plants. As no large SWRO plant uses a beachwell intake or UF pretreatment to date, caution must be used when extrapolating the results of this study to large SWRO projects with capacities of 100,000 m^3/d or more.

Neodren sub-seabed drains, which are a special configuration of natural subsurface systems, could provide sufficient flow rates for large facilities, depending on the number of drains installed (page 24). Although the technology is used in a few SWRO plants and has been described as technologically sound and very environmentally friendly [349], it was decided to exclude it from the list of alternatives because of limited data availability.

We assumed that the beachwell intake is followed by an additional sand filter. Conventional pretreatment is assumed to consist of coagulation-flocculation followed by a single stage pressurized DMF. Because cylindrical pressure vessels are limited in diameter to about 3 m, they are commonly used in smaller SWRO plants [22]. However, several *large* new plants in Algeria, Southern Europe and Australia (Table 5, page 30) also reported single or dual stage pressure filters. Single stage gravity filters still seem to be more common in large SWRO plants though, mainly because they enable larger filtration areas, require less energy, and are therefore cheaper [22].

Two alternatives for conventional pretreatment were considered in the MCA: without treatment (conventional I) and with treatment of backwash waters and spent cleaning solutions (conventional II). Treatment will likely result in a higher land use, energy use and cost, but lower marine environmental impact. Data on the costs and land use of a sludge and waste water treatment facility could not be obtained from the literature. To be accurate, one would also have to include the land use impacts for the sludge landfill, which depends on the sludge quantities and the disposal site, and the cost of operation and transportation [230]. Small sludge amounts may be dewatered in a simple and relatively inexpensive drying bed on-site. A worst case scenario would require a clarifier/thickener followed by a sludge dewatering system, using a belt press or centrifuge, in a separate building. The energy use of sludge treatment can be estimated to range between 6% and 54% of the pretreatment energy costs [53, Table 9]. Due to the complexity and highly site-specific nature of land use, energy use, and costs, a general increase of 10% was assumed for the alternative conventional II in all three criteria.

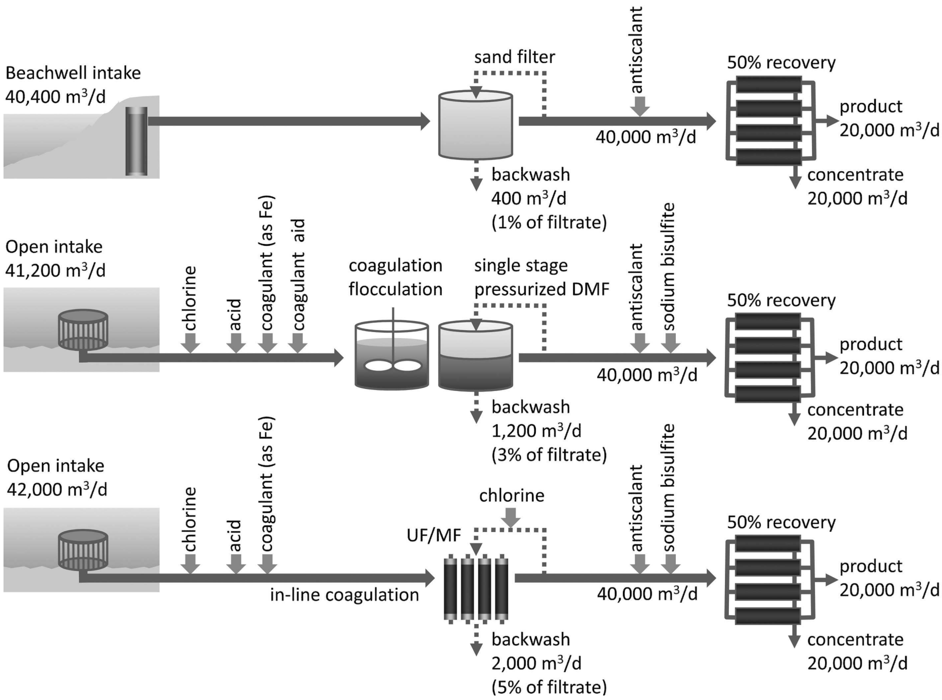

Figure 33: Process flows for a 20,000 m^3/d facility. In beachwells (top), the seawater is naturally filtered through the sand bed and then pumped into the SWRO plant for further pretreatment, which usually consists of a sand filter and acid and/or antiscalant addition. Either conventional (middle) or membrane pretreatment (bottom) is used where the feedwater is received from an open intake. Both usually involve chlorination followed by coagulation, filtration, acid and/or antiscalant addition, and dechlorination. For the purpose of this study, it is assumed that conventional pretreatment uses a single stage pressurized dual media filter. In UF/MF pretreatment, coagulants are typically used in-line and lower dose rates are possible, no coagulant aid is used, and no flocculation step is required, but UF/MF typically requires chemically enhanced backwashing with chlorine.

For UF/MF pretreatment, it was assumed that the backwash and CEB wastes are discharged into the sea, as no UF-SWRO plant in the literature reported treatment. This standard practice has also been confirmed by a UF membrane supplier. Moreover, the use of a disinfectant, either continuously or intermittently, and the use of in-line coagulation, is assumed to be common practice in pilot and full-scale UF plants to improve the process performance and filtrate quality of the pretreatment (cf. section 2.3.2).

The study was limited to single pass RO, as the need of a second RO pass depends on the product water specifications. As pretreatment solutions are usually site-specific, the number of possible alternatives is theoretically infinite. To be practical, only the standard designs of the major pretreatment alternatives were considered, and it was assumed that the set of alternatives is thus complete, which is one prerequisite for MCA.

Table 42: MCA reference values, as given below, were selected from Tables 3-7, page 29ff. in chapter 2, and used to calculate the scores for a 20,000 m^3/d plant, assuming the flow rates and dosing points shown in Figure 33.

Criterion	⊖/⊕ scale[a]	**Beachwell** **selected reference value**	**calc. score for 20,000 m^3/d**	**Conventional pretreatment I[b]** **selected reference value**	**calc. score for 20,000 m^3/d**	**UF pretreatment** **selected reference value**	**calc. score for 20,000 m^3/d**
Chlorine (feed water)	⊖ ratio	**–** water from beachwells is not chlorinated (Table 3)	–	**1 mg/l** if continuous chlorine use: 0.5-1 mg/l residual usually maintained throughout pretreatment line (Table 5)	**41 kg/d**	**1 mg/l** if continuous chlorine use: 0.5-1 mg/l residual usually maintained throughout pretreatment line (Tables 6,7)	**42 kg/d**
Chlorine (back-wash)	⊖ ratio	**–** does not apply	–	**–** does not apply	–	**100 mg/l 1x/d** varied between 5-20 mg/l 1x/15-60 min. up to 500 mg/l 1x/d (Tables 6,7)	**4 kg/d**
Sodium bisulfite	⊖ ratio	**–** no dechlorination required	–	**3 mg/l** dosing rate usually three times the residual chlorine concentration [15, 22]	**120 kg/d**	**3 mg/l** same assumption as for Conv. I	**120 kg/d**
Coagulant	⊖ ratio	**–** not used after beachwells (Table 3)	–	**1.2 mg/l Fe** values were 0.5-3.0 mg/l as Fe and <1-20 mg/l as $FeCl_3$ or $FeSO_4$ (≡ 0.1-8.0 mg/l as Fe, 40% active ingr.) 1-6 mg/l $FeCl_3$ (≡ 0.4-2.4 mg/l as Fe) assumed to be the rule (Table 5)	**49 kg/d**	**0.4 mg/l** 0.3-10.0 mg/l as Fe in full scale plants, 0.2-4.0 mg/l as Fe in pilot plants, it is assumed that UF can be operated with about one third of the coagulant dose as in Conv. I (Tables 6,7)	**17 kg/d**
Coagulant aid	⊖ ratio	**–** not used after beachwells (Table 3)	–	**0.85 mg/l** values were 0.2-1.5 mg/l (Table 5)	**35 kg/d**	**–** typically not used	–
Acid (H_2SO_4)[c]	⊖ ratio	**–** no pH adjustment required (no coagulation step)	–	**30 mg/l** values were: 22-55 mg/l (Table 5), 20-25 mg/l [membrane textbook, 22] 30-100 mg/l [literature review, 15]	**1236 kg/d**	**10 mg/l** no doses available, 2 full scale plants used acid in addition to Fe, which is assumed to be for coagulation and not for scale control, as both plants also used antiscalant (Table 6), selected value is 1/3 of H_2SO_4 dose in Conv. I	**420 kg/d**
Anti-scalant[d]	⊖ ratio	**2 mg/l** values were 0.9-1.2 mg/l for polymer antiscalants and 8 mg/l for phosphate antiscalant (Table 3)	**80 kg/d**	**2 mg/l** values were 1-2 mg/l or 1-4 mg/l if one counts an earlier literature review [15] and company information (Table 5)	**80 kg/d**	**2 mg/l** probably similar to Conv. I; 1 pilot plant reported 3 mg/l [119], a literature review reported 1-3 mg/l [126] (Table 7)	**80 kg/d**

continued on next page

Criterion	⊖/⊕ scale[a]	Beachwell selected reference value	calc. score for 20,000 m^3/d	Conventional pretreatment I[b] selected reference value	calc. score for 20,000 m^3/d	UF pretreatment selected reference value	calc. score for 20,000 m^3/d
Land use for pre-treatment	⊖ ratio	**0.8 m^2 per 1000 m^3/d** inst. capacity 4.7 m^2 was given for a 6,000 m^3/d plant with a vertical beachwell [58]	**16 m^2**	**38 m^2/1000 m^3/d** value selected from a range of 35-40 m^3/d [22]	**Conv. I: 760 m^2** **Conv. II: 836 m^2**	**27 m^2/1000 m^3/d** ≡30% reduction from Conv. I, 25% and 33% reduction reported [22, 114]	**540 m^2**
Energy use for pre-treatment	⊖ ratio	**0.1 kWh/m^3** of pretreated water for a vertical beach well incl. sand filter [58]	**4000 kWh/d**	**0.15 kWh/m^3** of pretreated water for intake + 1 stage pressurized DMF, reported values ranged between: 0.035 kWh/m^3 RO permeate (min.) ≈0.018 kWh/m^3 pretreated water for single GF excl. intake [53, Table 9]; 0.35 kWh/m^3 RO permeate (max.) ≈0.16 kWh/m^3 pretreated water for 2 stage DMF incl. intake [60, Table 4]; 0.12 kWh/m^3 pretreated water for single stage pressure filter [131]	**Conv. I: 6000 kWh/d** **Conv. II: 6600 kWh/d**	**0.15 kWh/m^3** of filtrate reported values ranged between: 0.03-0.2 kWh/m^3 filtrate for UF only (incl. BW and CEB but not intake and screening, Table 10)	**6000 kWh/d**
Water use for BW, CEB, CIP	⊖ ratio	**1%** of pretreated water assuming a sand filter backwashed 1x every 3 days	**400 m^3/d**	**3%** of pretreated water assuming a 1 stage DMF backwashed 1x every day; 2-3% of the filtrate capacity per filtration stage used for backwashing [22]	**1200 m^3/d**	**5%** of filtrate assuming a BW/CEB 24-48x/d and CIP 1-2x/a	**2000 m^3/d**
Costs	⊖ ratio	no data, assumed to be 50% of the conventional pretreatment cost Capex: **0.4 US¢/m^3** of pretreated water Opex: **1.1 US¢/m^3** of pretreated water total: **1.5 US¢/m^3** of pretreated water	 **160 US\$/d** **440 US\$/d** **600 US\$/d**	assumed costs for a single stage pressure DMF [131]: Capex: **0.7 US¢/m^3** Opex: **2.2 US¢/m^3** total: **2.9 US¢/m^3**	Conv. I / II in US\$/d: **280 / 308** **880 / 968** **1160 / 1276**	assumed costs for a DOW SFP-2880 operated at 75 lmh [131]: Capex: **1.2 US¢/m^3** of filtrate Opex: **1.9 US¢/m^3** of filtrate total: **3.2 US¢/m^3** of filtrate	 **480 US\$/d** **760 US\$/d** **1280 US\$/d**
Material use	⊕ binary	material is mostly recyclable, and no large material wastes are produced?	**yes**	material is mostly recyclable, and no large material wastes are produced?	**yes**	material is mostly recyclable, and no large material wastes are produced?	**no**
Potential ecological impact	⊖ binary	impingement/entrainment of organisms? discharge of backwash water/sludge?	**no** **yes**	impingement/entrainment of organisms? discharge of backwash water/sludge?	**yes** **yes for Conv. I** **no for Conv. II**	impingement/entrainment of organisms? discharge of backwash water/sludge?	**yes** **yes**
Water quality	⊕ binary	is an SDI<3 achieved?	**yes**	is an SDI<3 achieved?	**yes**	is an SDI<3 achieved?	**yes**

[a] This column indicates whether the criterion is a 'cost' (⊖) or 'benefit' (⊕) criterion, and the scale of the criterion (ratio or binary).
[b] For the alternative conventional II (with treatment of backwash and waste waters), which is not included in this table, a 10% higher energy use, cost and land use was assumed.
[c] Acid can be added to adjust the pH for coagulation and/or to control scaling. Since the dosing of an antiscalant is already assumed ahead of the RO units, acid dosing is assumed to take place before the pretreatment and primarily for the purpose of coagulation. This is also the case in several large SWRO plants, e.g., Barcelona, Fujairah, Perth, Sydney and Tugun/Gold Coast.
[d] Antiscalant dosing takes place after the pretreatment system and concentrations were assumed to be the same for all four alternatives.

9.3.2 Input data

The four alternatives were scored on all 15 criteria, which can be categorized into quantitative and qualitative criteria. In a first step, a data base was established for all quantitative criteria through a literature review of operational and pilot plant data and by personal communications with plant operators (see chapter 2, Tables 3 to 7, page 29ff.). In a second step, a value was selected which is considered to be representative of the standard design of an alternative (Table 42). It should be mentioned that the compiled baseline data were quite variable, and that a single 'true' value, which represents all operating conditions and feed water qualities, does not exist. The selected values should therefore be understood as one scenario which can be refined and revised in the light of new or better data. To deal with data variability in the input data, a sensitivity analysis was carried out (page 205) and two revised scenarios were considered (page 209).

Scores

The MCA scores (Table 42) were calculated using the reference value, assuming a reference plant size with a capacity of 20,000 m^3/d, and taking the respective flows and dosing points given in Figure 33 into account. For example, the dosing concentration of 1 mg/l of chlorine is multiplied by the intake volumes of 41,200 m^3/d and 42,000 m^3/d, which results in a daily load of 41 and 42 kg/d for conventional and UF pretreatment.

The only qualitative criteria are water quality (SDI), ecological impact, and recyclability of materials. For SDI, a value of 3 is selected assuming that all pretreatment alternatives are running in good condition. An SDI range of 2-4 is typically required by membrane manufacturers for conventional pretreatment. UF/MF has a reported SDI range of 0.8-2.2 [50, 99, 107, 112, 119, 126], and beachwells have reported values of 0.4-2.8 [58, 60, Tables 3 to 7, page 29ff.]. Concerning ecological impact, it is assumed that the open intakes of UF and conventional pretreatment systems have the potential to cause the impingement and entrainment of marine organisms ('yes'), which is not the case for beachwell intakes ('no'). The other criterion refers to the potential impact resulting from the discharge of untreated backwash waters from filters and UF membranes and cleaning solutions in all three pretreatment systems ('yes'). The alternative conventional II assumes that the backwash waters are treated and that the sludge is deposited on land ('no' impact). For material use, only the UF system was given a score of 'no' assuming that the UF/MF membranes are currently not recycled, while it is being assumed that the filter media are recycled or have a beneficial reuse ('yes').

9.4 MCA results

9.4.1 Graphical evaluation

The effects table (Table 42), which contains the pretreatment alternatives' scores on the various criteria, as well as the units of measurement or type of scales, and a statement whether a criterion is a 'cost' or 'benefit', is an intermediate working product, which can be graphically evaluated before the start of the MCA (see Figure 30, MCA roadmap). Graphical evaluation has the objective to illustrate the relative performance of the alternatives without assigning weights to the criteria. The scores were transformed into a dimensionless value between 0 and 1 by a linear interval standardization function, which is the default setting of the DEFINITE software (Figures 34, 35).

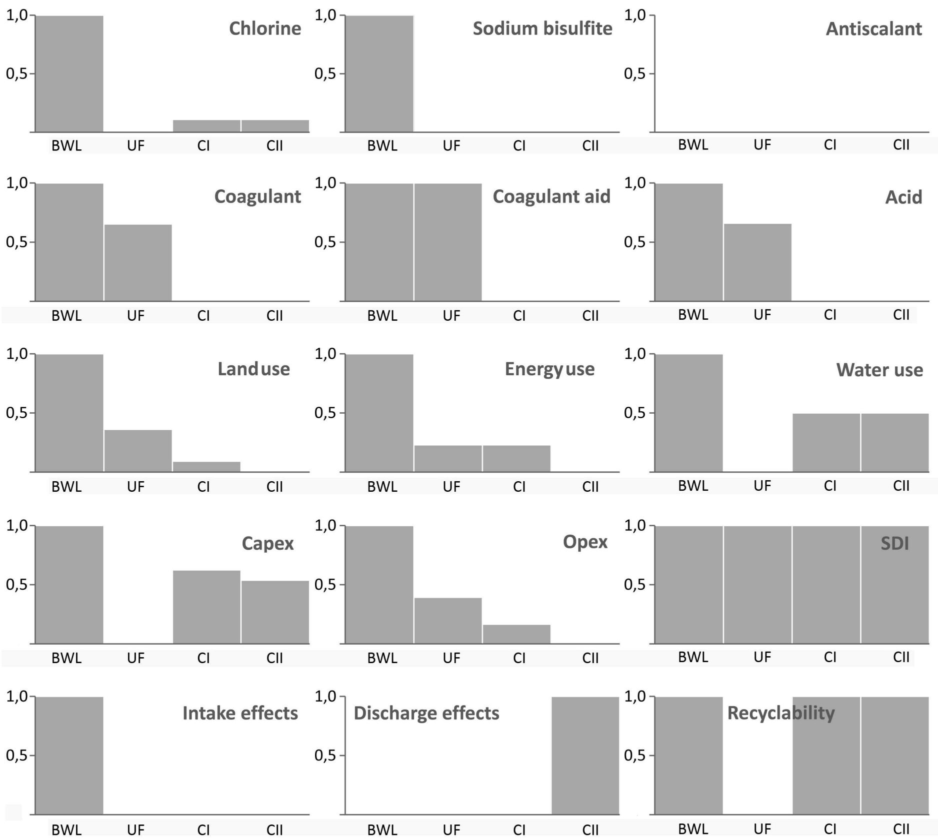

Figure 34: Graphical evaluation of scores using interval standardization, in which the best score receives a value of one (full bar), the worst a value of zero (empty bar), and all remaining alternatives are scaled in between. For ‘benefit’ criteria (recyclability, SDI), a higher bar represents the better alternative. Note that for ‘cost’ criteria (all remaining criteria, such as chemical or energy use), a higher bar indicates a lower negative effect and therefore also a better alternative.

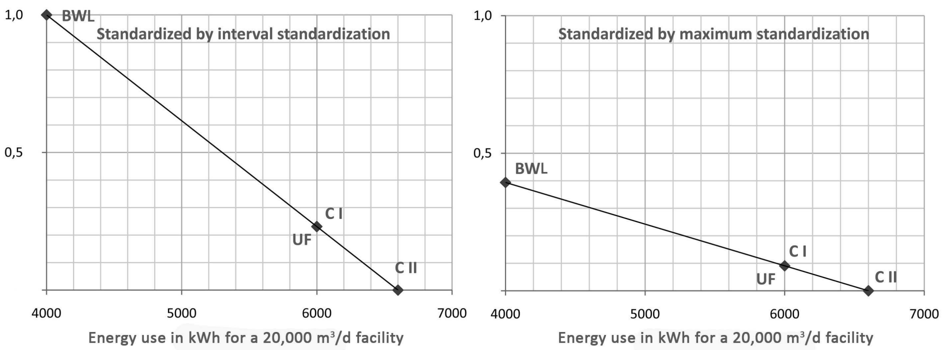

Figure 35: Linear interval (left) and maximum standardization (right) of the criterion energy use.

The graphical presentation in Figure 34 shows that a beachwell:

- is the best alternative in terms of chemical use (chlorine, sodium bisulfite, coagulant and acid) because these chemicals are typically not required,
- has the lowest land use, water use, energy use, and costs, and does not cause ecological impacts from impingement and entrainment (intake effects),
- is equal to the other pretreatment alternatives in terms of antiscalant use (all are assumed to use 2 mg/l), and with regard to SDI (all assumed to achieve an SDI<3),
- is equal to UF pretreatment with regard to the use of coagulant aid (none),
- is equal to UF and conventional pretreatment I in terms of ecological impact from filter backwash waters if these are assumed to be discharged into the sea.

Accordingly, UF pretreatment:

- is the second best alternative in terms of coagulant and acid use, but worst in terms of chlorine and sodium bisulfite use similar to conventional I and II,
- is the second best in terms of land use and operating costs (opex), but worst in water use and investment cost (capex),
- is equal to conventional pretreatment I in terms of energy use,
- is the worst alternative in terms of recyclability of materials and ecological impact, if one assumes that the intake causes impingement and entrainment, and that the backwash waters from CEB and CIP are discharged without treatment.

Conventional pretreatment I and II:

- both are equal to beachwells in terms of recyclability of materials,
- are the second best alternatives in terms of water use and investment cost as well as chlorine use (or second worst in the latter case, depending on the perspective), but the worst with regard to all other chemicals, as well as land use and operating costs,
- are the worst alternative in terms of ecological impact (same as UF), if one assumes that the intake causes impingement and entrainment, and that the backwash waters from filters are discharged without treatment.

It is noteworthy that conventional I is equal to UF in terms of energy use. Conventional II is the best alternative in terms of impacts caused by waste water discharges, because a waste water treatment step has been included in this alternative, but at the expense of additional energy and land requirements of this treatment step.

It can be concluded that beachwells are dominant over all other alternatives if one excludes the criterion of discharge effects. Dominance occurs when one alternative performs at least as well as another alternative on all criteria and strictly better than the other on at least one criterion [350]. In the considered scenario, it is assumed that the untreated discharge of backwash waters from sand filters is the norm for small SWRO plants. However, three out of four beachwell plants in Table 3 (page 29) reported a sludge treatment and landfill [59, 60], and one plant injected its waste water into a deep well [58]. Even if the backwash is discharged without treatment, sludge amounts are small because of a low solids content in the feed water and no coagulant use, which results in a low backwash frequency of once every few days [59]. It is thus safe to assume that beachwells are dominant over the other alternatives. In practice, dominance of one alternative is rare and the extent to which it can support real decisions is correspondingly limited [350].

9.4.2 Standardization

Interval standardization was used in the previous step of graphical evaluation. The scores are standardized with a linear function between the best score, which receives the value of one, and the worst score, which receives the value of zero. For benefit effects, the best score is the highest absolute value. For cost effects, the best score is the lowest absolute value. The other alternatives are linear interpolations between zero and one according to their relative position (Figure 35 left, equations 9.1, 9.2). The standardized values are not proportional to the original scores and differences are accentuated. This type of standardization is often used for relative scales, e.g., an increase in costs [341, 342].

Maximum standardization is another linear function. For benefit effects, the highest absolute value is the best score, which receives the value of one (full bar), whereas the lowest absolute value becomes the lowest standardized score (i.e., closest to zero, or zero if the absolute value is zero). For cost effects, the highest score is the worst score, which receives the value of zero, whereas the lowest score is transformed into the highest standardized score (closest to one or one, Figure 35 right, eq. 9.3, 9.4). The standardized values are proportional to the original scores, e.g., if the original score is doubled, the standardized score is also doubled. This is not the case in interval standardization. However, for both interval and maximum standardization, the ratio of the difference between two scores and the difference between their standardized counterparts is constant (eq. 9.7). Interval and maximum standardization result in the same standardized scores for the criteria chemicals, recyclability, SDI, intake and discharge effects, because the lowest value ($score_{min}$) is zero and equations 9.1 and 9.2 are identical to equations 9.2 and 9.3.

A third linear standardization function is goal standardization, in which an ideal or goal value and a baseline value are specified. The scores are again normalized with a linear function between zero and one, similar to interval standardization (eq. 9.5, 9.6). If beachwell is selected as the benchmark (goal) against which the other alternatives are measured (with the worst alternative as baseline), the results are principally the same as for interval standardization with the difference that the lowest (best) score is now set to one, and the highest to zero for benefit effects. For cost effects, it is the other way round.

Although in practice the relation between a criterion score and its value is usually more complex, a linear standardization is often an acceptable approximation if the range of the scores is not too large [341]. In those cases that a linear approximation is not acceptable, other, non-linear, standardization or value functions can be used. The DEFINITE programme includes S-shaped, concave, convex and free-form shapes [341]. However, the use of tailor-made standardization functions bears the risk that the standardized scores are distorted compared to the original scores if a wrong function is selected. Therefore, experts should be consulted to identify the best possible shape of the curve, especially when tailor-made non-linear functions are used [351].

A linear interval standardization has therefore been selected for all criteria in graphical evaluation and also in the MCA. A linear relationship is adequate as the range of values in this study is not too large. Interval standardization produces the same results as maximum standardization for the criteria chlorine, coagulant, coagulant aid and acid, as the lowest value ($score_{min}$) is zero for these criteria. It implies that the standardized score is proportional to the original values, i.e., a twice as high chlorine use is twice as bad. Interval standardization has been preferred over maximum standardization to accentuate the differences in the criteria energy use, water use, land use, and cost. For the *binary* criteria recyclability, SDI, intake and discharge effects, a standardization of midpoint values is not necessary, as the standardized scores are either zero or one.

Linear interval standardization :

$$\text{BENEFIT}_{\text{criteria}}: \quad score_{stand} = \frac{score_{orig} - score_{min}}{score_{max} - score_{min}} \tag{9.1}$$

$$\text{COST}_{\text{criteria}}: \quad score_{stand} = 1 - \frac{score_{orig} - score_{min}}{score_{max} - score_{min}} \tag{9.2}$$

Linear maximum standardization :

$$\text{BENEFIT}_{\text{criteria}}: \quad score_{stand} = \frac{score_{orig}}{score_{max}} \tag{9.3}$$

$$\text{COST}_{\text{criteria}}: \quad score_{stand} = 1 - \frac{score_{orig}}{score_{max}} \tag{9.4}$$

Linear goal standardization :

$$\text{BENEFIT}_{\text{criteria}}: \quad score_{stand} = \frac{score_{orig} - score_{base}}{score_{goal} - score_{base}} \tag{9.5}$$

$$\text{COST}_{\text{criteria}}: \quad score_{stand} = 1 - \frac{score_{orig} - score_{base}}{score_{goal} - score_{base}} \tag{9.6}$$

$$\text{const} = \frac{score_j - score_i}{score_{stand\ j} - score_{stand\ i}} \tag{9.7}$$

9.4.3 Weights

An advantage of MCA is that subjective preferences about a project are made explicit, which increases transparency of the decision making process and allows for a systematic evaluation of different stakeholder perspectives. The subjective preferences are expressed in the form of weights, which are attached to the decision making criteria. In order to gather different perspectives, a questionnaire was prepared and sent to four groups of participants with different perspectives in the area of seawater desalination:

- Environmental perspective (three responses): two persons with university background, one person working in an international company that develops desalination projects.
- Operators' perspective (four responses): three plant managers and one person working in a water authority overseeing desalination projects, all four representing plant capacities between 50,000 and 330,000 m^3/d (two from the Mediterranean region, two from Australia).
- Commercial perspective (five responses): two persons from an international company, three from commercial research institutes.
- Research perspective (six responses): six professors and researchers from universities with expertise in process operation and optimization.

The questionnaire was sent to an equal number of stakeholders within each group, but not all questionnaires were returned. The imbalance between the groups, with three responses for the environmental perspective and six responses for the research perspective, results in a different reliability of the weights derived for each group, i.e., more responses in the group of environmental experts may have resulted in a different set of weights.

The expected value method was used to elicit the expert weights because it is easy to explain and use. The aim was to have a single page with questions that would take no more than 10 minutes to answer. The main group of criteria (chemical, energy, water and land use, material recyclability, water quality (SDI), costs and ecological impact, see value tree in Figure 32) were ranked from highest to lowest importance based on the personal background and experience of each participant (Figure 36). In addition, the single chemicals within the chemicals group were ranked (Figure 37). It was possible to assign the same rank for two or more criteria if these were considered equally important. Weights within a category are often given by experts, as in this study. However, weights between categories that require trade-offs between environmental, social and economic aspects should be given by politicians with a mandate for decision making.

The average rank was calculated for each criterion within each group, and the criteria were sorted accordingly in decreasing order of priority from rank 1 to lowest rank. Figure 36 shows that the persons with an environmental and university background considered ecological impacts as most important, followed by energy use, chemical use, and water quality as lower ranks. In contrast, persons who represent the operators' and commercial point of views gave highest priority to water quality (SDI) and costs, followed by ecology, chemical use and ecology as ranks 3 to 4. The criteria water use, land use and material use occupied the lowest ranks in all groups.

Concerning the different chemicals, all four groups considered coagulant use (rank 1 or 2) of high priority, which indicates that it is important for process operations as well as in terms of environmental impact. Chlorine was also considered very important from an environmental perspective (rank 1), but less important under all other perspectives (rank 3 or 4). Antiscalant and acid are considered moderately important (ranks 2 to 4) in all groups except for the university group, which considers antiscalant as most important (rank 1) and acid as least important (rank 6). The criteria coagulant aid and sodium bisulfite (SBS) ranked lowest in all groups (ranks 3 to 5).

The expected value method converts the rank order of the criteria ($c_1 \geq c_2 \geq c_3$) into a set of quantitative weights ($w_1 \geq w_2 \geq w_3$) by the following algorithm [352, after 353]:

$$w_k = \frac{1}{K} \sum_{i=k}^{K} \frac{1}{i} \quad \text{where } K = \text{ the number of criteria} \tag{9.8}$$

The sum of all quantitative weights is 1. When two or more criteria are considered equally important, that is they have tied ranks, the weight given to each criterion is the average weight for the tied ranks. For example, the responses from the 'university' group led to the following ranking and weights: ecology ($w_1 = 0.340$) ≥ chemicals use ($w_2 = 0.215$) ≥ energy use = water quality ($w_{3,4} = 0.131$) ≥ costs ($w_5 = 0.079$) ≥ land use ($w_6 = 0.054$) ≥ material use ($w_7 = 0.033$) ≥ water use ($w_8 = 0.016$). The higher the importance, the higher the weight of the criterion and the bigger the difference between the next smaller criterion. The expected value method therefore emphasizes the more important criteria, and gives only little weight to the less important criteria. For exam-

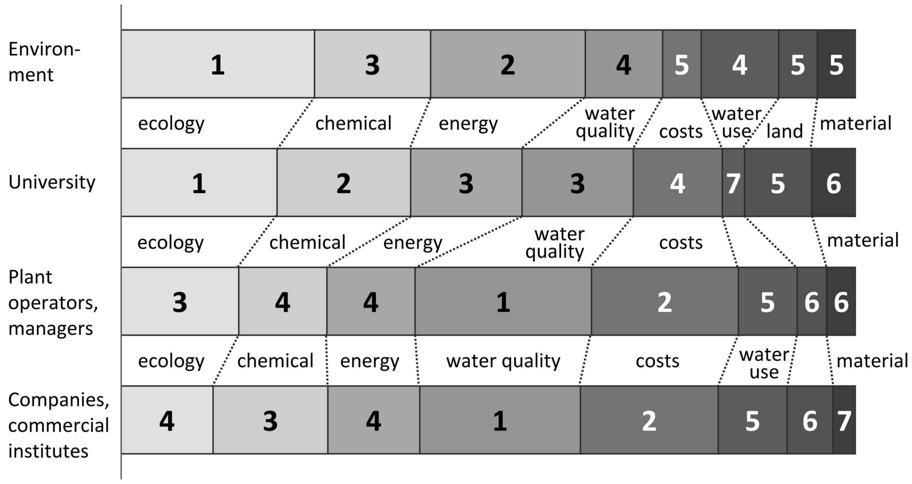

Figure 36: Ranking of the main criteria by the four expert groups. The main criteria, which are given on the horizontal axis of the diagram, were ranked by each expert in the order of their relative importance. A rank of 1 indicates highest importance and is illustrated by the longest segment on the bar that is shown for each group, 2 indicates the second highest importance and is illustrated by a shorter segment, and so on. For example: in the expert groups of plant operators/managers and companies/commercial institutes, water quality was considered on average as the most important criterion (rank 1), followed by costs (rank 2). In contrast, water quality and costs were considered less important in the other two groups, i.e., they ranked 3rd and 4th in the university group, respectively 4th and 5th in the environment group.

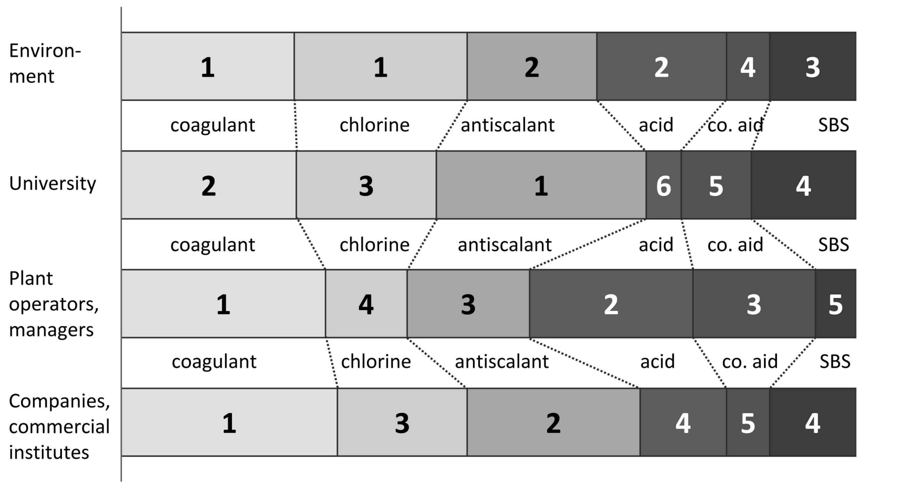

Figure 37: Ranking of the chemical criteria by the four expert groups (see Figure 36 for an explanation of the diagram).

ple, ecology accounts for 34% of the total weight, followed by chemical use with 21%, whereas the two least important criteria account for less than 4% and 2%.

The weights derived for the main group are first level weights, which are multiplied by the second level weights to derive the overall weight of a criterion. Second level weights were established for the single chemicals by expert ranking and the expected value method for each group. The criteria capex and opex in the main group 'costs' were considered equally important, as were the intake and discharge effects in the main group 'ecological impact'.

9.4.4 MCA models

The synthesis of the weights and scores into a ranking is a computational step which depends on the selected MCA model. Weighted summation, which was used in this study, is a compensatory aggregation method, that is, poor performance in any one criterion can be compensated by overall good performance in the other criteria. The overall performance of an alternative is the sum of the alternative's score for each criterion multiplied by the weight given to that criterion [341]:

$$\text{total score } a_j = \sum_{i=1}^{N} w_i \cdot \hat{s}_{ij} \tag{9.9}$$

Where:
A is the set of alternatives with a_j (j=1, ..., M),
C is the set of criteria with c_i (i=1, ..., N),
$\hat{s}_{ij}$ is the standardized score of alternative a_j for effect c_i and
w_i is the weight of effect c_i.

9.4.5 Ranking

The ranking results for the four groups are presented in Figure 38. The outcome that beachwells are also the favored option under different perspectives was to be expected, because the alternative was already found to be dominant by graphical evaluation of the effects table. The analysis however shows that the ranking is generally similar in the two groups that represent plant operators and companies, and in the two groups that represent university and environmental backgrounds. Beachwells obtained the best result in the company and operators groups (total scores of 0.93 and 0.92), followed by conventional pretreatment (0.51,0.53) and UF pretreatment (0.49, 0.47) as ranks two to four with only marginal differences. Beachwells reached a lower total score in the university and environment groups (0.79, 0.82), followed in some distance by conventional II with sludge treatment as the second best alternative (0.41, 0.37), followed by conventional pretreatment I and UF with small differences in total scores as ranks three and four (0.22-0.29).

9.4.6 Sensitivity analysis

The sensitivity of the ranking to changes in weights and scores was investigated. First, the *weights* of the main criteria group were systematically altered. Weights were distributed equally (12.5%) between criteria and then successively changed to 50% for each criterion to emphasize that perspective, while the remaining 50% were equally distributed among the remaining seven criteria (7% each). The sub-criteria (in chemicals, in costs, in ecology) were given equal weight within their groups (Figure 38 bottom, Figure 39 top). The same analysis was carried out for the expert's weights while maintaining the original weights within the subgroups (Figure 39).

In the following, only the changes in ranks two to four are summarized, as the ranking of beachwells is not sensitive to changes in weights. The sensitivity analysis of weights, with results shown in Figure 38 (bottom) and 39, shows that:

- conventional pretreatment I and II rank second, with similar or equal scores, followed by UF as the last rank, if equal weights are allocated to the main criteria, and if 50% is allocated to the criteria water quality (SDI), water use, or material use.
- conventional pretreatment I is the second best alternative if 50% weight is given to cost or energy use, and conventional pretreatment II is the second best alternative if 50% weight is given to ecological impact.
- UF ranks last and only becomes second if 50% weight is allocated to the criteria chemical and land use, or third if 50% weight is given to energy use. However, the differences are marginal between UF and the next lower alternative in all three cases.

Second, the effects of *score* uncertainty on the ranking was analyzed under the assumptions that the scores could be 50% or 100% higher or lower than the selected scores in Table 42. The programme calculates the probability that an alternative ranks at a certain position based on the specified uncertainty values. The probability is calculated from 2000 repetitive outcomes, in which scores are drawn at random from the specified uncertainty limits, assuming a normal distribution function.

Figure 40 shows that beachwell has a high probability of 88-100% to rank at first position in all groups, even given a high uncertainty in the scores. In all groups, either conventional I or II is likely to rank second. In the university and environment groups, conventional II is most likely (69-92%) at the second rank. UF has a high probability of ranking at third or fourth position. A more detailed analysis was carried out for the environmental perspective, the operators' perspective and the neutral perspective in order to identify the reversal values for the five most important criteria which bring about a change in the ranking of the alternatives conventional pretreatment I, II and UF.

For the *environmental perspective*, the most important criteria were ecological impacts, energy use, chemical use, water use, and water quality (SDI, Figure 36). The original ranking was CII as second rank (total score of 0.37), followed by CI (0.26) and UF (0.22, Figure 38). The reversal values were as follows:

- Both CI and UF would rank second if the ecological intake effects or the discharge effects caused by backwash waters were to be eliminated.
- CI would also rank second if the energy demand was reduced by 23%. UF would rank third if energy demand was reduced by only 7%, and second if reduced by 30%.
- With regard to chemical use, only chlorine brought about a change in the ranking: UF could improve its position to third rank if chlorine use was reduced by a significant 65%. The other rankings were not sensitive to changes in chemical use.
- UF would rank third if water use was reduced by 34%.
- The sensitivity of the criterion SDI was not investigated[a].

[a] In an initial value tree, objective 8 (Figure 32) included three criteria (SDI, microorganisms removal and TOC removal) and SDI was defined as a ratio scale, not a binary scale. The main reasons for deleting the other two criteria were a lack of consistent data and the criteria's inaccuracy as a measure for feed water quality. According to Wilf et al. [22], the indicator most relied upon in SWRO is SDI, and even this method is not very accurate. All pretreatment systems are designed to produce the desired feed water quality with an SDI<3 ('yes'). A reversal to 'no' is therefore not a reasonable assumption.

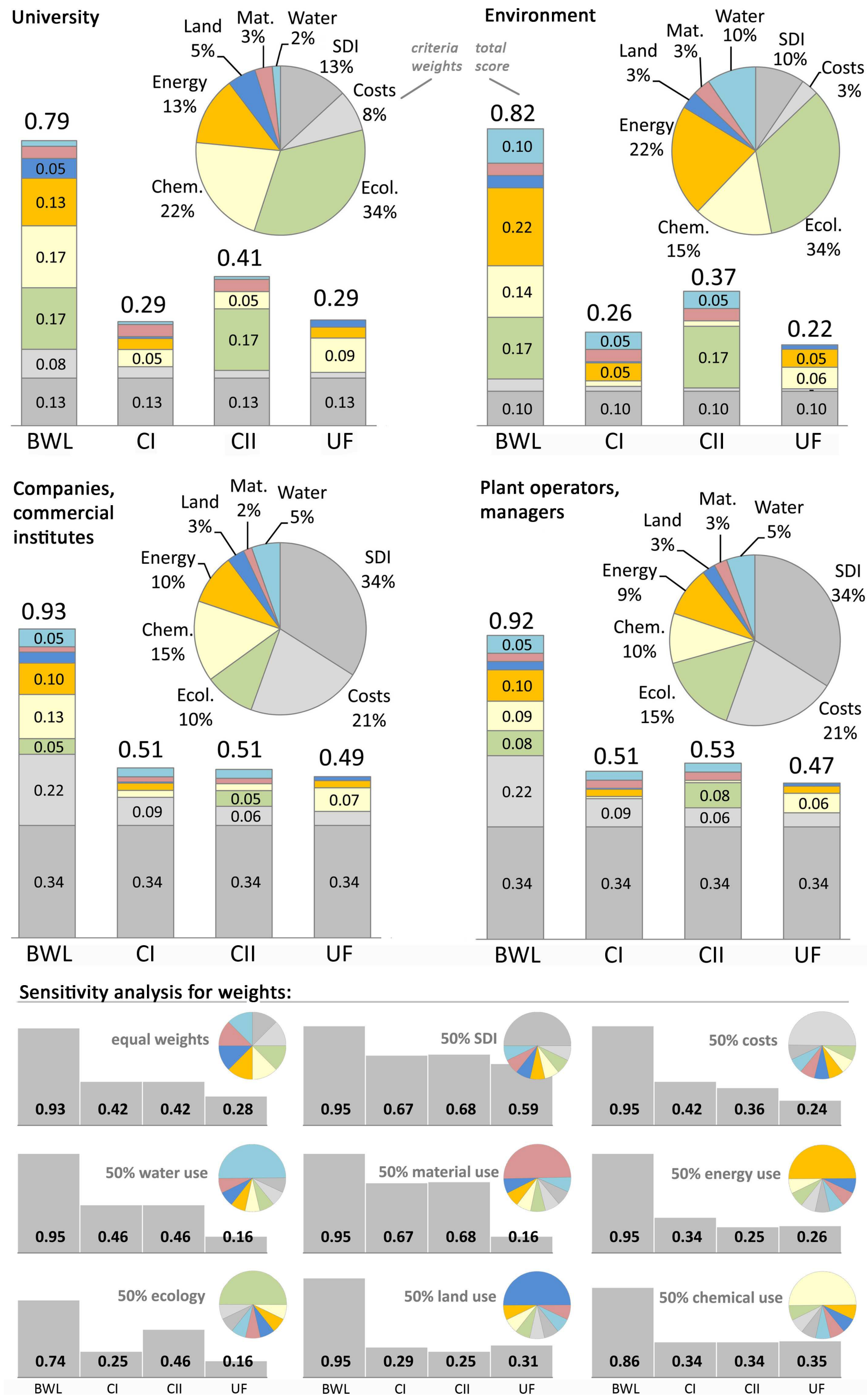

Figure 38: MCA ranking results of the four alternatives by four expert groups (top) and variation of weights to investigate different perspectives (bottom). BWL: beachwell, UF: ultrafiltration, CI: conventional pretreatment, CII: conventional pretreatment with waste water (sludge) treatment.

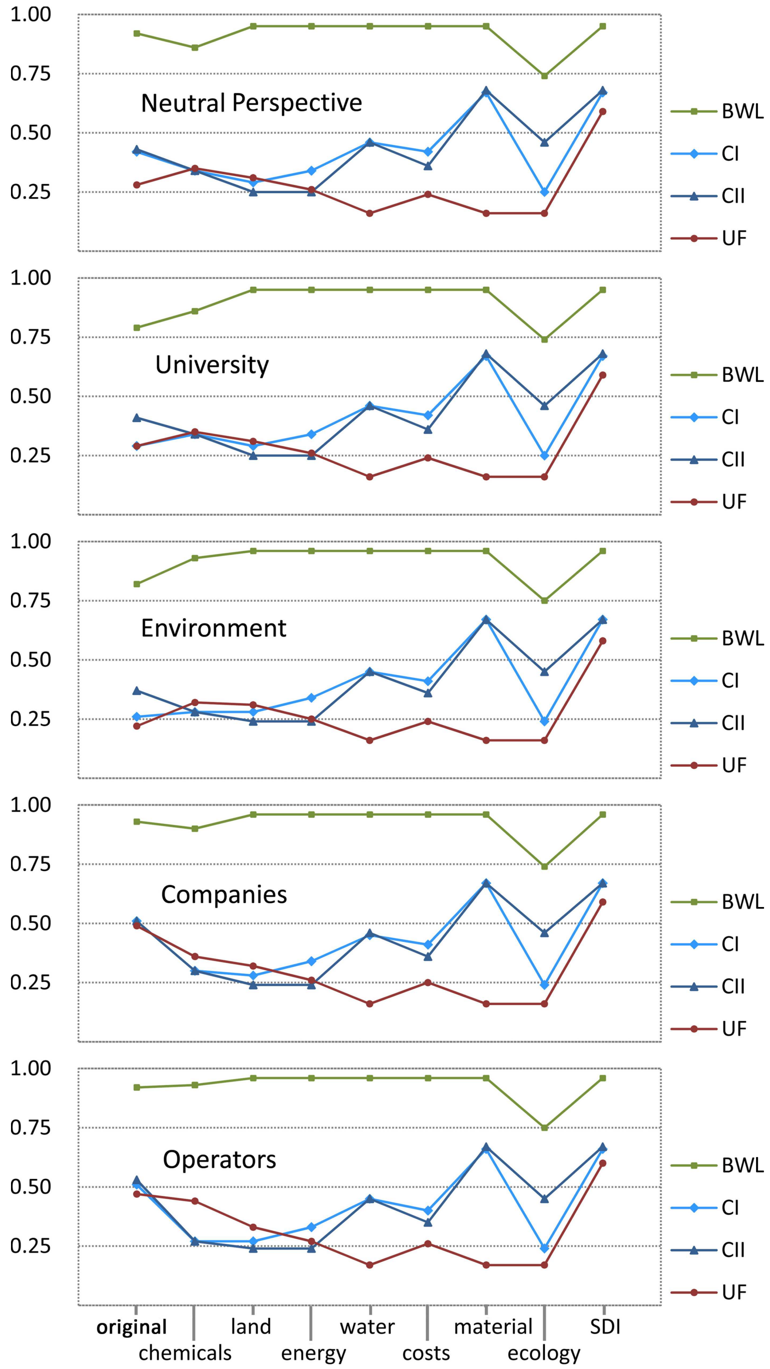

Figure 39: MCA sensitivity analysis of weights: The diagrams show the total scores for the four alternatives as a function of weights. The original ranking with the original weights is shown to the left of each diagram (note: neutral perspective shows the ranking with equal weights in the main group of criteria (12.5% each), and in the sub-group criteria. The data points to the right of the original ranking show how the scores, and hence the ranking changes, if the weights are one by one increased to 50% with the other 50% being equally divided between the remaining main criteria.

From the *operator's point of view*, the most important criteria were water quality (SDI), costs, ecological impact, chemical, and energy use (Figure 36), but the sensitivity of the criterion SDI was not investigated (see footnote on page 206). The order of ranking was the same as for the environmental perspective, but the differences in the total scores were smaller (CII with 0.53 as second rank, CI with 0.51 as third, and UF with 0.47 as fourth rank, Figure 38). The reversal values were as follows:

- CI would rank second if capital cost (capex) was reduced by 26%, or if operational cost (opex) was reduced by 14%. UF would rank third or second if capex was reduced by 31% or 37%, or if opex was reduced by 23% or 39%, respectively.
- Both CI and UF would rank second if the intake effects or the discharge effects caused by backwash waters were to be eliminated.
- CI would rank second if coagulant use was reduced by 65 %. The other rankings were not sensitive to changes in chemical use.
- CI would also rank second if energy demand was reduced by 11%. UF would rank third or second if energy demand was reduced by 16% or 28%, respectively.

For the *neutral perspective* with equal weights (Figure 38 bottom and Figure 39 top), CII ranked second (total score of 0.421), closely followed by CI on third position (0.415). Both clearly outranked UF pretreatment as the fourth position (0.28). CI would rank before CII if:

- the use of the pretreatment chemicals chlorine, coagulant, coagulant aid, acid, or sodium bisulfite were reduced by about 30%, or
- if energy use, land use, or water were reduced by 2%, 5%, or 7%, respectively, or
- if capex was reduced by 11%, or opex by 6%, or
- if intake or discharge effects were to be eliminated.
- The ranking of UF was not sensitive to changes in scores except for the criterion water use. If reduced by 63% or 65%, UF could rise to third or even second rank.

9.4.7 Revised scenarios

The decision problem of the first design round was revised in two consecutive scenarios, in which both the set of alternatives and the number of criteria were changed.

Scenario 1: comparison with existing plants
Case studies with different intake and pretreatment systems were used to investigate if these produce the same ranking results as the selected reference values in the previous MCA. As the reference values had been chosen independently for each criterion from a wide range of literature values, it is possible that the overall result gives a distorted image of the operating conditions of existing plants. For example, chlorine use is still often reported as continuously in the literature, whereas it is in fact used intermittently in many plants. Moreover, beachwells are generally assumed to have a low energy demand in the literature, however, the reported energy demand of a Neodren beachwell intake was slightly higher than that of an identical plant in the same location with an open intake [60]. A third example is that many of the recently commissioned conventional plants have a sludge and waste water treatment system.

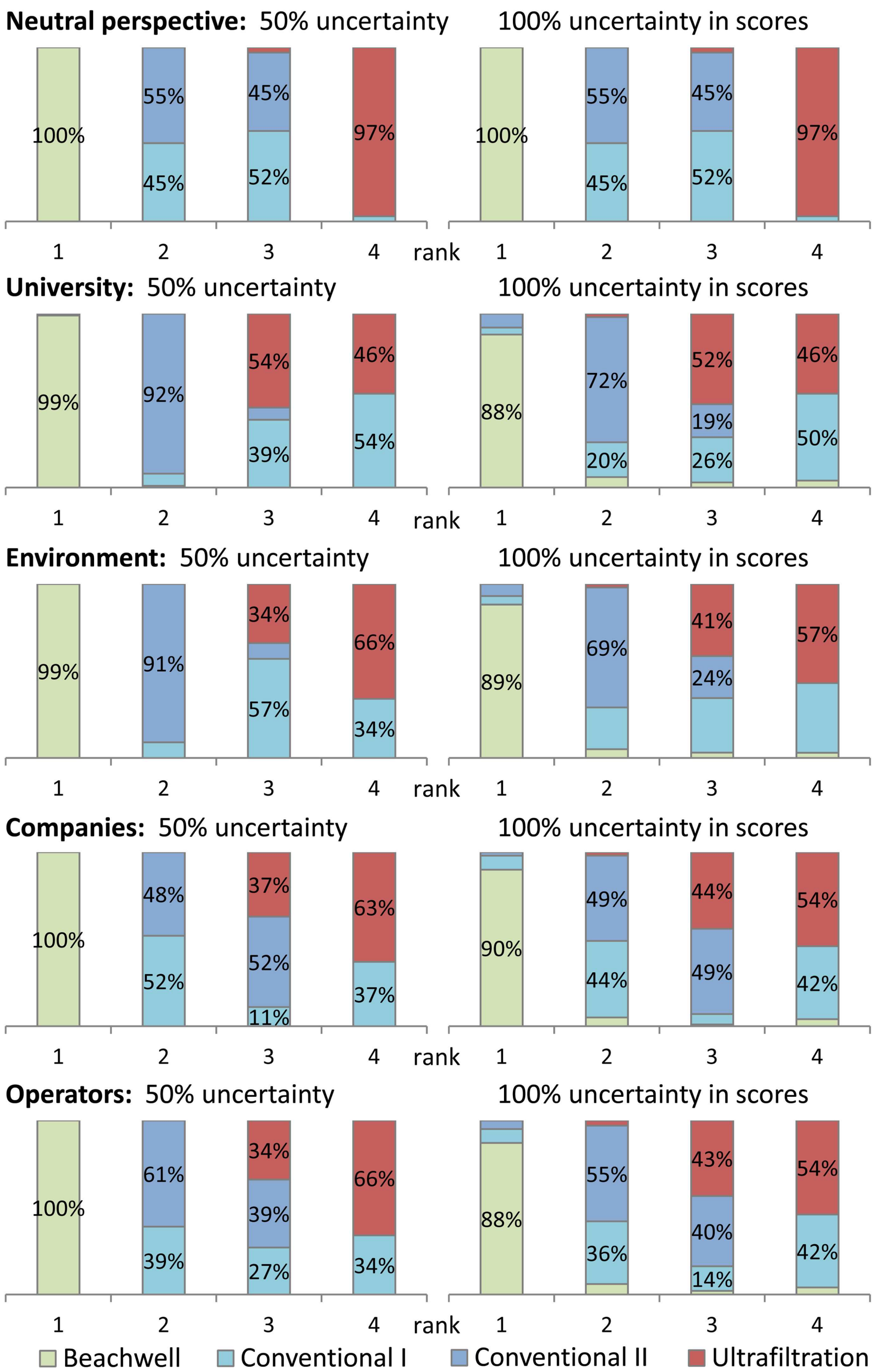

Figure 40: MCA sensitivity analysis of scores.

The revised scenario has the objective to compare the 'hypothetical textbook values' to real plant data. The intake and pretreatment alternatives included the twin plants at San Pedro del Pinatar, Spain (Table 4), of which one has a Neodren beachwell followed by a sand filter, and the other a conventional pretreatment with a two stage dual media filter. A second conventional pretreatment system was included, assuming a single gravity filter as in the Tugun plant in Australia (Table 5). Both conventional systems had a sludge treatment, but were quite different in terms of chemical usage and dosage. Furthermore, a large SWRO plant with a UF pretreatment (Jumeirah Dubai, Table 6) was included.

The value tree was reduced to four main environmental criteria, namely chemical use, energy use, water use, and ecological impact. The other two environmental criteria, i.e., land and material use, were no longer considered because their scores were already rather hypothetical without a good data basis in the first place. Moreover, all expert groups considered them of very low importance. As cost data for the plants could not be obtained, and all alternatives are assumed to achieve an SDI<3, these two criteria were also eliminated. The main objective of the revised scenario was therefore to identify the intake and pretreatment system with the lowest overall impact on the environment.

Although the case studies generally provided a good data basis, a few assumptions were still necessary, as outlined in Table 43. The case study data was used to calculate the scores for a reference plant of 60,000 m^3/d operated at 45% recovery, a design which bears more resemblance to the selected case studies than the assumptions of the original MCA. The scores were standardized using interval standardization. The criteria ranking order from the environmental expert group was used to calculate the weights and the total scores, using weighted summation. A perspectives analysis was carried out, consecutively assigning equal weights and then 50% weight to each main criterion in order to investigate how the ranking changes. The results were as follows:

- The Neodren beachwell turned out to have the lowest overall environmental impact given the selected criteria (total score 0.73), followed by the twin plant in the same location with an open intake and minimal chemical pretreatment (0.54), closely followed by the more extensive conventional pretreatment (0.52).
- This ranking applied to all perspectives except if 50% weight is placed on energy use, in which case UF pretreatment ranked first (0.59) but is almost equal to the two conventional pretreatments (CII-b: 0.58, CII-a: 0.57). UF pretreatment ranked at the lowest position except for this and the chemical use perspective, in which it was only marginally better (0.01) than the more extensive conventional pretreatment.

The results of the revised scenario using the operation and design specifications of real full-scale plants support the findings of the previous MCA, in which beachwells ranked first, followed by conventional pretreatment and UF pretreatment.

Table 43: MCA reference values and calculated scores − revised scenario 1.

Criterion	Beachwell San Pedro del Pinatar I	calc. score for 60,000 m^3/d	Conventional II-a San Pedro del Pinatar II	calc. score for 60,000 m^3/d	Conventional II-b Tugun, Australia	calc. score for 60,000 m^3/d	UF Jumeirah, Dubai	calc. score for 60,000 m^3/d
Details of case study plant	65,000 m^3/d capacity, Neodren wells, sand filters, sludge treated		65,000 m^3/d capacity, open intake, two stage DMF, sludge treated		125,000 m^3/d capacity, offshore submerged intake, gravity DMF, sludge treated		64,000 m^3/d capacity, UF recovery of 90%, backwash every 45 min., backwash discharged	
Chlorine (feed water)	**–** none used	**–**	**–** none used	**–**	**10 mg/l** shock dose, assumed to take place for 1 h/day	**57 kg/d**	**15 mg/l** for 20 min. 1x/day	**31 kg/d**
Chlorine (backwash)	**–** does not apply	**–**	**–** does not apply	**–**	**–** does not apply	**–**	**200 mg/l** chlorine CEB 1x/day	**83 kg/d**
Sodium bisulfite	**–** none used	**–**	**–** none used	**–**	**3 mg/l** prior to RO, assumed to take place for 1 h/day	**17 kg/d**	**–** normally not required (only if hypochlorite break-through is detected)	**–**
Coagulant	**–** none used	**–**	**1.6 mg/l Fe** calc. from 4 mg/l $FeCl_3$ assum. 40% active ingr.	**213 kg/d**	**7.2 mg/l Fe** calc. from 18 mg/l $FeSO_4$ assum. 40% active ingr.	**982 kg/d**	**0.3 mg/l Fe**	**44 kg/d**
Coagulant aid	**–** none used	**–**	**–** none used	**–**	**0.2 mg/l** 0-0.4 mg/l polyelectrolyte	**27 kg/d**	**–** none used	**–**
Acid (H_2SO_4)	**–** occasionally	**–**	**50 mg/l** acid to pH 6.5, assumed to require 50 mg/l H_2SO_4	**6,667 kg/d**	**35 mg/l** H_2SO_4	**4,772 kg/d**	**275 mg/l** for 20 min. 1x/day	**560 kg/d**
Antiscalant	**1 mg/l** range of 0.9-1.2 mg/l	**133 kg/d**	**1 mg/l**	**133 kg/d**	**2 mg/l**	**267 kg/d**	**2 mg/l**	**267 kg/d**
Energy use	**0.18 kWh/m^3** 0.40 kWh/m^3 permeate for intake/pretreat., value of 0.18 kWh/m^3 calculated assuming 45% RO recovery	**24,000 kWh/d**	**0.16 kWh/m^3** 0.35 kWh/m^3 permeate for intake/pretreat., value of 0.16 kWh/m^3 calculated assuming 45% RO recovery	**21,000 kWh/d**	**0.14 kWh/m^3** value extrapolated from 0.16 kWh for intake + two stage DMF [60], 0.018 kWh for one gravity DMF [53]	**18,667 kWh/d**	**0.12 kWh/m^3 filtrate** 0.03 kWh/m^3 for UF 0.08 kWh/m^3 for intake 0.01 kWh/m^3 for screening	**16,000 kWh/d**
Water use	**–** backwash with concentrate	**–**	**–** backwash with concentrate	**–**	**2.25%** of pretreated water assumed (1000 m^3 1x/8 h)	**3000 m^3/d**	**10%** of filtrate	**13,333 m^3/d**
Pot. ecol. impact	impingement/entrainment? backwash discharged?	**no** **no**	impingement/entrainment? backwash discharged?	**yes** **no**	impingement/entrainment? backwash discharged?	**yes** **no**	impingement/entrainment? backwash discharged?	**yes** **yes**

Table 44: MCA reference values and calculated scores – revised scenario 2.

Criterion	**UF-a (Dow) selected reference value**	**calc. score for 60,000 m^3/d**	**UF-b (Pall) selected reference value**	**calc. score for 60,000 m^3/d**	**UF-c (Inge) selected reference value**	**calc. score for 60,000 m^3/d**	**UF-d (Norit) selected reference value**	**calc. score for 60,000 m^3/d**
for further details on reference values see Table 45	partial data from 3 full-scale and 2 pilot plants and empirical values of the manufacturer		partial data from 1 full-scale and 3 pilot plants		partial data from 1 full-scale and 1 pilot plant		partial data from 3 full-scale and 4 pilot plants	
Chlorine (feed water)	**2 mg/l** resulting in 0.5 mg/l chlorine residual	**285 kg/d**	**1 mg/l**	**140 kg/d**	**1.5 mg/l**	**214 kg/d**	**15 mg/l** for 20 min. 1x/day	**31 kg/d**
Chlorine (backwash)	**15 mg/l** every 60 min. (every backwash)	**140 kg/d**	**10 mg/l** every 30 min. **500 mg/l** every day	**67 kg/d** **69 kg/d**	**20 mg/l** every 24 h	**6 kg/d**	**200 mg/l** chlorine CEB 1x/day	**83 kg/d**
Sodium bisulfite	**1.5 mg/l** 3x residual chlorine level	**200 kg/d**	**0.75 mg/l** 1 mg/l chlorine dose assumed to result in 0.25 mg/l chlorine residual, extrapolated from Dow value	**100 kg/d**	**1.2 mg/l** 1.5 mg/l chlorine assumed to result in 0.4 mg/l chlorine residual, extrapolated from Dow value	**150 kg/d**	**–** normally not required (only if hypochlorite breakthrough is detected)	**–**
Coagulant	**–** none used	**–**	**0.4 mg/l Fe**	**56 kg/d**	**–** none used	**–**	**0.3 mg/l Fe**	**44 kg/d**
Energy use (UF only, without intake/ screening)	**0.095 kWh/m^3**	**12,000 kWh/d**	**0.1 kWh/m^3** no data available, energy use assumed similar to the Dow value	**13,333 kWh/d**	**0.05 kWh/m^3**	**6,667 kWh/d**	**0.03 kWh/m^3**	**4,000 kWh/d**
Water use	**7%** 93% UF recovery	**9,333 m^3/d**	**5%** 95% UF recovery	**6,667 m^3/d**	**7%** 93% UF recovery	**9,333 m^3/d**	**10%** 90% UF recovery	**13,333 m^3/d**

Table 45: UF membrane types and their pretreatment and energy requirements.

Membrane supplier	Chlorine to feedwater	Coagulant	CEB	CIP	Water	Energy
Inside feed – always in housing, always pressurized, mostly PES (Norit, Hydranautics, Inge membranes)						
Norit partial data from 3 full-scale and 4 pilot plants	1.5-25 mg/l ClO_2 1x/day for 15-30 min. in 1 full-scale plant reported; 2 pilot plants did not use chlorine, the rest did not specify this information	0.3 mg/l as Fe in full-scale plant, 6 out of 7 pilot/full-scale plants used coagulant in doses of 1.3-1.5 mg/l as Fe and 5 mg/l as $FeCl_3$ (≡ 2 mg/l Fe, 40% active ingredient)	200 mg/l Cl_2 1 min./day (full-scale), NaOCl used in 4 pilot plants, 2 reported a dose of 200 mg/l 1x/6-18 h 3 pilot and 1 full-scale plant used acid (HCl, pH 2.5) 1x/3-24 h	CIP: 0.5% oxalic acid and 0.25% ascorbic acid 1x/9 months (full-scale)	94% recovery / BW every 45 min. (pilot plant), 90% recovery (full-scale plant)	full-scale plant: 0.03 kWh/m^3 filtrate plus 0.08 kWh/m^3 for intake 0.01 kWh/m^3 for screening 0.12 kWh/m^3 total; 0.1-0.2 kWh/m^3 reported range in papers
selected reference values	shock chlorination 15 mg/l for 20 min. 1x/day	0.3 mg/l as Fe	200 mg/l Cl_2 1x/day	0.5% oxalic acid and 0.25% ascorbic acid CIP 1x/9month	90% recovery; BW every 45 min.	0.03 kWh/m^3
Hydranautics partial data from 2 full-scale and 5 pilot plants and empirical values of the manufacturer	3 full-scale plants gave the information that CaOCl is used for shock treatment at the intake; 1-5 mg/l manufacturer's empirical value for intermittent chlorine use (optionally)	3 pilot plants specified a dose of 0.1-0.7 mg/l as Fe; 1 full-scale plant found the design dose of 1 mg/l as Fe not necessary; manufacturer's empirical value: 0.5-1 mg/l as Fe	Cl_2 in 1 full-scale and (20 mg/l Cl_2 1x/8-2 h) 1 pilot plant (20-50 mg/l, pH 12, 1x/6 h); acid CEBs with H_2SO_4 (full-scale) and pH 1.5-2 (pilot); manufacturer's empirical value: 50 mg/l Cl_2 1x/4 h	CIP 1x every 1-2 months	95% recovery / BW every 45-70 min. (1 full-scale), 94% recovery / BW every 30 min. (1 pilot), i.e., 6% for BW and rinse air enhanced BW 1x/day	manufacturer's empirical value: 0.1 kWh/m^3 of filtrate
selected reference values	shock chlorination 3 mg/l assumed for 20 min. 1x/week	none (0.5 mg/l Fe optionally)	20 mg/l Cl_2 1x/12 h H_2SO_4 (pH 1.5-2)	no dosing information CIP 1 x per month	95.5% recovery BW every 45 min	0.1 kWh/m^3
Inge partial data from 1 full-scale and 1 pilot plant	1-2 mg/l in the pilot plant, not specified in the full-scale plant	full-scale plant: none; pilot: 0.25 mg/l as Fe found to be sufficient	20 mg/l Cl_2 1x/day in the full-scale plant; 50 mg/l NaOCl 1x/2-3h in the pilot plant	CIP with HCl 1x/3days in the full-scale plant if necessary (pH 12); citric acid (1% or 10,000 mg/l) in the pilot plant	93% recovery in the full-scale plant, BW every 45 min.	pilot plant: TMP of 0.1-0.2 bar without $FeCl_3$, 0.2-0.25 bar with $FeCl_3$, 0.05 kWh/m^3 filtrate at 0.25 bar full-scale plant: 0.3 bar without $FeCl_3$
selected reference values	1-2 mg/l optionally	none (0.25 mg/l Fe optionally)	20 mg/l Cl_2 1x/24 h	acid CIP 1x/3 days HCl at pH 12	93% recovery BW every 45 min	0.05 kWh/m^3

continued on next page

Membrane supplier	Chlorine to feedwater	Coagulant	CEB	CIP	Water	Energy
Outside feed – in housing and pressurized, mostly PVDF (Dow/Omexell, Pall/Asahi)						
Dow partial data from 3 full-scale and 2 pilot plants and empirical values of the manufacturer	full-scale plants: (a) 2 mg/l (0.5 mg/l residual), followed by SBS; (b+c) no chlorine; pilot plant: 0-6 mg/l; empirical value: 0.5 mg/l as residual	no coagulant in the full scale plants; 0-2.4 mg/l as Fe used in the pilot plants	full-scale plant (a) also uses 15 mg/l NaOCl with every BW (1x/h); full-scale plant (c) no CEB pilot plants: 20 mg/l NaOCl 1x/30 min. and 500 mg/l 1x/12-120 h, respectively	2 full-scale plants use CIPs: (a) 500 mg/l NaOH and 2000 mg/l NaOCl followed by 3600 mg/l HCl once per year (c) 20,000 mg/l oxalic acid followed by 2000 mg/l NaOCl	empirical value: 7% for BW, CEB, CIP, 93% recovery; BW every 60 min. with air scrub	empirical value: TMP 0.5 bar 0.09-0.1 kWh/m^3 filtrate
selected reference values	2 mg/l (0.5 mg/l residual) optionally	none	15 mg/l NaOCl 1x/h optionally	500 mg/l NaOH and 2000 mg/l NaOCl followed by 3600 mg/l HCl, 1x/year	93% recovery BW every 60 min.	0.095 kWh/m^3
Pall partial data from 1 full-scale and 3 pilot plants	1 pilot plant disinfects the MF filtrate with UV and chlorine (1-2 mg/l for 30 min./day), no information besides this	0.3-1.5 mg/l $FeCl_3$ (≡ 0.1-0.6 mg/l Fe, 40% active ingredient) in the full-scale plant; pilots used 0-3.5 mg/l $FeCl_3$ (≡0-1.4 as Fe)	10 mg/l Cl_2/NaOCl or 25 mg/l H_2O_2 every 17-40 min., and/or 500 mg/l Cl_2/NaOCl 1x/day in the pilot plants, no information given for the full-scale plant	CIPs with caustic, chlorine and acid in 2 pilot plants: 10,000 mg/l NaOH and 1000 mg/l NaOCl, 10,000 mg/l citric acid 1x/month	recovery in pilot plants was 95-96% with a BW every 20-40 min., simultaneous air scrub reverse flow	no information
selected reference values	1 mg/l assumed	0.4 mg/l as Fe	10 mg/l NaOCl 2x/h 500 mg/l 1x/day	10,000 mg/l NaOH and 1000 mg/l NaOCl and 10,000 mg/l citric acid 1x/month	95% recovery BW every 30 min.	0.1 kWh/m^3 assumed

Selected reference values: Information from full-scale plants was preferred over pilot plants, where available. If several values were available, a representative midpoint value was selected. If only data from pilot plants was available, the lowest reported value was selected to take into account that pilot studies may test higher doses than would be used under real operating conditions.

Scenario 2: comparison of UF pretreatments
In order to implement a UF pretreatment successfully, the backwashing and CEB intervals, and the chemical doses added to the feed and CEB need to be optimized. For example, it would be possible to operate a UF plant with no or only little coagulant and chlorine addition to the feedwater, but with more frequent backwashing and chlorine enhanced backwashing (see page 43). Another factor in this optimization problem is energy use. The TMP increases when deposits build up on the UF membranes, and backwashing is usually carried out at 2-4 times the filtration flow. Both cause an increase in energy demand. Moreover, some UF manufacturers employ an air scrub, in which pressurized air is introduced with the backwash water. This improves the cleaning process, possibly reducing chemical but increasing energy demand.

The hypothesis was that different UF pretreatment systems optimize their chemical use, energy use, CEB and backwashing intervals differently. The objective was to compare different membrane types in terms of chemical, energy and water use. Four different UF membranes were considered as alternatives, two are pressurized inside-out UF membranes (Norit, Inge) and two are pressurized outside-in membranes (Dow, Pall). A reference value was selected, based on an inventory of full-scale plants, pilot plants, and general manufacturer information, and used to calculate the scores. Preference was given to data from full-scale plants, if available (Table 45).

The value tree was further reduced from scenario 1, eliminating the criteria coagulant aid, acid (none used), and antiscalant. Antiscalant is usually added after the filtration stage and does not tell something meaningful about the different UF systems. Acid can be used in SWRO to lower the risk of scaling on the RO membranes and is not related to the filtration step in that case. Acid can also be used to control the pH during coagulation before the filtration step (as assumed in the previous MCA). However, most UF pretreatment systems use in-line coagulation, for which acid use is not anticipated here to reduce complexity. The criterion ‘ecological impact’ was eliminated as all UF systems alike are assumed to operate on open intakes and to discharge their backwash waters without treatment. The remaining criteria were chlorine, SBS and coagulant addition to the feedwater, chlorine use in backwashing, energy and water use.

As in scenario 1, interval standardization and weighted summation was used to calculate the scores for a reference plant size of 60,000 m^3/d (Table 44). A perspectives analysis of the weights was carried out assuming (i) the original ranking of the environmental group, (ii) equal weights for all main criteria, and (iii) 66% weight allocated to energy, chemicals or water use. The results were as follows:

- The inside-out UF systems (Norit, Inge) ranked at first and second positions and the outside-in systems (Dow, Pall) at third and fourth position in all perspectives, except if 50% weight is given to water use. In that case, Pall ranked first, followed by Inge, Dow, and Norit on last position. As PVDF in outside feed formats (Dow and Pall membranes) allows the use of air scour, which reduces water consumption for backwashing, one would have expected Dow to score better. However, the same water use (7%) has been reported for backwashing and CIP of Dow and Inge systems (Table 45).

- Norit ranked first for the environmental group’s perspective and the energy use perspective, while Inge ranked first for the equal weight and chemicals perspectives.

- Pall ranked last for the environmental group’s perspective, the energy and chemicals perspective, while Dow ranked last if qual weights are given to all criteria.

It is noteworthy that the Dow reference values are based on the full-scale plant in Wang Tan, China (9,600 m^3/day UF capacity, 3,120 m^3/day RO capacity, cf. Table 6), which does not use any coagulant but high amounts of chlorine due to both feed and backwash chlorination. If one assumes the Dow Magong plant in Taiwan as baseline instead, which uses neither coagulant nor chlorine, the ranking of Dow changes to first position in the 'chemicals perspective' and to second position behind Norit in the 'equal weights perspective'. A main difference between the Wang Tan and Magong plants is the frequency of CIP employed for the UF membranes, which are given with once a year for Wang Tan and once a month for Magong [86, 87].

CIP is another decisive factor in the design of an integrated membrane system, and refers to the CIP frequency of both the UF and SWRO membranes. CIP has not been included in the MCA because representative quantitative values are difficult to establish. The CIP intervals for UF plants generally vary between once a month to once a year, as do the intervals for the SWRO membranes. Most typically, strong alkaline solutions up to 10,000 mg/l NaOH are used to clean the UF membranes, in combination with chlorine (2,000 mg/l NaOCl), and/or followed by acid (up to 10,000 mg/l, Table 45). For example, the CIP in Magong involves a 2% oxalic acid ($H_2C_2O_4$) solution followed by a 0.2% sodium hypochlorite (NaOCl) solution [60]. The cleaning solutions are circulated through the UF modules in concentrations equal to 20,000 mg/l $H_2C_2O_4$ and 2,000 mg/l NaOCl, equivalent to about 0.4 and 4 mg/l if they were to be used continuously [131].

9.5 Summary and conclusions

The objective of this chapter was to develop a decision support system for seawater desalination plants using MCA, which may facilitate the planning and EIA process of new desalination projects. The decision support system was to be implemented and tested in a case study in order to evaluate its feasibility for seawater desalination plants. Three levels of conclusions can therefore be drawn from this study – conclusions pertaining to the MCA case study in specific, conclusions with regard to MCA in general, and conclusions and recommendations with regard to the role of MCA in the EIA and decision making process for new desalination developments.

Case study
As outlined in the introduction, a comparison of different intake and pretreatment systems for SWRO plants was chosen as the case study for the MCA. The approach deliberately eclipsed other desalination technologies and other aspects of designing a desalination plant in order to limit the complexity of the decision problem. This can be justified because the decision between the two main technologies, i.e., reverse osmosis and distillation, is a fundamental one and usually depends on the availability of a cheap energy source (therefore, it could have taken place in a preliminary design round). Other design considerations, such as the number of RO stages or post-treatment requirements, depend on the product water specifications and are also independent from the pretreatment.

Even though the decision problem had been narrowed down in complexity and level of detail, it proved difficult to establish a complete set of relevant intake and pretreatment alternatives and operational criteria, and to gather the necessary input data for the MCA. It was realized during problem definition that the study, which had been assumed to be only a prelude to more detailed analyses, would not be as straightforward as expected. On the one hand, a promising alternative with a presumably low impact on the environment (Neodren) had to be eliminated because data was mostly confidential or unavailable.

On the other hand, certain criteria, such as those used to measure water quality, simply proved inadequate. Although SDI had been included in the main MCA for the sake of completeness, it contained no real information that allowed for a differentiation between the alternatives. For some criteria, such as land use or costs, information was hardly available, so that values had to be estimated and extrapolated. Even for an important aspect such as energy use, which is widely discussed in the literature, it was difficult to establish reference values. A value of 0.1-0.2 kWh/m^3 is usually given for intake and pretreatment in general, which needed to be broken down into the specific demands of different media filters for conventional and membrane types for UF pretreatment.

The problems that were encountered may be attributed to the hypothetical nature of the study. Data uncertainty is an intrinsic problem of EIAs, and the same holds true for the MCA. In both cases, the results can only be as good as the underlying data. However, it makes a difference if one considers real or hypothetical alternatives. In real project EIAs and MCAs, a whole team of consultants, engineers and scientists ideally works on the task of providing the data basis for the analyses. One participant in the plant operators' group criticized that the MCA was 'overly simplistic' and not 'fully scoped and developed', based on the person's own experience with MCA for selecting a desalination plant site[b]. The fact is that the MCA has been deliberately narrowed down after it had been realized that the given data would not support a more complex decision problem, as it would possibly be the case in a real life scenario.

The main obstacle in this study was clearly to establish a complete set of representative reference values for all alternatives on the various criteria. The confidence in some of the scores, despite an extensive literature search and communications with consultants and plant managers, is therefore limited and could be improved by more accurate data from operational plants. A sensitivity analysis was therefore carried out to investigate the effects of data uncertainty on the ranking. Despite this data uncertainty, a few general conclusions can nevertheless be drawn from the MCA results:

- Beachwells were found to be dominant over the other alternatives, that is, beachwells scored better or at least as well as the other alternatives on all criteria. Beachwells therefore also ranked first in the MCA, irrespective of the weights that were attached to the criteria to represent different perspectives.
- The MCA showed that the value judgments were generally similar in the two groups that represented plant operators and companies on the one hand, and in the two groups that represented university and environmental backgrounds on the other hand. While plant operators gave highest priority to the criteria water quality and costs, the university and environmental groups gave highest priority to ecological impact. Chemical and energy use varied in importance between ranks two to four in all groups. The most important chemical substance was considered to be coagulant.
- The MCA ranking showed that conventional pretreatment II (with sludge treatment) was the second best alternative according to the value judgments of the expert groups. The preference over conventional pretreatment I was more distinct in the university and environmental groups (>0.1 total score difference) than in the plant operators group, in which conventional II ranked second by a narrow margin (0.02 difference), and in the company group, in which both conventional pretreatments had equal preference.

[b] The person returned the questionnaire and the answers did not seem to deviate significantly from the other participants in that group, i.e., the person cannot be assumed to be an 'outlier'.

- Ultrafiltration ranked at the last position in all groups, although the difference to the next best alternative, i.e., conventional I, was small in all four groups (0-0.4 difference).
- A sensitivity analysis of weights showed that conventional pretreatment I would rank before conventional II if the weights of the criteria cost or energy were to be increased to 50%. This can be attributed to the assumption in this study that sludge treatment causes a 10% increase in cost and energy use. UF would rank second if the weights of the criteria chemical or land use were to be increased to 50%.
- A sensitivity analysis of scores, assuming 50% and 100% score uncertainty, showed that beachwells had a high probability to rank first. Either conventional pretreatment I or II was likely to rank second. UF pretreatment had a high probability of ranking on third or fourth position. In order to reverse a ranking, either discharge or intake effects had to be eliminated, or the scores for the various aspects of resource consumption had to be reduced by about 20-60%. A decisive criterion was energy use. If reduced by 11% or less, it brought about a change in the ranking between the alternatives conventional pretreatment I and II, which can again be attributed to the assumed higher energy use for conventional pretreatment II.

The MCA ranking and sensitivity analysis support the conclusion that a beachwell is the preferred intake and pretreatment alternative for SWRO plants. Where a beachwell is not feasible, for example, due to an impermeable geologic substratum or due to size restrictions, the preferred option would most likely be a conventional pretreatment, either with or without sludge treatment, followed by ultrafiltration.

The results of the revised scenario 1, in which the specific operational conditions of selected full-scale plants were used as baseline for the MCA instead of selected literature values, also support these findings. Natural intake systems (i.e., Neodren) were also the preferred choice for larger SWRO plants, followed by conventional and UF pretreatment.

The results of the revised scenario 2, in which different UF membranes and modes of operation were compared, showed that the successful operation of an integrated membrane system is essentially an optimization problem, which has to be solved plant- and site-specifically, balancing energy demand, chemical use, filtration time, CEB and CIP intervals. Similarly to conventional pretreatment systems, which have diversified into various pretreatment options over the years ranging from minimalist to an extensive three stage design, not all UF systems are alike, let alone their modes of operation. The range of possible operation modes of UF systems shows that a more sustainable approach with a low energy and chemical demand is feasible in principle, and that this approach could be altogether equal to or even better than a conventional pretreatment.

This MCA should be understood as an exercise, which can always be revised and refined in the light of better data and new information. The results are only valid for the given alternatives and criteria. The inclusion of new or modified alternatives and criteria may alter the ranking results. For example, the alternative UF pretreatment will score better on the ecology criterion if one assumes that the backwash water is treated instead of discharged, i.e., by modifying the present assumptions for the alternative. Although discharge seems to be the common practice of the few operational UF-SWRO plants today, future projects in Australia or California may require a treatment of the backwash. Also, UF might perform better if better indicators for water quality were available. Pilot studies often found UF pretreatment superior to conventional pretreatment in difficult feed waters, which may have secondary beneficial effects on plant operations, such as lower cleaning frequencies of SWRO membranes, lower energy demand or lower operating costs. However, quantifying these effects in an MCA would still be highly speculative at the moment, due to both, a lack of reliable indicators for water quality and data.

Natural intake and pretreatment systems, such as beachwells or sub-seabed infiltration galleries, performed best in this MCA. One recommendation for MCAs is therefore to include a criterion which considers the feasibility of a natural system during site selection. However, a natural intake will not be feasible for all projects, for example, because of size limitations of the beachwell, or may not be the preferred alternative in all cases, for example, due to anoxic or anaerobic conditions in the ground water. Anoxic or anaerobic feed water from a beachwell poses the risk that if oxygenated, iron or manganese flocs may form, which need to be retained by a filtration step. If the solids content of the filter backwash is too high to allow for discharge, treatment of the backwash water may be required. In that way, the advantages of a beachwell over an open intake with conventional pretreatment or ultrafiltration could be diminished.

An acceptable alternative where a subsurface intake is not possible is an open intake with conventional pretreatment, followed by an open intake with ultrafiltration. However, only conventional pretreatment systems have been implemented and successfully used as a pretreatment for large SWRO plants to date. Experiences with regard to subsurface intakes, which would probably have to be horizontal drains in the offshore sediments to provide a sufficient feed flow, and ultrafiltration for large SWRO projects is lacking.

The conclusion of this MCA that conventional pretreatment systems altogether outrank ultrafiltration is supported by a life cycle analysis (LCA) that was carried out simultaneously at the Technical University of Berlin. Input data for the two analyses were exchanged and discussed in several meetings. While the MCA allows for an objective analysis of the performance of alternatives as well as subjective value judgments with regard to multiple criteria, the LCA primarily evaluates the performance with regard to chemical and energy streams over the life cycle of a project. The researchers at TU Berlin concluded that "the gravity media filter is currently still a more sustainable technological solution" when working with non difficult waters, and call for "further optimization of UF design and operation concerning the over-all process sustainability" [354].

The potential for future improvements is high, given the fact that UF pretreatment is still a young technology in its learning stage. In this regard, it may also learn from past shortfalls of conventional pretreatment systems, such as that backwash waters and spent cleaning solutions should be collected and treated, rather than discharged. To increase the acceptance of UF, most membrane suppliers and researchers have focused on operational and economic aspects so far. According to the value judgment of plant operators and company representatives in this case study, these aspects have a higher priority than ecological aspects, energy demand and chemical use. However, it may be shortsighted to neglect the latter because they are considered to be of secondary importance. Obviously, UF technology will never achieve a real breakthrough over conventional pretreatment if it does not perform equally well or better in terms of water quality. However, environmentally-friendly designs may increase the acceptance and could tip the balance in favor of UF despite increased costs, especially in emerging markets such as Australia. In Australia, everything that was required has been done so far "to ensure desal works environmentally, with price being a secondary consideration" [297].

MCA approach

It can be concluded that MCA is a suitable tool for assessing both the site and process alternatives for desalination plants. As outlined in the introduction, the approach has previously been used and proved useful in selecting desalination plant sites. This study showed that MCA can furthermore be used to compare and rank process alternatives, despite the limitations of a hypothetical study. It is anticipated that MCA will prove to be more powerful in real applications, in which uncertainty is limited to weights and scores, and where the decision maker does not have to worry about the additional uncertainty that is imminent to a study that is hypothetical in nature.

As site and process design alternatives are usually interdependent (e.g., a beachwell is not feasible in all locations), both aspects should preferably be evaluated simultaneously and not in separate design rounds, which may produce sub-optimal results [295]. An integrated approach would inevitably result in a higher complexity of the decision problem, due to a larger number of alternatives and criteria that need to be considered. However, this study also showed that even a decision problem of presumably limited complexity may face methodological challenges if put to practice.

One main challenge in this study was structuring the decision problem, which had to be revised several times just to be able to carry out a first analysis. The MCA was subsequently refined again in two scenarios, and even at that stage, it had not arrived at an end but could still have been revised in the light of better data and new information. This leads to another main challenge of this study, which was the establishment of a sound data and information basis for calculating the scores. In order to justify the selected reference values, it was necessary to delve into the details of different process alternatives, even if this was not always successful because the sought-after information was unavailable or sometimes confidential.

This tells us two aspects about MCA as a decision support tool, which have also been stated in this form or another in the scientific literature. First, MCA requires that the decision maker thinks thoroughly about the decision problem, that is, about its objectives, all relevant alternatives, and meaningful evaluation criteria. Second, it compels the decision maker to build a knowledge base and to gain a thorough understanding of the decision context. Both facilitate the decision maker's learning about the problem faced, and both pertain to the two initial steps of MCA, which are problem definition and generation of input data (see Figure 30: MCA roadmap, page 188).

According to Belton and Stewart [345], problem definition has the objective to surface and capture the complexity which undoubtedly exists in all decision making contexts. In the consecutive phase of developing the multi-criteria model, the *essence* of a decision problem is extracted in a way which supports an evaluation of different courses of action. The overall MCA process is described as "through complexity to simplicity". MCA models therefore often appear simple, and have been criticized as simplistic. However, the apparently simple model does not deny the complexity, but has emerged from it as a distillation of the key factors in a form which is transparent, easy to work with and which can generate further insights [345]. A similar process can be observed in this study, which was narrowed down step by step during the process of developing the MCA model to a point at which it has been criticized as being simplistic (page 218).

An advantage of choosing a simple MCA approach can be that more time is available for the more important aspects of decision making, which are problem definition and a sensitivity analysis of the results. The main methodological challenge is not the development of more sophisticated MCA methods, but to support problem definition [295]. Neglecting problem definition may result in mathematically sophisticated but naive approaches [355]. In the majority of problems, scores and weights are uncertain and the different multi-criteria methods involve different assumptions [339]. Neglecting sensitivity analysis means to deliberately ignore this uncertainty and to have confidence in the results which cannot be justified. In some cases, sensitivity may also be deliberately ignored to limit the discussion on the reliability of the MCA results [295].

Three perspectives for sensitivity analysis can be distinguished [345]: A technical perspective is to identify those input parameters which have a critical influence on the overall ranking, that is, can a small change affect the overall preference order? An individual perspective can be to test the understanding of the problem against a sounding board, i.e., does one feel comfortable about the results and if not, why? The group perspective explores the ranking with different sets of criteria weights, i.e., how does an industry representative or environmentalist look at the problem? In this study, a sensitivity analysis of scores and weights was conducted and the sensitivity analysis was driven by all three perspectives. However, it was not investigated how the ranking might change if a different multi-criteria method was used.

This leads to a third challenge of MCA, which are the different MCA models, their underlying theories, and how they are implemented in practice. The choice of a model is hardly an issue for the average MCA user, despite the intensive debate in the scientific literature on the best method [295]. A similar effect was experienced in this study: the literature review of different models and theories was so overwhelming that it was impossible to discern the supposedly best MCA model for this case study. In the end, the methodology was kept deliberately simple and involved linear interval standardization, the expected value method, and weighted summation.

The single steps were implemented by the software DEFINITE and in parallel by a spreadsheet approach. Although good software support and documentation was available, not all steps were sufficiently clear at first and therefore required further investigation and direct 'hands on' experience in order to gain a better understanding of the methodology and the results. With a more sophisticated MCA approach, the feeling of a 'black box' would surely have prevailed, as implementation in a spreadsheet may not have been possible. Furthermore, a more sophisticated model does not necessarily produce better but possibly different results, and deprives the average decision maker of the possibility to understand what is going on and thereby gain confidence in the results.

MCA as decision support in EIAs

EIAs for desalination plants are complex multi-stage, multi-participatory and multi-disciplinary processes, in which a limited number of site and process alternatives have to be simultaneously evaluated with regard to a large number of potential impacts, despite often limited or uncertain information. MCA, in theory, can support the EIA in different stages by providing structure to a complex decision context, by capturing the preferences of different participatory groups, by aggregating the performance of different alternatives with regard to a large number of quantitative and qualitative criteria, and by allowing for a systematic analysis of data uncertainty (Table 41). This section will discuss some of the considerations for implementing MCA in EIAs for desalination projects.

The first stage in an EIA, in which MCA can be used as a decision support tool, is during *scoping*. In scoping, the number of possible site and process alternatives is narrowed down to a few preferred alternatives which are then investigated and evaluated in full detail in the main EIA stage. As one moves from scoping to the *main EIA stage*, the number of alternatives decreases, but the number of criteria will probably increase as the studies become more detailed. The main EIA stage may involve more than one design round, i.e., revising, adding or eliminating alternatives in the process. The EIA ends when a preferred project option is identified by the project proponents and consultants. At this stage, the EIA is submitted for *decision making*, and the competent authority will have to evaluate the overall benefits and impacts of the main alternatives, based on the EIA results, as well as stakeholder interests. Decision making usually involves bargaining and trade-offs in order to select one single alternative [295].

For a desalination plant, it would be typical to select one broader project area, possibly with different options for the intake and outfall, and to choose a main desalination technology with different options for pretreatment, process and outfall design at the end of scoping. The choice is usually limited to one project site because it is often simply too expensive to carry out a full-fledged environmental monitoring programme and EIA for two or more sites. However, it is possible to eliminate, refine or add alternatives at each stage of the EIA, which also applies to the project area if it proves unsuitable, after all.

Eliminating an alternative requires that a screening step is included in each stage to evaluate the alternatives against 'non-compensatory' or 'non-negotiable' criteria. This could be the overall cost of a project or an emission limit value which may not be exceeded, or other normative values which describe certain rules of conduct. For example, a normative approach would be to include a criterion that will eliminate alternatives that will have significant impacts on certain protected species or habitats.

Given the high level of detail in most EIAs, structuring the decision problem in an MCA becomes even more important. In chapter 5, 150 environmental concerns of desalination projects were identified, without even counting social, technical or economic considerations. For comparison, MCAs in EIAs typically have between six and twenty criteria, although complex projects may have up to a hundred criteria [295]. Scholles [351] even recommends to keep the number of criteria below ten.

A high number of criteria will result in an overly complex MCA, which may be both impractical and of limited usefulness. As the number of criteria increases, intransparency also increases. It will be more difficult to keep track of how the scores and weights of each single criterion influence the overall result, or how changes influence the ranking. Moreover, differences in the total scores of the alternatives are leveled out, as poor performance in one criterion is more likely to be compensated by good performance in another criterion. The gain of insight will therefore be limited, and it will be more difficult to discern a preferred course of action – the opposite of what the MCA actually tries to achieve [351]. Belton and Stewart [345] also emphasize that an MCA should reflect the essence of a decision problem, that is, the complexity of the decision problem has to be reduced to the *essential* aspects for decision making. The number of criteria should consequently be kept as low as is consistent with making a well-founded decision [341, 342].

At the end of the EIA process and before decision making, those criteria (or effects) which are indispensable for decision making therefore need to be identified within each category, preferably with the help of experts, i.e., environmental scientists should identify the most important environmental criteria, industry representatives the most important operational aspects, and so on. Moreover, as public participation is an elemental part of EIAs, the value judgments of the public should be considered when making a selection (e.g., as elicited during scoping consultations and meetings).

In chapter 5, about 20 out of the 150 concerns were selected as 'high priority' for EIAs. The assessment followed a methodological evaluation which is not unlike MCA. The different environmental *effects* were compared and ranked in terms of *importance* for EIA, instead of ranking *alternatives* in terms of *performance*. Four criteria (intensity, duration and spatial extend of a potential impact) were used to evaluate the effects. Their scores were measured on ordinal scales (high, medium, low) and were formally aggregated into one overall rating by a pre-defined aggregation logic. These 'high priority' environmental effects could then be re-structured and used to evaluate the performance of different project alternatives in a real application using MCA.

To conclude, an EIA for a desalination project may have three basic design and decision making rounds in the simplest case, in which MCA may prove useful:

- during scoping, when the number of alternatives is typically narrowed down to one broader project area and one main desalination technology with different options for intake, pretreatment, process, and outfall design,
- at the end of the EIA when the main findings are typically aggregated to support a recommendation for a preferred project design,
- in decision making when the competent authority evaluates and integrates the overall benefits and impacts of the main alternatives, based on the EIA findings, as well as stakeholder interests.

A two tier approach is recommended for aggregating the results of the main EIA stage. First, the most important criteria should be identified within each impact category, following an approach similar to the one outlined in chapter 5 of this thesis, and based on expert judgment and public submissions during scoping, followed by an MCA of the main alternatives and criteria. The role of MCA may not be to single out the correct or best decision but to dynamically evaluate a set of alternatives in order to gain information about the effects of different courses of action [332]. No MCA technique can eliminate the need to rely heavily on sound knowledge, data, and judgments, or the need for a critical appraisal of the results [334]. The final selection of an alternative should therefore be supported by a weight of evidence discussions and qualitative considerations.

Part III

General conclusions and recommendations

Summary conclusions and recommendations | 10

The overall objective of this thesis was to elaborate and validate a systematic environmental impact assessment and decision making framework for seawater desalination projects. Special attention was given to the process of SWRO. Although thermal distillation plants still account for 61% of the global seawater desalination capacity, they are mainly restricted to the sea areas of the Arabian Gulf and the Red Sea, whereas SWRO is the predominant process in most countries outside the Middle East. The development potential for SWRO seems boundless in the face of an ongoing urbanization and industrialization of many coastal areas. In the Arabian Gulf and Red Sea areas, where desalination is already a major source of water supply, distillation plants have also been classified as a main source of land-based pollution. The question is therefore if SWRO can pose a more sustainable solution to water shortages in coastal urban areas.

A recent review in *Nature* described desalination as a water treatment technology that is often "chemically, energetically and operationally intensive, focused on large systems", and thus requires "considerable infusion of capital, engineering expertise and infrastructure". The costs as well as the environmental concerns are still an impediment to the widespread use of desalination technologies today [285]. This indicates that desalination is a resource-intensive industrial process with significant environmental impacts. At the same time, some SWRO projects in California and Australia made headlines claiming that desalination is a "green" technology and that project developers are working towards "sustainability" [61, 68, 69, 264]. The seemingly contradictory statements are indicative of the current debate on the extent in which desalination plants will actually affect the environment. As desalination capacities and in particular SWRO capacities are expected to grow rapidly in the future, a critical examination and appraisal of the resource consumption and environmental impacts of desalination technologies was carried out in this thesis, followed by the identification and development of measures to increase the sustainability of desalination projects if necessary.

10.1 Environmental impacts and measures for impact mitigation

In this regard, the first objective of this PhD study was to conduct a systematic analysis and evaluation of potentially significant impacts of desalination projects.

The main environmental concerns of new projects usually revolve around a few key issues (Figure 41). Specific to all desalination processes is the discharge of a concentrate, which may contain residual chemicals from pretreatment and cleaning solutions, and which may impair coastal water quality and ecosystems. When evaluating these impacts, one has to distinguish between the salt, which is a natural component, and the chemicals and their reaction products. Correspondingly, different standards should apply for the disposal of the concentrate and disposal of pretreatment and cleaning wastes.

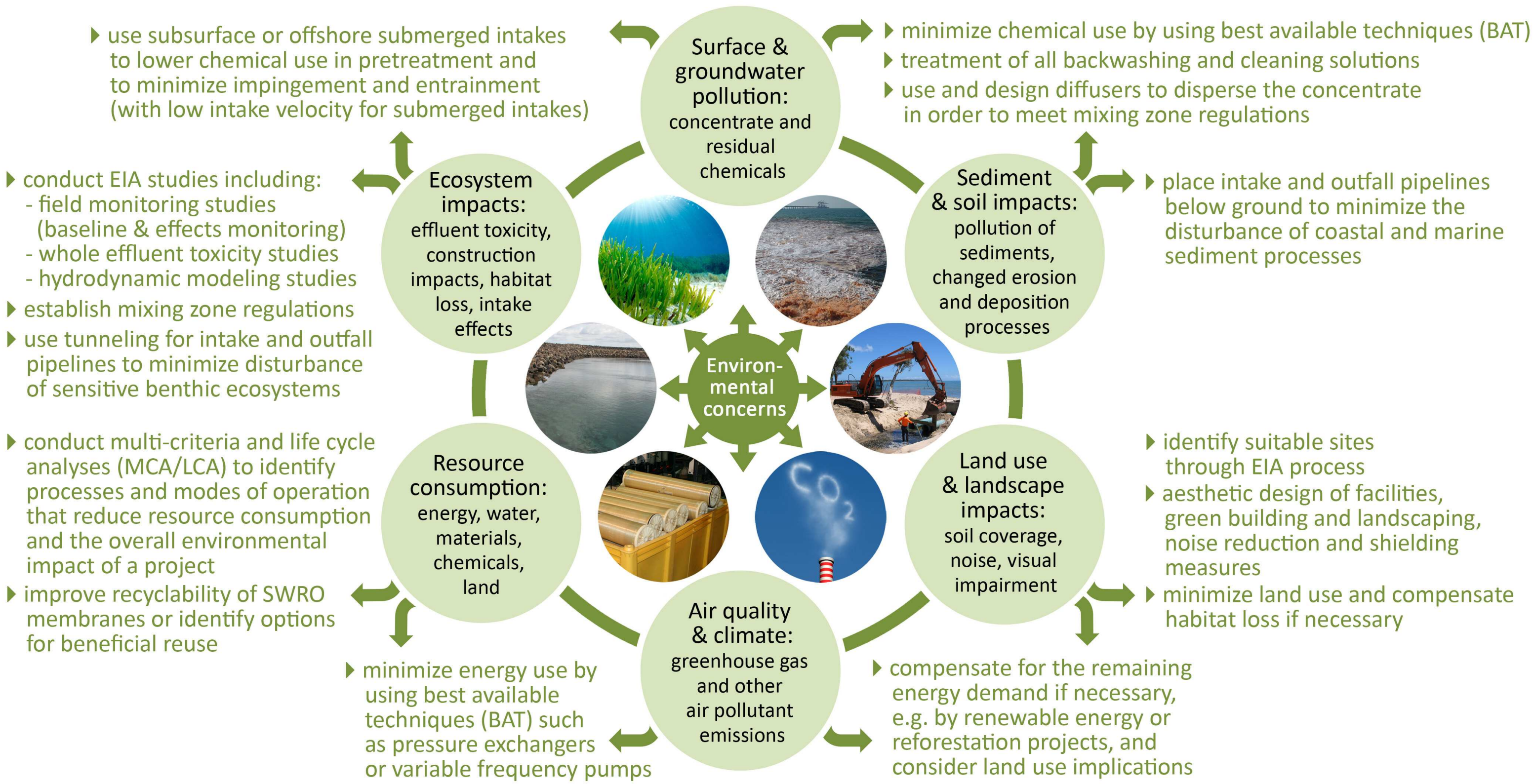

Figure 41: Key environmental concerns of seawater desalination projects and measures for impact mitigation.

Concentrate disposal
A commonly raised concern is that concentrate disposal may increase the overall salinity of an enclosed sea region in the long term. Considering the Arabian Gulf as an example with the world's highest density of desalination plants in a semi-enclosed body of water, the probability of such an effect seems to be rather low for two reasons. First, desalination only accounts for a fraction of the water losses in the Gulf as compared to natural evaporation. Second, evaporation is the driving force behind water mass circulations in the Gulf and the formation of a deep water current which exports higher salinity water from the Gulf and results in a net inflow of lower salinity water from the open ocean through the Strait of Hormuz. As a result, the seawater in the Gulf is assumed to be completely exchanged every 3 to 5 years, which should effectively counteract any long-term increase in salinity. In the absence of field monitoring data, however, any statement on the long-term impacts of desalination plants on the Gulf's marine ecosystems, particularly with regard to the chemical residuals, remains speculative.

What has been ascertained through several toxicity studies, however, is that locally elevated salinity levels caused by desalination plant discharges can be potentially harmful to marine organisms depending on the species' sensitivity and life cycle stage. The concentration of salts and the exposure time to the discharge is the problem rather than the salt itself. Discharge into the sea is therefore an adequate means of concentrate disposal if dilution to ambient levels is achieved within a very short distance from the outfall and if sensitive ecosystems are not impacted by the dispersing plume. An adequate approach to minimize the impacts of the concentrate is to establish a restricted mixing zone around the outfall in which dilution to a level close to ambient is achieved by a combination of technical means (multi-port diffusers) and natural processes (strong currents).

Mixing zone regulations combine parameters that define the spatial extent of the allowable mixing zone with water quality standards that apply at the edge of this mixing zone. As it is generally difficult to develop a universal set of mixing zone regulations and standards that apply equally to the wide range of marine ecosystems, a common approach is to tailor regulations and standards for local conditions. Moreover, single standards for each physical and chemical stressor (e.g., salinity and residual chlorine) do not take potentially synergetic effects into account. The basic concept of the tailor-made approach is to derive a single threshold or trigger value from a suite of bioassays that use the whole effluent of a given discharger to measure the acute and chronic toxicity to different local marine species representing different taxonomic and trophic levels.

In a first step, the NOEC[a] or EC_{10}[b] is established for each species. From this data set, a threshold value is calculated that indicates the safe dilution for a given percentage of species. A water quality standard would combine the dilution ratio, e.g., 1:50 corresponding to 2% return water or a salinity of 40, with the species protection level, e.g., a level which protects 95% of the marine species from experiencing a sub-chronic effect of greater than 10%. Depending on the ecosystem, a higher or lower species protection level may apply. In Australia, for example, a level of 95% applies to slightly to moderately disturbed ecosystems and 99% to ecosystems with a high conservation value. The return water dilution usually has to be met at the edge of the mixing zone in a given percentage of time (e.g., 99%) to avert negative effects. This is because it is assumed that marine organisms can temporarily cope with increased salinity levels. This approach reflects that toxicity is a function of species sensitivity, concentration and exposure time.

[a] no observed effect concentration
[b] statistically calculated concentration that causes a 10% effect

To conclude, narrow mixing zone regulations should be established site-specifically which specify (i) the spatial extent of the mixing zone depending on the local conditions (open coast, estuary, bay, etc.) and the marine ecosystems in the vicinity of the outfall, as well as (ii) the dilution rate that must be met at the edge of this mixing zone in a given percentage of time. The safe dilution ratio which protects a given percentage of the local species can be calculated from a set of WET tests. The best location and design of the outfall to meet mixing zone regulations should be determined through field monitoring in combination with hydrodynamic modeling studies that investigate the mixing conditions under different ambient, including quiescent 'worst case' conditions.

Pretreatment and cleaning chemicals

Contrary to the concentrate, impact mitigation for pretreatment and cleaning chemicals should emphasize (i) avoidance and minimization of chemical use by using best available techniques (BAT), (ii) substitution of harmful chemicals by less harmful compounds, for example by using chlorine dioxide instead of chlorine, or acid instead of polymer antiscalants, and (iii) treatment of effluents where possible. The latter explicitly applies to all side-streams from filter backwashing, UF or SWRO cleaning operations. Untreated discharge of these wastes into the sea is not considered BAT.

The most preferred pretreatment for a SWRO plant from an environmental perspective is a subsurface intake. A subsurface intake completely avoids impingement and entrainment of marine organisms and, as a biofiltration process, can potentially provide a consistently high feed water quality with advantages for pretreatment, cleaning and membrane life, hence considerably reducing chemicals, materials and energy use over the life-time of a project. Moreover, land use and landscape impacts of structures embedded in the offshore or beach sediments are lower than for plants with an open intake and a conventional pretreatment. A constraint of vertical beachwells is the limited production capacity, so that their use is restricted to smaller SWRO plants. Horizontal drains in the offshore sediments may be a promising alternative to beachwells, also for large SWRO plants, however, experience and performance data of horizontal drains is still very limited. A constraint of all subsurface intakes is the possible appearance of iron (II) and manganese (II) in anoxic/anaerobic well water, which may precipitate when oxidized, and which may necessitate additional media filtration after the subsurface intake.

An acceptable alternative where a subsurface intake is not possible due to geological or environmental constraints is a submerged intake in deep water in an offshore location. It should have a large surface area resulting in low flow velocities (passive screen), velocity caps, and fine-mesh screens which can be backwashed with air. A suitable alternative to conventional pretreatment, which is often needed for surface water and even submerged intakes, may be UF with a low chemical approach.

Energy use

A second key issue in the permitting process of new desalination projects besides concentrate disposal is energy use. Modern SWRO plants can achieve a *specific* energy demand <2.5 and a total energy demand <3.5 kWh/m^3 under favorable conditions (i.e., a salinity <35, a temperature $>15°$C, a low fouling potential) and by using state of the art equipment (i.e., pressure exchangers, variable frequency pumps and low-pressure membranes). The total energy demand is generally between 4 and 5 kWh/m^3 if one includes the pretreatment and other auxiliary equipment in the plant, the transfer of the water, and the materials and construction stages.

Carbon dioxide emissions can be estimated with a high degree of certainty as they mainly depend on the carbon content of the fuel and the energy mix of the electricity grid. For example, the global warming potential of the Australian SWRO projects (Perth, Melbourne, Sydney, Gold Coast) ranges between 2.3 and 7.8 kg CO_2-equivalents[c] per cubic meter of desalinated water. The lower value is for the lowest recorded specific energy demand, whereas the upper value reflects the full life cycle cost of desalinated water including electricity for desalination and water transfer, emissions attributed to the transportation of workforce, offsite waste decomposition, chemical and material transportation and use during construction and operation of the project.

On the one hand, SWRO plants require much less energy per cubic meter product water than thermal distillation plants if one takes the thermal energy requirements of distillation plants into account. On the other hand, SWRO is much more energy-intensive than conventional treatment of local ground and surface waters with an energy demand of 0.2-0.4 kWh/m^3. In locations where the water has to be transported over long distances, the relative energy increase caused by a seawater desalination plant may be marginal or desalination may also be the more energy-efficient option, as in the Perth metropolitan area. Compared to other activities and amenities of modern lifestyles, such as air conditioning or heating, desalinated water is not an overly energy-expensive product.

In the end, it depends on the perception and definition of significance and on local circumstances whether or not energy use of desalination is considered as a *significant* factor. In many countries, policy initiatives and stricter technological standards are being introduced in order to reduce energy consumption and increase energy-use efficiency in all sectors of use. The increasing use of energy-intensive desalination technologies counteracts these efforts to some extent. Technological standards for desalination processes are missing and should be introduced in the form of a catalogue of best available techniques in order to facilitate the selection of resource-saving technologies at the project level. Furthermore, a decision may be taken to compensate for the remaining energy use of a project. For example, carbon dioxide emissions resulting from energy use of all major Australian SWRO projects are being compensated by renewable energy projects.

Other environmental impacts

The remaining environmental concerns of desalination projects (Figure 41) show certain parallels to other coastal development projects. They include:

- impingement and entrainment effects, which also occur at coastal power plants and which can be avoided by using subsurface intakes, or minimized through proper facility design (using a combination of screens and velocity caps), siting (in offshore submerged locations), and operation (using low intake velocities of ≤0.15 m/s),
- impacts on sediment processes, which may be similar to the construction of harbors and jetties, and which can be minimized by using tunneling techniques in sensitive coastal areas and by placing pipelines below ground,
- construction, land use and landscape impacts, which are intrinsic to any development project, and which can be minimized by identifying suitable sites with preference on existing industrial sites, facility design and landscaping measures, and
- resource consumption including material, chemical, water and energy use, which can be minimized by selecting best available techniques (BAT).

[c] CO_2-e estimate the global warming potential of all climate change gases by transforming the non-CO_2 emissions into an equivalent amount of CO_2 emissions that would have the same global warming potential.

Although SWRO plants consist of numerous membrane elements which have to be replaced every few years, material use only plays a minor role in the overall resource consumption of desalination projects. A life cycle assessment found that material use and disposal has little weight (10%) compared to plant operation (90%) due to the high energy demand of all desalination processes [147–149]. Assessments of the Sydney and Melbourne desalination projects arrived at similar conclusions, that is, the total project emissions of CO_2 and other greenhouse gases associated with the materials and construction stages were $\leq$ 5% whereas operation accounted for 95% of all emissions.

Impact mitigation measures

Impact mitigation measures are usually required for all impacts which are found to be significant for a given project. The significance of impacts depends on the size, process and location of a project and on the environmental characteristics of the project site. Moreover, a universally valid standard for significance does not exist. What is considered as significant depends on a society's values, on environmental regulations, on economic potentials, and the availability of water supply alternatives. For these reasons, a final evaluation of the environmental impacts of desalination and the identification of the best practicable environmental option is only possible at a project- and site-specific level through an environmental impact assessment (EIA, see section 10.2 on page 233).

A catalogue of best available techniques (BAT) should furthermore be established to guide practitioners, consultants and decision makers in selecting adequate techniques and modes of operation to minimize impacts on the environment. According to the general concept and definition of BAT, it is proposed to consider the following order of measures when determining individual BAT solutions for desalination projects:

- selection of the desalination process with the highest energy use efficiency,
- optimization of energy and water use efficiency of that process,
- lowering the chemical use of that process by
  - reducing the occurrence of fouling and corrosion through process design (i.e., intake design and location) and thus minimizing cleaning and pretreatment requirements,
  - giving preference to no or low chemical respectively no or low waste designs,
  - substitution of harmful substances with less harmful substances,
  - optimizing the application and dosage of pretreatment chemicals based on pilot testing and/or monitoring of the feedwater quality,
  - treatment of wastes before discharge / disposal,
- selection of manufacturing materials that can be reused or recycled, and identification of appropriate waste disposal options at the end of their useful life,

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If it would be possible to choose freely between the different process alternatives, leaving out technical, economical and site-specific environmental limitations and taking only environmental benefits into account, the most preferred design would be a SWRO plant with a subsurface intake and enhanced multi-port diffuser design at a suitable oceanic site. A subsurface intake completely avoids impingement and entrainment of marine organisms and, as a biofiltration process, has the potential to provide a consistently high feed water quality with manifold advantages for process operations. It can considerably reduce chemical use for pretreatment and cleaning, and energy use by simplifying the pretreatment and by reducing the fouling potential on the SWRO membranes. It avoids landscape and land use impacts, and may increase the membrane life-time, which reduces material and energy use in the manufacturing process.

10.2 EIA and DSS

The second main objective of this PhD thesis was to develop an EIA approach applicable to large SWRO projects and including an environmental monitoring framework (monitoring and assessment protocols) and a decision support system (DSS) for the appraisal of SWRO facilities based on multi-criteria analysis (MCA).

EIA approach
The primary goals of an EIA are to provide information on the environmental consequences of a project for decision making, and to promote environmentally sound and sustainable development through the identification of appropriate alternatives and mitigation measures. EIAs are usually not limited to environmental aspects, but where appropriate also address public health and socio-economic concerns. Public participation is therefore an integral element of EIAs in many legislative systems. As a result, EIAs are often multi-stage, multi-disciplinary, and multi-participatory processes.

A ten step EIA process for desalination projects has been proposed in this thesis. The pre- or initial EIA phase includes the steps of screening and scoping of the project, in which a decision is taken on whether or not an EIA is required for a particular project, and in which the scope, contents and methodologies of the EIA and expert studies are specified in the terms of reference (ToR). A reference list for preparing the ToR has been provided as part of this thesis, which may also serve as a blueprint for preparing the EIA report. During the main EIA phase, a detailed description of the preferred project configuration including site and process alternatives is provided, and other statutory permits applicable to the project are identified and obtained. The scientific studies and analyses are conducted during this phase including baseline studies, the prediction and evaluation of impacts, and the identification of alternatives and impact mitigation measures. The final EIA phase involves decision making and a review of the EIA process. An environmental management plan is often established at this stage, which specifies the monitoring requirements during the installation and operation of the plant.

In principle, EIAs for large desalination projects will not differ in terms of complexity and level of detail from those of other water supply infrastructure projects. Depending on the proposed project, it is generally the responsibility of the competent authorities to individually define the need, scope and complexity requirements of each EIA study. When dealing with a larger number of desalination proposals, a collaborative effort between the main government agencies and participatory groups should be initiated to elaborate a national EIA guideline for desalination projects. It would facilitate the EIA process by establishing equal standards for the environmental studies to be undertaken and the information to be submitted as part of the EIA for each individual project in the future. Moreover, as a number of agencies usually have permitting authority over the project, a lead agency should be nominated to coordinate the process by involving other agencies and by informing the project proponent about permitting requirements.

To this day, only a limited number of EIA studies for desalination projects have been carried out and published. Most of them are from Australia and the United States. In the

EU, the EIA Directive[d] regulates which project categories have to be made subject to an EIA by member states. It lists major water supply projects such as groundwater abstraction schemes, dams, and works for the transfer of water resources between river basins. The list should be expanded to include desalination projects. EIAs for desalination plants from other parts of the world, particularly the Middle East, are scarcely available. The reason may be that EIA studies are considered to be the intellectual property of the project proponent. This is contradictory to the notions of transparency and public participation, and EIAs should generally be made available to a wider public audience.

Environmental monitoring
As EIA studies make predictions about the expected impacts of a desalination project on the environment, it is necessary to validate the accuracy of these predictions against observations during project implementation and operation. The longest effects monitoring programmes for a few SWRO projects worldwide have just accumulated two to three years of cohesive monitoring data. Although this may allow for some conclusions regarding those particular projects, it is too early to use these results as conclusive evidence concerning the long-term impacts of desalination plants. A review of existing monitoring studies revealed that most other studies were either limited in scope – addressing only one aspect such as salinity, short-term – without a continuous baseline and effects monitoring, and localized – without adequate spatial replication.

The core of the problem is to design a monitoring programme that provides sufficient statistical power to be able to distinguish the effects of the desalination project from natural processes. For this reason, sufficient spatial and temporal replication is needed in field monitoring studies. Field monitoring for desalination projects should include the project (impact) site and several control sites which adequately represent the habitats of the impact site. To capture the temporal variance at these sites, paired sampling should be carried out at several times before, during and after project implementation (BACIPS approach). Baseline monitoring for major development projects is usually carried out over a period of two years before project start-up and effects monitoring for two to three years during commissioning and operation. A holistic monitoring framework for desalination projects should furthermore integrate field monitoring with hydrodynamic modeling and bioassay studies, preferably using whole effluent toxicity tests. These concepts have been implemented in the monitoring programmes for the Sydney and Gold Coast projects, which will hopefully provide useful experimental results in the near future.

Decision support system
EIAs typically generate large volumes of complex information which often exceed the capacities of decision makers to integrate and process this information. Moreover, different decision makers and stakeholders often have conflicting preferences about a project, so that the process of decision making in an EIA can be described as a conflict analysis between different value judgments. The process can be facilitated by a formalized decision support tool, such as multi-criteria analysis (MCA), which allows the integration of multiple quantitative and qualitative criteria and different value judgments. As EIAs for large desalination projects do not differ in terms of complexity from those for other infrastructure projects, one objective of this thesis was to explore the usefulness of MCA for evaluating desalination projects and to apply MCA to a case study.

[d] Directive 85/337/EEC on the assessment of the effects of certain public and private projects on the environment, amended by Directive 97/11/EC.

Case study
The case study compared different intake and pretreatment systems of SWRO plants under environmental, operational and economic criteria. Structuring the decision making problem and generating the input data was clearly the most time-intensive and crucial part of the study. The set of alternatives and criteria was revised several times and finally narrowed down to the most essential aspects for decision making. A few of the initial alternatives and criteria were also found to be not operational due to a lack of useful data, which is attributed to the hypothetical nature of the study which mostly used literature sources. Different value judgments on the importance of the single criteria for decision making were gathered by means of a questionnaire. Emphasis was placed on the sensitivity analysis to investigate the effects of uncertain data on the ranking.

The ranking results and sensitivity analysis support the conclusion that subsurface intakes are generally the best option for SWRO plants, followed by conventional pretreatment with or without treatment of backwashing and cleaning solutions, followed by ultrafiltration. The MCA showed that the overall ranking order was generally similar in all four expert groups, although plant operators and companies gave highest priority to the criteria water quality and costs, and the researchers and environmental groups gave highest priority to the ecological impact. While the outcome with regard to beachwells as the best option was to be expected based on the gathered input data (beachwells were dominant), the inferior performance of ultrafiltration was unexpected.

UF pretreatment is still a relatively new technology. The potential for future improvements is therefore high, and lessons should also be learnt from past shortcomings of conventional pretreatment systems. Chemical use needs to be minimized and that backwash waters and spent cleaning solutions should be collected and treated rather than discharged. To increase the performance and acceptance of UF, most membrane suppliers and researchers have focused on operational and economic aspects in the past. It is clear that UF will not achieve a real breakthrough over conventional pretreatment if it does not perform equally well or better in terms of water quality. Based on SDI$\leq$3, there is no real difference between conventional and UF pretreatment, however UF pretreatment, especially with in-line coagulation, can achieve a potentially lower organic fouling on the SWRO membranes. In case of similar performance, environmentally-friendly designs should be considered as an advantage which may tip the balance in favor of UF even if costs of UF pretreatment remain high. This may be a particular advantage in emerging markets such as Australia, where UF pretreatment has recently been selected for two large SWRO projects.

MCA as decision support tool for EIAs of desalination projects
MCA has already been successfully used in evaluating alternative sites for desalination plants [64, 293, 294]. This thesis showed that MCA can also be used to compare process alternatives, despite the limitations of a hypothetical study. As it is anticipated that MCA will prove more powerful in real applications, it is concluded that MCA is a suitable tool for assessing both the site and process alternatives of desalination plants.

Although the method of life cycle analysis (LCA) is more common for comparing process alternatives, a disadvantage of LCA is that it is mostly limited to quantitative aspects of resource consumption. LCAs have also been conducted to evaluate and compare the main desalination processes [126, 147–149, 152, 354] and the different pretreatment options. The studies analyze the main material and energy streams, but fall short of integrating ecological or other impacts which are difficult to quantify but are nevertheless important considerations in the contexts of EIAs. A main benefit of MCA is that qualitative and non-commensurable criteria can be integrated into the analysis. Another benefit of MCA is that it emphasizes structuring of the decision problem, that is, one has to be clear about the objectives, the relevant alternatives and evaluation criteria for decision making. Moreover, it requires that a deeper understanding of the different alternatives is developed. Both facilitate the decision maker's learning about the problem.

EIAs for desalination plants are multi-stage, multi-disciplinary and multi-participatory processes in which a few site and process alternatives have to be simultaneously evaluated with regard to many potential impacts, despite often limited or uncertain information. MCA can provide structure to the decision context by capturing the preferences of different participatory groups, by aggregating the performance of different alternatives with regard to a large number of quantitative and qualitative criteria, and by carrying out a systematic sensitivity analysis with regard to data uncertainty. In the simplest case, an EIA for a desalination project may have three basic design and decision making rounds, in which MCA may prove useful:

- during scoping, when the number of alternatives is typically narrowed down to one broader project area and one main desalination technology with different options for intake, pretreatment, process, and outfall design,
- at the end of the EIA when the main results are typically aggregated to support a recommendation for a preferred project configuration,
- in decision making when the competent authority evaluates and integrates the overall benefits and impacts of the main alternatives as well as stakeholder interests.

An important consideration for successfully using MCA in EIA for desalination projects is to reduce the complexity of the decision making problem to those criteria which are essential to decision making. In this thesis, 150 environmental concerns of desalination projects were identified, twenty of which can be generally considered as significant. In real world EIAs, the step of scoping has the objective to generate a complete list of issues, based on scoping consultations with experts, stakeholders and the community. The list of issues should then be evaluated in terms of relative importance and significance, and can form the basis of the MCA.

A second consideration for using MCA in EIA is to incorporate a screening mechanism by which alternatives are eliminated which do not comply with certain 'non-compensatory' or 'non-negotiable' criteria, such as the overall cost of a project or an emission limit value which may not be exceeded. A third consideration is that a critical appraisal of the MCA is necessary. The role of MCA is not to formally single out the correct decision but to dynamically evaluate a set of alternatives in order to gain information about the effects of different courses of action [332]. The final selection of an alternative should therefore rely on sound judgment supported by a weight of evidence discussion.

10.3 Final remarks

As stated in the introduction of these conclusions, desalination has been described as a water treatment process that is often "chemically, energetically and operationally intensive, focused on large systems", so that costs as well as environmental concerns are still an impediment to the widespread use of desalination technologies today [285].

The logical solution to this problem must be the development of green, sustainable technologies. 'Green technology' means the application of environmental science to conserve the natural environment and resources and to curb the negative impacts of human involvement [280]. Consequently, 'green desalination' should include EIA and environmental monitoring studies to investigate and minimize impacts on the natural environment, and the implementation of BAT standards to curb the use of natural resources. EIA and BAT complement each other: BAT is a technology-based approach, while EIA aims at minimizing impacts at a site- and project-specific level.

Public acceptance is a key project objective of new desalination developments. Main concerns raised during public consultation processes are often the impacts on the marine environment and energy consumption. A green image campaign to increase public acceptance is only credible if it is well-grounded and in fact more than a lip service. Some project developers have recognized this and started to work towards green, sustainable desalination. Sustainable desalination does not seem to be some distant utopia but can be achieved with existing technologies. In particular the Australian projects, including Sydney, Perth or the Gold Coast project, set a good example for incorporating environmental protection measures that will hopefully encourage others to follow in their footsteps. The industry, regulators and communities alike, however, have to pave the way by making a commitment to green, sustainable desalination projects which really live up to the expectations. It also requires a commitment to providing water at a price which does not only include the usual construction and operating costs, but also the costs that are necessary to reduce the environmental footprint through environmental studies, advanced technology, or compensation measures. Environmental protection measures will most certainly increase the cost of water production, but, as the Australian project show, sustainable solutions are economically viable.

The sustainability of desalination is nevertheless still questioned by environmentalists on the grounds of potential marine impacts and high energy use. Opposing a desalination project on these grounds tacitly assumes that (i) the existing water supply schemes or other water supply alternatives are sufficient to meet the demand and (ii) are also more sustainable than desalination. Both is not necessarily the case.

(i) As desalination is more expensive than most traditional forms of water supply, it is usually used after other options have been fully exploited, and is usually only one aspect of a whole package of water management measures including more efficient water use, fixed water quotas, water restrictions, and water reuse, as for example in Israel. In many countries, water use efficiency can still be increased. For example, the per capita water use of Perth residents is 290 liters per day for domestic purposes and 420 liters per day if one includes indirect uses[e], which is high by world standards. Nevertheless, a second large desalination plant for the Perth region is in development. Desalination projects – like other water infrastructure projects – often consume considerable community resources which are not reflected in the investment and operating costs. These may be in the form of subsidies, access to coastal land, or the provision of connecting infrastructure. The desalinated water should therefore be valued as a community asset, by non-wasteful use and by looking for opportunities of multiple use. Consequently, desalination projects (where not already the case) should be an integral part of water resources management planning that not only considers the development of new or existing water supplies, but also the economic use and reuse of water where possible.

(ii) The status quo of many existing water supply schemes is that local ground and surface water resources are being overexploited or that entire river systems are being diverted or dammed, both often with severe impacts on the aquatic ecosystems. According to the World Commission on Dams, the true profitability of large dams remains elusive,

[e] such as commerce or irrigation of parks, but not 'virtual water' embedded in food and consumer products

as a considerable portion of the world's large dams deliver less water and electricity than promised while overrunning costs and having led to an irreversible loss of species and ecosystems in many cases [274]. The Commission's report endorsed many of the environmentalists most trenchant criticisms [1]. Desalination may therefore well be the lesser of two evils with regard to ecosystem impacts if well designed and operated. However, it is without question the more energy-intensive option, but at the same time energy is wasted in other sectors of use, such as in households by old and inefficient appliances. Like water, energy is not to be wasted, and more efficient energy use in households may probably compensate for the energy needed to produce and deliver desalinated water.

In the end, decisions about desalination developments revolve around local circumstances including the need for water and available alternatives to desalination, the costs of the project and financing options, the significance of environmental impacts, and the definition of significance. Desalination will not be the solution to all of the world's water problems, but modern SWRO projects seem to be a more sustainable alternative than many existing water supply schemes and can alleviate pressures on freshwater ecosystems. However, a project has to be designed and operated according to environmental criteria in order to not spread the problem from the freshwater to the marine ecosystems. Many good concepts have been implemented in the latest desalination projects in Australia to minimize the environmental footprint of desalination. What is furthermore needed is ongoing research and demonstration projects to gain experience, knowledge and trust in new, environmentally friendly technologies, as well as political incentives through policies or financial support to implement best available technologies.

Appendices

Executive summary

The seawater desalination industry is experiencing a tremendous growth. The combined capacity of all desalination plants worldwide increased from 28 million m^3 per day (Mm^3/d) in 2007 by 30% to 36 Mm^3/d in 2009. While thermal distillation plants predominate in the Middle East, seawater reverse osmosis (SWRO) is the preferred process in most other countries. The development potential for SWRO seems boundless in the face of an ongoing urbanization and industrialization of many coastal areas. However, as the need for desalination accelerates in many parts of the world, concerns are raised over the various adverse impacts of desalination plants on the environment.

About 150 potential impacts were identified in this PhD study. A key concern that is specific to all desalination plants is the disposal of the concentrate, which may contain chemical residuals from operation and maintenance and which may be harmful to the flora and fauna in the discharge area. Other key concerns, although not exclusive to desalination, are the seawater intakes, which may cause impingement and entrainment of marine organisms, and air pollutant and greenhouse gas emissions due to energy use.

The growing number of desalination plants worldwide and their potential impacts on the environment emphasize the need for *'greener'* desalination *technologies* and more *sustainable* desalination *projects*. This can be achieved through the implementation of standards for best available techniques (BAT) and the conduct of methodologically sound environmental impact assessment (EIA) and monitoring studies. BAT aims at identifying suitable processes at the technology level, which can facilitate the identification of individual BAT solutions at a project- and site-specific level through EIA studies.

The objectives of this PhD study were therefore (i) to develop strategies and identify measures for impact mitigation by applying the concept of BAT to desalination plants, (ii) to develop guidance on EIAs for desalination projects, including specifications for laboratory, modeling and field monitoring studies, and (iii) to develop and implement a decision support system based on multi-criteria analysis (MCA) which can facilitate decision making in various stages of the EIA process. The emphasis was on SWRO and the different feedwater intake and pretreatment designs in SWRO plants.

The most preferred intake under environmental criteria is a subsurface intake. It completely avoids the impingement and entrainment of marine organisms and, as a biofilter, considerably reduces the chemicals, materials and energy use of a project over its lifetime. Constraints, however, are the possible appearance of iron (II) and manganese (II) in anoxic/anaerobic well water, the limited production capacity of beachwells, and the limited experience with horizontal offshore wells. An acceptable alternative where a subsurface intake is not possible is a submerged offshore intake with low flow velocities and velocity caps. A suitable alternative to conventional pretreatment for surface/submerged intakes may be UF pretreatment with a low chemical approach, although UF pretreatment has only been selected for a limited number of smaller SWRO plants to date.

An adequate approach to mitigate the impacts of concentrate disposal is to minimize the mixing zone around the outfall by a combination of technical means (i.e., multi-port diffusers) and natural processes (i.e., strong currents). The dilution ratio that is required to protect the local fauna and flora should be determined from a set of whole effluent toxicity tests, and the best location and design of the outfall should be determined through field monitoring and hydrodynamic modeling studies. In order to mitigate the potential impacts from chemical discharges, chemical use in pretreatment and cleaning should be minimized through the use of BAT, i.e., harmful substances should be substituted and effluents from filter backwashing, UF or SWRO cleaning operations should be treated.

The total energy demand to produce 1 m^3 of water is less than 3.5 kWh in SWRO plants that use BAT. Compared to other amenities of modern lifestyles, such as air conditioning or heating, desalinated water is not an overly energy-expensive product, but it is far more energy-intensive than the use of conventional ground and surface water resources. In the end, it depends on a society's values and economic potentials whether or not energy use of desalination is perceived as a *significant* issue. Renewable energy projects can be implemented to compensate for the energy use of desalination, as it is done for some Australian projects where energy use is considered as a significant factor.

A final evaluation of the environmental impacts of desalination projects is only possible at a project- and site-specific level through an EIA process. As EIA studies make predictions about the expected impacts before a project is implemented, monitoring during construction and operation is paramount to verify the EIA results. To this day, only a limited number of EIA studies of desalination projects has become available, and the longest monitoring programmes have gathered 2-3 years of cohesive data. As there is still a surprising paucity of useful experimental data from field monitoring and laboratory studies, a monitoring framework for desalination plants was proposed in this thesis.

The primary goals of an EIA are to promote sustainable development through the identification of appropriate alternatives and mitigation measures, and to provide information on the environmental consequences of each alternative for decision making. EIAs are often complex multi-stage, multi-disciplinary, and multi-participatory undertakings. A systematic ten step EIA process for desalination projects has therefore been proposed in this thesis, ranging from project screening to decision making.

Decision making in an EIA can be facilitated by a formalized decision support tool, such as multi-criteria analysis (MCA), which allows a comparison of alternatives under various quantitative and qualitative criteria, as well as different stakeholder perspectives. The usefulness of MCA as a decision support tool for EIAs of desalination projects was explored in this thesis, and, as a practical example, MCA was used to evaluate different intake and pretreatment options for SWRO. The MCA consolidated the conclusion that a subsurface intake offers many operational and environmental benefits, which makes it the preferred intake and pretreatment option for SWRO where feasible.

In conclusion, sustainable desalination is not a utopia but technically feasible. However, it requires a commitment to providing water at a price which does not only include the usual construction and operating costs, but also the costs that are necessary to reduce the environmental footprint through environmental studies, best available technology, or compensation measures. The recent Australian SWRO projects set a good example for environmental protection measures that will hopefully encourage others to follow in their footsteps. The industry, regulators and communities alike have to pave the way by making a commitment to greener and more sustainable desalination projects.

Samenvatting

De sector voor de ontzilting van zeewater maakt een enorme groei door. De gezamenlijke capaciteit van alle ontziltingsinstallaties wereldwijd is van 28 miljoen m^3 per dag (Mm^3/d) in 2007 gestegen tot 36 Mm^3/d in 2009 – een stijging van 30%. In het Midden-Oosten wordt vooral gebruik gemaakt van installaties op basis van thermische distillatie, maar in de meeste andere landen wordt de voorkeur gegeven aan het proces van omgekeerde osmose voor het ontzilten van zeewater (seawater reverse osmosis – SWRO). Het ontwikkelingspotentieel voor SWRO lijkt oneindig, gezien de voortschrijdende urbanisatie en industrialisatie van tal van kustgebieden. Nu de behoefte aan ontzilting in grote delen van de wereld almaar toeneemt, ontstaat evenwel bezorgdheid over de uiteenlopende negatieve milieueffecten van ontziltingsinstallaties.

In dit promotieonderzoek zijn circa 150 potentiële effecten in kaart gebracht. Een belangrijk probleem dat bij alle ontziltingsinstallaties een rol speelt is de verwijdering van het concentraat, dat chemische residuen kan bevatten die verband houden met de werking en het onderhoud van de installatie en dat schadelijk kan zijn voor de flora en fauna in het lozingsgebied. Andere belangrijke problemen – die echter niet alleen bij ontzilting spelen – betreffen de winning van het zeewater, waardoor mariene organismen beïnvloed en meegevoerd kunnen worden, en de uitstoot van vervuilende stoffen en broeikasgassen als gevolg van het verbruik van energie.

Door de toename van het aantal ontziltingsinstallaties wereldwijd en de potentiële milieueffecten daarvan klinkt de roep om *'groenere' technologieën* en *duurzamere projecten* op het gebied van ontzilting steeds luider. Een en ander kan gerealiseerd worden door normen voor best beschikbare technieken (BBT's) te implementeren en methodologisch verantwoorde milieueffectbeoordelingen (MEB's) en monitoringstudies uit te voeren. Toepassing van BBT's heeft tot doel op technologisch niveau passende processen te identificeren waarmee het eenvoudiger wordt op projectniveau en voor iedere afzonderlijke locatie via MEB's specifieke BBT-oplossingen te vinden.

De doelstellingen van dit onderzoek waren derhalve de volgende: i) ontwikkeling van strategieën en identificatie van maatregelen gericht op beperking van milieueffecten via toepassing van het BBT-concept op ontziltingsinstallaties, ii) uitwerking van richtsnoeren voor MEB's inzake ontziltingsprojecten, met onder meer specificaties voor laboratoriumonderzoek, modelstudies en monitoringstudies in de praktijk en iii) ontwikkeling en implementatie van een op multicriteria-analyse (MCA) gebaseerd beslissingsondersteunend instrument waarmee de totstandkoming van besluiten in de diverse fasen van de MEB-procedure vereenvoudigd kan worden. Het accent lag op SWRO en de verschillende modellen voor winning en voorbehandeling van voedingswater zoals die in SWRO-installaties toegepast worden.

Het vanuit milieuoogpunt meest aan te bevelen procedé voor het winnen van zeewater is ondergrondse winning. Daarbij worden beïnvloeding en meevoering van mariene organismen volledig vermeden, en dankzij de biofilterwerking zijn aanmerkelijk minder chemicaliën, materialen en energie nodig gedurende de looptijd van een project. Beperkende factoren zijn evenwel de mogelijke aanwezigheid van ijzer (II) en mangaan (II) in zuurstofloos/anaëroob water in putten, de beperkte productiecapaciteit van putten op het strand en het gebrek aan ervaring met horizontale putten in zee. Een aanvaardbaar alternatief in situaties waarin ondergrondse winning niet mogelijk is, is open winning in zee bij lage stroomsnelheden en met gebruikmaking van velocity caps. Een geschikt alternatief voor conventionele voorbehandeling bij ondergrondse/open winning zou voorbehandeling op basis van ultrafiltratie (UF) met laag gebruik van chemicaliën kunnen zijn, al is deze techniek tot op heden slechts bij een beperkt aantal relatief kleine SWRO-installaties toegepast.

De milieueffecten van lozing van het concentraat kunnen effectief beperkt worden door het menggebied rond de uitlaat zo klein mogelijk te houden via een combinatie van technische middelen (multiport diffusors) en natuurlijke processen (sterke stroming). De verdunningsverhouding die nodig is om de lokale flora en fauna te beschermen moet bepaald worden op basis van een serie totaal-effluentbeoordelingen; de vraag naar de beste locatie en het beste ontwerp voor de uitlaat moet beantwoord worden op basis van monitoringstudies in de praktijk en hydrodynamische modelstudies. Om de potentiële milieueffecten van chemicaliënlozingen te beperken, moet het gebruik van chemische stoffen bij de voorbehandeling en reiniging teruggedrongen worden door toepassing van BBT's. Dat houdt in dat schadelijke stoffen vervangen moeten worden en dat de effluenten afkomstig van reinigingsprocessen in verband met filterspoeling, ultrafiltratie of SWRO behandeld dienen te worden.

De totale hoeveelheid energie die nodig is om 1 m^3 water te produceren is minder dan 3,5 kWh bij SWRO-installaties waar BBT's worden toegepast. Vergeleken met andere hedendaagse voorzieningen, zoals airconditioning of verwarming, is ontzilt water geen uitgesproken energieverslindend product, maar de productie ervan vergt wel aanzienlijk meer energie dan wanneer gebruik wordt gemaakt van conventionele voorraden grond- en oppervlaktewater. In laatste instantie zijn het de waarden en het economisch potentieel van een samenleving die bepalen of het energieverbruik in verband met ontzilting al dan niet als *significant* probleem beschouwd wordt. Ter compensatie van het energieverbruik in verband met ontzilting kan gekozen worden voor op hernieuwbare energie gerichte projecten, zoals hier en daar in Australië gebeurt, waar dit energieverbruik significant geacht wordt.

Een definitieve beoordeling van de milieueffecten van ontziltingsprojecten is alleen mogelijk wanneer dergelijke projecten afzonderlijk, rekening houdend met de specifieke locatie, via een MEB geëvalueerd worden. Aangezien MEB's bedoeld zijn om voorafgaand aan de implementatie van een project prognoses op te stellen omtrent de te verwachten effecten daarvan, is het van groot belang dat in de aanlegfase en na de inbedrijfstelling van de installatie controle plaatsvindt ter verificatie van de uitkomsten van de MEB. Tot op heden is slechts een klein aantal MEB's inzake ontziltingsprojecten uitgevoerd, en met de langstlopende monitoringprogramma's zijn over een periode van slechts twee à drie jaar coherente data verkregen. Aangezien er nog steeds sprake is van een opvallend gebrek aan bruikbare experimentele data van monitoringstudies in de praktijk en laboratoriumonderzoek wordt in dit proefschrift een monitoringkader voor ontziltingsinstallaties voorgesteld.

De primaire doelen van een MEB zijn het bevorderen van duurzame ontwikkeling door het identificeren van passende alternatieven en mitigerende maatregelen, en het verstrekken van informatie over de gevolgen voor het milieu van ieder alternatief waarvoor in het besluitvormingstraject gekozen kan worden. MEB's zijn vaak ingewikkelde procedures die bestaan uit meerdere fasen en waarbij meerdere disciplines en partijen betrokken zijn. Daarom wordt in dit proefschrift een systematisch tienstappenplan voor MEB's inzake ontziltingsprojecten voorgesteld dat het gehele proces, van projectscreening tot besluitvorming, omvat.

De besluitvorming in een MEB kan vereenvoudigd worden door gebruik te maken van een geformaliseerd beslissingsondersteunend instrument, zoals MCA, waarmee alternatieven op basis van uiteenlopende kwantitatieve en kwalitatieve criteria en vanuit verschillende, stakeholderspecifieke invalshoeken met elkaar vergeleken kunnen worden. In dit proefschrift is de toepasbaarheid van MCA als beslissingsondersteunend instrument voor MEB's inzake ontziltingsprojecten onderzocht, en bij wijze van praktisch voorbeeld is MCA gebruikt om verschillende opties voor winning en voorbehandeling in het kader van SWRO te beoordelen. De MCA bevestigde de conclusie dat ondergrondse winning vanuit operationeel en milieuoogpunt veel voordelen oplevert en derhalve daar waar deze techniek toepasbaar is beschouwd moet worden als de voorkeursoptie voor winning en voorbehandeling bij toepassing van SWRO.

Afsluitend kan gesteld worden dat duurzame ontzilting geen utopie, maar een technisch haalbare optie is. Die vereist echter wel de bereidheid water beschikbaar te stellen tegen een prijs die niet alleen de gebruikelijke kosten voor de aanleg en werking van installaties omvat, maar ook de kosten die gemaakt worden ter beperking van de ecologische voetafdruk en die voortvloeien uit de uitvoering van milieustudies, de toepassing van best beschikbare technologieën of de tenuitvoerlegging van compenserende maatregelen. De recentelijk geïmplementeerde Australische SWRO-projecten laten zien hoe milieubescherming in ontziltingsinstallaties geïntegreerd kan worden en kunnen als voorbeeld dienen voor andere projecten. De sector, regelgevende instanties en de samenleving dienen tezamen het pad te effenen door zich nadrukkelijk in te zetten voor groenere, duurzamere ontziltingsprojecten.

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List of abbreviations and units

ADC	acoustic doppler current profiler
AEB	air enhanced backwash
AHP	analytic hierarchy process
AOC	assimilable organic carbon
AS	ambient standards
BACI	before-after, control-impact
BAT	best available technique
BEP	best environmental practice
BW	backwash
BWL	beachwell
BWRO	brackish water reverse osmosis
CA	cellulose acetate
CAM	cyanoacetamide
CaOCl	calcium hypochlorite
CBA	cost-benefit analysis
CBPs	chlorination by-products
Cc	concentrate
CCC	criterion continuous concentration
CEB	chemically enhanced backwashing
CF	cartridge filters
CI	conventional pretreatment w/out sludge treatment
CII	conventional pretreatment with sludge treatment
CIP	cleaning in place
Cl_2	chlorine
ClO_2	chlorine dioxide
CMC	criterion maximum concentration
CO_2	carbon dioxide
CO_2-e	CO_2 equivalents
CP	conventional pretreatment
CTD	conductivity temperature depth profiler
CW	cooling water
CWA	United States Clean Water Act
DADMAC	diallyldimethylammonium chloride
DAF	dissolved air flotation
DBAN	dibromoacetonitrile
DBAM	dibromoacetamide
DBNPA	2,2-dibromo-3-nitrilopropionamide
DE	diatomaceous earth filter
DMF	dual media filter/filtration
DOC	dissolved organic carbon
DSS	decision support system
DWEER	dual work exchanger
EC_{10}	effect concentration, 10%
ED	electrodialysis
EDS	Enercon Desalination System
EDTA	ethylenediaminetetraacetic acid
EES	environmental effects statement
EIA	environmental impact assessment
EPA	United States Environmental Protection Agency
ES	effluent standards
FAC	free available chlorine
FB	flocculation basin
$FeCl_3$	ferric chloride
$FeSO_4$	ferrous sulphate
FL	flocculation
FLC	flocculation chamber
GCC	Gulf Cooperation Council (Bahrain, Kuwait, Oman, Qatar, Saudi Arabia and the UAE)
GF	gravity filter
GHG	greenhouse gases
GIS	geographic information system
GJ	gigajoule
GMF	granular media filter/filtration
HCl	hydrochloric acid
HDD	horizontally drilled/directed drains
HELCOM	Convention on the Protection of the Marine Environment of the Baltic Sea Area
HF	hollow fiber
$HOBr/OBr^-$	hypobromous acid/ion
$HOCl/OCl^-$	hypochlorous acid/ion
H_2SO_4	sulfuric acid
$H_2C_2O_4$	oxalic acid
Hm^3/a	hecto cubic meters per year
IMS	integrated membrane system

IPPC	European Directive 2008/1/EC concerning Integrated Pollution Prevention and Control
KCl	potassium chloride
kJ	kilojoule
kWh/m^3	kilowatt hours per cubic meter
LBS	Land-Based Sources Protocol of the Mediterranean Action Plan
LCA	life cycle analysis
LC_{50}	lethal concentration, 50%
LIW	Levantine Intermediate Water
lmh, $l/m^2/h$	liters per square meters and hour
MBNPA	monobromonitrilopropionamide
MCA	multi-criteria analysis
MED	multi-effect distillation
MED-TVC	MED with thermal vapor compression
MEDRC	Middle East Desalination Research Center
MENA	Middle East and North Africa
MF	microfiltration
MFI	modified fouling index
MJ	megajoule
Mm^3/d	million cubic meters per day
MSF	multi-stage flash distillation
MW	megawatt
MWh	megawatt hours
$NaHSO_3$	sodium bisulfite (SBS)
NaOCl	sodium hypochlorite
NaOH	sodium hydroxide
$(NaPO_3)_6$	sodium hexametaphosphate (SHMP)
NEC	National Emission Ceilings Directive 2001/81/EC of the European Parliament
NF	nanofiltration
NGO	non government organization
NOEC	no observed effect concentration
NOM	natural organic matter
NO_X	generic term for mono-nitrogen oxides (NO and NO_2) produced during combustion
NPDES	United States National Pollutant Discharge Elimination System
NRC	United States National Research Council
NSW	New South Wales
NTU	nephelometric turbidity unit
OSPAR	Convention for the Protection of the Marine Environment of the North-East Atlantic
OTC	once through cooling systems
PAA	polyacrylic acid
PAN	polyacrylonitrile
PEC	predicted environmental concentration
PES	polyethersulfone
PF	pressurized media filter
PFS	polyferric sulphate
$PM_{2.5}$	particulate matter <2.5 µm
PM_{10}	particulate matter <10 µm
PNEC	predicted no effect concentration
PPCW	power plant cooling water
ppb	parts per billion (e.g. µg/kg)
ppm	parts per million (e.g. mg/kg)
ppt	parts per thousand (e.g. g/kg)
PS	polysulfone
psu	practical salinity unit
PVDF	polyvinylidene fluoride
PX	pressure exchanger
RO	reverse osmosis
ROPME	Regional Organization for the Protection of the Marine Environment
S	salinity
S_{CC}	concentrate salinity
SBS	sodium bisulfite ($NaHSO_3$)
SDI	silt density index
SDI_{15}	15-min. SDI
SED	sedimentation
SHMP	sodium hexametaphosphate ($(NaPO_3)_6$)
SM	static mixer
SO_2	sulfur dioxide
SO_X	sulfur oxides including SO, SO_2, SO_3, S_7O_2, S_6O_2
SPL	species protecting level
SPTV	species protection trigger value
SWRO	seawater reverse osmosis
t	metric tons
TDS	total dissolved solids
THMs	trihalomethanes
TJ	terajoule
TOC	total organic carbon
ToR	terms of reference
TRC	total residual chlorine
TRO	total residual oxidant
TWh	terawatt hours
UAE	United Arab Emirates
UF	ultrafiltration
UNEP	United Nations Environment Programme
WET	whole effluent toxicity
WHO	World Health Organization
%w/w	weight of the solute relative to the weight of the final solution
ZID	zone of initial dilution
ZLD	zero liquid discharge

Curriculum vitae

Of Sabine Lattemann, born on September 17th 1975 in Salzgitter, Germany

The PhD research was carried out at UNESCO-IHE Institute for Water Education in Delft, in a sandwich construction with the Institute for Chemistry and Biology of the Marine Environment at the University of Oldenburg, Germany.

Work experience

Jan. 2007 to Dec. 2009	Scientific employee at Oldenburg University, Germany, within the EU research project “Membrane based desalination, an integrated approach”
Dec. 2008 to Feb. 2009	Review of the draft report “Environmental impact assessment of nuclear desalination” on behalf of the International Atomic Energy Agency
Sep. 2008 to Dec. 2009	Technical advisory services in the “Brine beneficial use study for the MASDAR development initiative” in Abu Dhabi
Jan. 2008 to Dec. 2008	Participation in the MEDRC-project “Environmental planning, prediction and management of brine discharges from desalination plants”
Jan. 2008 to Dec. 2008	Review of the environmental impact statement for the Olympic Dam seawater RO project on behalf of Arup Pty Ltd, Australia
Nov. 2007 to Dec 2007	Peer review of the report “Desalination – a national perspective” on behalf of the United States National Research Council of the National Academies
Jan. 2007 to May 2007	Environmental assessment of the Enercon Desalination System on behalf Enercon GmbH, Germany
since Jul. 2004, currently on leave	Scientific employee at the German Federal Environmental Agency (UBA), Department of Transportation
Jan. 2006 to June 2006	Advisor to Water Consultants Intl. within the project “Environmental literature review and position paper for Perth seawater desalination plant two and Sydney seawater reverse osmosis plant”
May 2004 to Dec. 2007	Advisor to the World Health Organization Eastern Mediterranean Regional Office within the project “Guidance on desalination for safe water supply”, chair of the working group on environmental impacts
May 2003 to Apr. 2004	Preparation of EIA studies for two offshore wind energy farms in the North Sea with a combined capacity of 460 MW (92 wind turbines) on behalf of ENOVA Group, Germany
Aug. 2002 to Dec. 2002	Participation in the MEDRC-project “Assessment of the composition of desalination plant disposal brines”

Jul. 2002 to Dec. 2002	Preparation of "Guidelines for the environmental sound management of seawater desalination plants in the Mediterranean region" on behalf of the Mediterranean Action Plan, United Nations Environment Programme
Sep. 2001 to Apr. 2002	Stay in Berkeley, California, and participation in a joint project of the Monterey Bay National Marine Sanctuary, the California Coastal Commission, and the Central Coast Regional Water Quality Board, to elaborate recommendations and guidelines for desalination in the Sanctuary

Education

since May 2007	PhD participant at UNESCO-IHE, Delft, The Netherlands, in a sandwich construction with the University of Oldenburg, Germany
Jul. 2001	MSc Marine Environmental Science (with distinction) from the Institute for Chemistry and Biology of the Marine Environment (ICBM) at the University of Oldenburg, Germany
Dec. 1998	Postgraduate Diploma in Science with distinction in Marine Science from the University of Otago, Dunedin, New Zealand
Jun. 1997	BSc Marine Environmental Science (with distinction) from the Institute for Chemistry and Biology of the Marine Environment (ICBM) at the University of Oldenburg, Germany
1988-1995	Secondary education qualifying for university admission, A-levels (with distinction), Kranich-Gymnasium, Salzgitter, Germany

Awards

May 2009	Recognized by the European Desalination Society (EDS) for presenting the best student paper at the EDS Conference and Exhibition "Desalination for the Environment" in 2009
Apr. 2008	Highly Commended by The Global Water Awards 2008 of Global Water Intelligence for "addressing an important environmental blindspot with absolute academic integrity"
2004	Developing Countries Prize on "Water and Sustainable Development" from the Justus Liebig University Giessen, Germany, and KfW Development Bank, in the category MSc thesis
2003	Weser-Ems Science Award, second prize among university graduates in North-Western Germany

Memberships

- International Desalination Association (IDA)
  member of the peer review editorial board of the Journal Desalination and Water Reuse and editorial board member of Desalination & Water Reuse Quarterly
- European Desalination Society (EDS)
- German Society on Desalination (DME)
- German Society for Marine Research (DGM)

List of publications and presentations

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- S. Lattemann, M.D. Kennedy, J.C. Schippers and G. Amy. Seawater reverse osmosis: a sustainable and green solution for water supply in coastal areas? Submitted to Balaban Desalination Publications for a book on seawater desalination to be published in memory of Sydney Loeb.
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- S. Lattemann. Le dessalement, est-il écologique? La recherche, 421: 2–65, reprinted in Les Dossiers de La Recherche – La mer, 36: 64–67, 2009.
- S. Lattemann and T. Höpner. Why we must have impact studies and mitigation. The International Desalination and Water Reuse Quarterly, 17(2): 36–44, 2007.
- T. Bleninger, S. Lattemann, A. Purnama, H.H. Al-Barwani, R.L. Doneker and G.H. Jirka. BrineDis – environmental planning, prediction, and management of brine discharges from seawater desalination plants. Arab Water World 4: 6–19, 2009.

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- S. Lattemann, M.D. Kennedy and G. Amy. Best available techniques for seawater desalination. International Desalination Association (IDA) World Congress on Desalination and Water Reuse, Dubai, UAE, 2009.
- S. Lattemann, M.S. Anarna, J.C. Schippers, M.D. Kennedy and G. Amy. Multi-criteria decision support system for seawater reverse osmosis plants. European Desalination Society (EDS) Conference and Exhibition on Desalination for the Environment, Baden Baden, Germany, 2009.
- S. Lattemann. WHO guidance on desalination: results of the work group on environmental impacts. International Desalination Association (IDA) World Congress on Desalination and Water Reuse, Maspalomas, Gran Canaria, 2007.

Invited presentations 2007–2009

- Environmental impact and impact assessment of desalination plants. Stichting PostAcademisch Onderwijs, Nieuwegein, The Netherlands, June 2006.
- Why we need environmental impact assessment studies. Exhibition and Congress on Chemical Engineering, Environmental Protection and Biotechnology (ACHEMA), Session on desalination, Frankfurt, Germany, May 2009.
- Environmental aspects of desalination. Symposium of the German Desalination Society (DME) on Thermal Desalination, Essen, Germany, January 2008.
- Desalination plant impacts on the marine environment. AQUA 2008 Conference, Valencia, October 2008.
- Environmental issues in desalination. EXPO Zaragoza 2008, Thematic Week 10, New Water Resources: Reuse and Desalination, Zaragoza, Spain, September 2008.
- Seawater desalination in the Mediterranean – solution to the region's water woes? Seminaire International sur le Dessalement des Aux – Percees Technologiques et Maitrise des Couts, SONEDE, Tunis, July 2008.
- Water environmental aspects. Symposium of the German Desalination Society (DME) and Solar Institute Jülich (SIJ) on Desalination and Renewable Energies, Jülich, Germany, June 2008.
- Desalination's environmental footprint. International Desalination Association (IDA) and Global Water Intelligence (GWI) Conference on Water, Finance & Sustainability, New Directions for a Thirsty Planet, London, UK, April 2008.
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- Environmental impact and impact assessment of seawater desalination plants. European Desalination Society (EDS) Conference on Desalination and the Environment, Halkidiki, Greece, April 2007 (keynote).

Workshops

- S. Lattemann, T. Höpner and T. Bleninger. Seawater desalination and the environment. Impact of brine and chemical discharges on the marine environment. Three day seminar organized by the Middle East Desalination Research Center (MEDRC) and the Government of Germany, Amman, Jordan, December 2008.
- S. Lattemann. Seawater Desalination and the marine environment. Half day seminar hosted by the Water Desalination Conference in Arab Countries (ARWADEX 6), Riyadh, Saudi Arabia, April 2008.
- T. Bleninger, S. Lattemann, R. Doneker, A. Purnama, H. Al-Barwani and G.H. Jirka, Brine discharge management. Two day seminar organized by the Middle East Desalination Research Center (MEDRC), Muscat, Oman, November 2008.
- S. Lattemann and T. Höpner. Seawater desalination and the marine environment. Two day seminar organized by the European Desalination Society and University of L'Aquila, Faculty of Engineering, Italy, February 2008.

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Seawater desalination is a coastal-based industry. The growing number of desalination plants worldwide and the increasing size of single facilities emphasises the need for greener desalination technologies and more sustainable desalination projects. Two complementing approaches are the development and implementation of best available technology (BAT) standards and best practice guidelines for environmental impact assessment (EIA) studies. While BAT is a technology-based approach, which favours state of the art technologies that reduce resource consumption and waste emissions, EIA aims at minimizing impacts at a site- and project-specific level through environmental monitoring, evaluation of impacts, and mitigation where necessary. This book contains a comprehensive evaluation and synthesis of the potential environmental impacts of desalination plants, with emphasis on the marine environment and aspects of energy use, followed by the development of strategies for impact mitigation. A concept for BAT for seawater desalination technologies is proposed, in combination with a methodological approach for the EIA of desalination projects. The scope of the EIA studies are outlined, including environmental monitoring, toxicity and hydrodynamic modelling studies, and the usefulness of multi-criteria analysis as a decision support tool for EIAs is explored and used to compare different intake and pretreatment options for seawater reverse osmosis plants.

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