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Marine monitoring surveys for desalination plants—a critical review

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ABSTRACT

Environmental impact assessment (EIA) studies are standard practice and a regulatory requirement for most new desalination projects today. However, most of the EIA studies are limited to *predictive* information; that is, they gather information on the project and the project's environment before project implementation to make *predictions* about likely impacts. The EIAs may involve comprehensive studies, such as field monitoring, laboratory toxicity testing, and modeling studies. Consequently, the "surprising paucity of useful experimental data, either from laboratory tests or from field monitoring studies", which was observed by the US National Research Council in 2008, has been gradually decreasing. However, there is still a *long-term* research need on the site-specific effects of desalination plants *after project commissioning* has taken place. A main challenge of field research is the adequate design of the monitoring studies, which have to adequately distinguish the effects of the desalination project from natural processes over long periods of time. The existing monitoring studies have so far used a wide range of approaches and methods to investigate the environmental impacts of desalination plant discharges. Shortfalls are often that they are limited in scope, short-term, or localized. In essence, many studies fall short of recognizing the potentially synergetic effects of the single waste components of the discharges on marine organisms and the complexity of the 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 of desalination plant discharges, let alone the cumulative impacts on certain sea areas. This paper conducts a critical review of existing monitoring programs for desalination plants. Shortcomings of current practices are identified and relevant aspects to the design of marine monitoring programs outlined, including the scope of the studies as well as their scientific requirements.

Keywords: Marine monitoring; Desalination plants; Environmental impact assessment

1. Introduction

Environmental impact assessment (EIA) and best available techniques (BAT) are complementary concepts for minimizing the adverse effects of new development projects and technologies on the environment. BAT aim at identifying suitable processes at the technological level. The environmental impact furthermore

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Presented at the International Conference on Desalination for the Environment, Clean Water and Energy, European Desalination Society, 23–26 April 2012, Barcelona, Spain depends on the site-specific characteristics of the project site, which are investigated in EIAs at the project level [1].

EIAs usually comprise a "predictive process", aimed at predicting the likely impacts that would arise from a proposed activity on a given site, and the "postdictive process", aimed at quantifying the actual impacts after they have taken place. Both processes require extensive experimental data, usually involving extensive field monitoring programs accompanied by laboratory studies that may range from toxicity tests to miniature models and computer modeling approaches.

In 2008, the US National Research Council (NRC) still attested a "surprising paucity of useful experimental data, either from laboratory tests or from field monitoring" [2] to describe the actual effects of desalination plants on the environment. Site-specific assessments of the impacts of source water withdrawals and concentrate management are the long-term research needs identified by the NRC. Although more EIA studies of desalination projects have become available since the NRC published its report, these often contain information only from the *predictive* process. One of the first and longest studies that also provides experimental data after project implementation has been carried out for the Perth seawater reverse osmosis (SWRO) project [3]. Comprehensive pre- and postdictive studies have henceforth also been conducted for other Australian projects, such as for the Sydney, Gold Coast, and Olympic Dam SWRO projects [4–6]. The trend to conduct long-term monitoring after project implementation needs to be continued, also in other sea areas.

The NRC furthermore identified a need for improved monitoring and assessment protocols for evaluating the potential ecological impacts of concentrate discharge. In other words, still missing are the methodological frameworks for conducting sound environmental studies. The core of the problem is to design a monitoring program that can adequately distinguish the effects of the desalination project from natural processes. Many of the existing monitoring studies have used a wide range of approaches and methods to investigate the environmental impacts of desalination plant discharges. Shortfalls were often that they are limited in scope (i.e. addressing only one effect, such as elevated salinity on a specific species), short-term (i.e. lacking a continuous baseline and operational effects monitoring), and localized (i.e. not taking effects over a wider area into account which may arise from the dispersal of pollutants) [7]. In essence, 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 of desalination plant discharges, let alone the cumulative impacts on certain sea areas.

As mentioned above, one of the most comprehensive environmental monitoring programs to date has been 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 these preconstruction studies by the National Institute of Water and Atmospheric Research 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 could be accepted with a high degree of confidence [8]. As a result, more extensive studies were initiated, including marine baseline studies, a real-time monitoring system before and during operations, and laboratory tests on toxicity [9]. 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, including an external peer review.

2. Environmental monitoring concepts

Monitoring studies can follow a *stressor*-based approach or an *effects*-based approach, or preferably both. The stressor-based approach consists of identifying the potential stressors associated with a project over its lifetime (e.g. concentrate discharge), potentially affected receptors in the environment (e.g. marine organisms near the outfall), and pathways for interaction (e.g. stress from elevated salinity which causes a decline in species abundance). The approach assumes that all stressors associated with a project are known. However, it falls short of recognizing that cumulative stressor sources may exist within an aquatic ecosystem. These may be due to other projects in the area, may have natural causes, and may be partially unknown. The stressor-based approach should therefore be combined with an effects-based approach, which measures the "accumulated environmental state" of the ecosystem by comparing the environmental indicators between developed and undeveloped sites to identify the 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 [10].

2.1. Monitoring designs: BACI and BACIPS

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 programs based on the BACI design have the objective to isolate the impact from the "noise" that is introduced by natural temporal and spatial variability [11]. However, there are many practical problems with the BACI approach which need to be resolved by more sophisticated designs in order to actually detect any impacts.

One practical problem is the large temporal variance of many populations, which is reflected in very "noisy" abundances [12]. To capture this temporal variance, the BACI design can be refined by having several simultaneous dates before and after the perturbation in both the control and the impact sites ("paired sampling", BACIPS). 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 (Δ_B–Δ_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 [11]. 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 [10].

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 [13]. 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 [14].

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 whether 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, for example, 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 resolved 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 [12].

For example, if the outfall of a desalination plant is to be placed onto a marine headland 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. 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–that is, 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 [5]. 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 [4,15]. Adequate temporal and spatial replication was thus provided for both plants, but in the Sydney case, the reference sites were also selected in different distances from the outfall.

The examples illustrate what kind of temporal and spatial replication may be required to ensure adequate statistical power of the monitoring studies and to achieve a given level of confidence in the estimate of the impact size. Another problem remaining, however, is that a study that would permit an estimation of the number of sampling dates (temporal replicates) and control sites (spatial replicates) in the absence of sufficient preliminary data must often be designed.

2.2. Baseline and operational monitoring

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. The main objective of the preimpact studies is therefore 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 the following:

- *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.
- *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.

A second objective of the preimpact 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 to assess the accuracy of predictions, to detect new impacts (operational effects monitoring), and to ensure that regulatory requirements and quality standards are being met (compliance monitoring). 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 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 short for a variety of reasons [11]. Baseline studies for the Perth project, for example, were initially constrained to using mostly existing data due to significant time pressure. Generally, the greater the number of replicates, the greater the probability of distinguishing putative impacts from natural variation. Temporal replication should preferably involve a nonregular frequency of sampling to avoid coincidences with natural cycles [16].

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 sites, 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 [15].

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.

3. Marine monitoring 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. The scope of the EIA and specialist studies should have been determined during the scoping phase of an EIA. The proposed components and general scope of a marine monitoring program for desalination projects are illustrated in Fig. 1. Monitoring here refers to the living and nonliving *environmental* resources, although it should be noted that an EIA typically includes socioeconomic and public health implications as well. However, except for large projects with major public health risks and socioeconomic impacts, an EIA will often rely on existing and readily available data. Therefore, these aspects are not covered here.

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, such as 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 [15]. This is relevant to identify not only the potential cumulative impacts on the environment, but also the environmental consider-

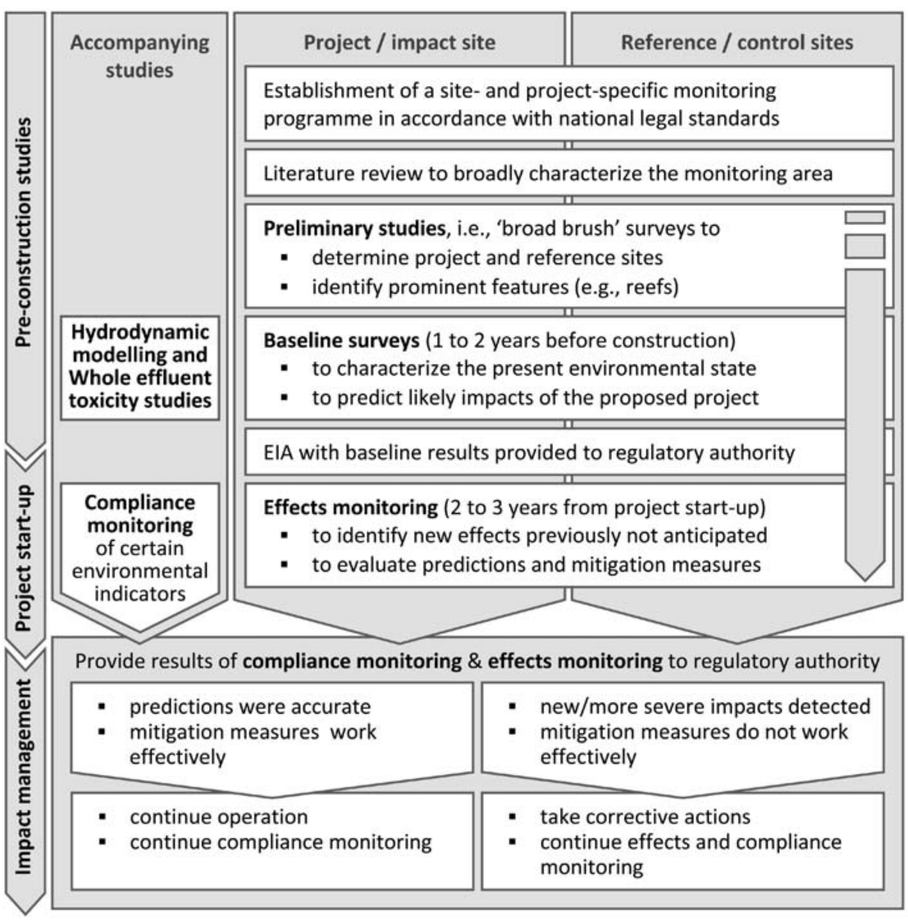

Fig. 1. Outline of a monitoring program for desalination projects.

ations 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 the 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 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.

3.2. Baseline and operational effects monitoring

3.2.1. 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 operational effects monitoring and can provide engineers with information on water quality conditions to determine a robust pretreatment process [15] and an 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. 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, and three buoys were deployed to measure salinity, temperature, and DO for the first Perth project [9]. 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 [17]. 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.

Chemical analysis typically includes the major nutrients (phosphate, silicate, nitrate, nitrite, ammonia), organic carbon content, and chlorophyll-a. Nutrient studies in tropical and subtropical waters, that is, where desalination plants are often located, may require the detection of compounds at extremely low levels and consequently a greater vigilance than might be needed for water samples from temperate regions. Depending on the results from the pollution source survey, 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 that may be present in the feed water and would be enriched in the concentrate [18]. 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 [19]. 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* that 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.

3.2.2. 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 the operational effects monitoring and can be used to identify the intake and outfall locations and pipeline transects. If the intake and outfall pipelines are to be drilled horizontally from an onshore site, information on the substratum sediments is additionally required. Surveys usually combine acoustic remote devices, underwater cameras, and sampling.

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 topographic features, underwater objects, and the texture of the seafloor surface (mud, sand, gravel, and 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 rocky areas (e.g. reefs) or soft-bottom areas (e.g. sea grass beds). For assessing near-field changes in seabed morphology such as scour and deposition around intake and outfall structures placed on the seafloor, a high-resolution bathymetry survey of the spread of sediment types across the site can be carried out.

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. metal analysis as carried out for the Ashkelon project). Sediment sampling is often carried out simultaneously with biological sampling. *Benthic species* are subdivided into those living in the seafloor sediments (infauna) and those occupying the surface of soft and hard bottom substrates (epifauna). As a third group, demersal fish species may be included in the surveys, 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, to more detailed and focused studies. The latter involve small-scale sampling in the project area, that is, 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 $150 \times 200\,\mathrm{m}^2$ [15]. 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 the migration of fish into coastal areas for spawning.

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, preexisting monitoring data for the power plant may be available and could be used to establish the baseline for the desalination plant.

Quantitative samples of biological resources result in species densities per volume or sample area, giving either numbers for individuals or percent coverage, for example 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.

The outputs of these inventories are species lists and distribution maps. Distribution maps require a multivariate analysis of species distributions at the habitat and species level (e.g. sea grass 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 seagrass). The spatial distribution data are often integrated into a geographic information system.

3.2.3. Nekton

Nekton refers to the aggregate of actively swimming organisms, which includes certain invertebrates such as squid or larger shrimps, fish, or reptiles (i.e. sea snakes, turtles). 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. The first step would be to identify whether important recruitment, feeding and overwintering areas, or migration routes exist within the project area, with particular emphasis on species that are of conservation importance. If this information is not available from existing data sources or if there is a local conservation concern, a more detailed monitoring program 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 min of duration, while sampling with small trawls should be 5–15 min of 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 [20].

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, 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 with 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) [6], 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 [21].

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 noninvasive measures. In most cases, a reasonable approach will be to carry out a literature survey 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 *species list* and identify the conservation status of species.

3.2.4. Seabirds and mammals

Monitoring of seabirds and mammals has the objective to establish a list of species that may occur in the project area and to ascertain whether a special conservation interest exists for that area. This typically involves a literature review and qualitative surveys 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.

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 at all times. An EIA is therefore to some extent always implicitly employing an indicator approach, 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 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 DO 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 socioeconomic importance (in terms of fisheries and public health) of a species. It is not recommended to build a rigid set of criteria into regulatory frameworks for selecting an indicator; rather, a variety of taxa from different trophic levels should be considered [22].

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 3.2);
- characterization of the desalination concentrate through whole effluent toxicity (WET) testing (section 3.5), using selected local species;
- selection of an indicator that 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 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.

The monitoring of indicators can take place as part of the operational 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 operational effects monitoring studies, which are usually limited to 2–3 years after project implementation.

3.4. Hydrodynamic modeling

Hydrodynamic modeling studies are usually part of the baseline investigations (Fig. 1). They have the objective to predict the 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 concentrations of process 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 can subsequently be modified if necessary.

The mixing behavior of an effluent mainly depends on the oceanographic conditions in the receiving water body, the discharge practice, and 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 [15], such as tidal cycles or seasonal currents. Brine discharge modeling should adequately cover the near-field and far-field processes, 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 [23].

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 [23]. Alternatively, tracer experiments with dye can be carried out in the project area to evaluate the model's ability to reproduce advection and dispersion [17]. Finally, it is important that the quality of the models and the modeling results are reviewed and accredited by an independent expert or institution [15].

3.5. Bioassay studies

If bioassay toxicity studies are carried out, these should preferably be 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 [15]. 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. If bioassay studies are carried out as part of the baseline investigations, 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 [24]. On the basis of 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 1). A species protection level (SPL) 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. On the basis of the WET tests with 15 species from four trophic levels, it was predicted that a dilution ratio of seawater to desalination concentrate of 45:1 (SPTV) will protect 99% (SPL) of the marine species in the area, corresponding to a salinity increase of 0.7 units above ambient. On the basis of the hydrodynamic modeling studies, this dilution will be achieved within 300 m from the outfall in 90% of all times. A 100% species protection level 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) [6]. The bioassay studies for the Sydney project showed that salinity was the key source of toxicity of the whole effluent [4,15].

A similar methodology for testing the long-term salinity tolerance of marine species was applied for two SWRO projects in California [25]. On the basis of the 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 biometric test with 18 species in a single aquarium over a period of 5 months was carried out to investigate the chronic effects at this salinity. In addition, salinity tolerance tests were carried out over a range of salinities to investigate whether marine organisms will be able to survive periodic extreme (worst case) salinity conditions. Three local species that 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 the 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 US EPA [26,27].

4. Summary and conclusions

An increasing number of EIAs for desalination projects has become available in recent years. However, most studies contain information from the preconstruction phase only, while data from post-construction are still relatively scarce. In some locations, comprehensive perennial monitoring programs have been initiated, which will likely provide a better database for assessing the actual environmental impacts of SWRO projects in the future. The importance of these studies is evident, if one considers the following example from the power industry. Ambrose et al. [28] compared the actual impacts of the cooling water discharges from a power plant in Southern California, established by a 15-year monitoring program, with predictions made in the EIA which had been generated in three different ways. The following conclusions were drawn:

- Almost all the testimonies of scientists before the permitting agency, *which were based on professional judgment with little scientific analyzes*, were wrong;
- The accuracy of the final environmental statement, *based on standard assessment methods at that time*, was mixed but generally not too high;
- The predictions of the marine review committee, *based on a comprehensive baseline study over several years*, were the most accurate, but still showed inaccuracies.

The findings show a clear correlation between effort and accuracy of the predictions; however, even comprehensive perennial studies cannot predict with complete certainty what will actually happen in the environment. EIAs, like other observational studies, are likely to remain "messy" even after a conscientious effort to apply the appropriate techniques and mathematical statistics [29].

The number of publications discussing the environmental impacts of desalination plants has been steadily increasing over the last few years. Most of the published work remains hypothetical in the absence of more rigorous follow-up studies than is presently the case. Reputable journals would reject results that were derived with less than good scientific practice in "academic" field experiments. The same standard must apply for EIAs (although EIAs are "applied" science) unless the entire assessment should become a random process [12].

Table 1
WET data for the Perth [3], Sydney [4], Gold Coast [5], and Olympic Dam [6] projects: The 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 SPL. The SPTV is compared with the dilution ratio of the diffuser design

Plant	SPL	SPTV	Diffuser dilution ratio	WET test species
Perth	95%	12.3:1	45:1	Tests at commissioning and after 12 months of operation
	99%	15.1:1		72-h macroalgae germination (*Ecklonia radiata*)
				72-h macroalgae growth test (*Isochrysis galbana*)
				48-h mussel larval development (*Mytili sedulis*)
				28-d copepod reproduction test (*Gladioferens imparipes*)
				7-d larval fish growth test (*Pagrus auratus*)
Sydney	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	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	Six species from more than three 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	99%	45:1	45:1 dilution within:	15 species from more than four trophic levels representative of the local ecosystem, including acute and chronic tests with early life cycle stages, juveniles and adults:
			0.3 km (90% of time)	72-h microalgae chronic growth rate inhibition test (*Nitzschia closterium and Isochrysis galbana*)
			1.1 km (99% of time)	
			2.2 km (100% of time)	
	100%	85:1	85:1 dilution within:	72-h macroalgae chronic germination success (*Ecklonia radiata and Hormosira banksii*)
			1.1 km (90% of time)	48-h chronic copepod reproduction (*Gladioferens imparipes*)
			2.8 km (99%of time)	96-h actue prawn post-larval toxicity test (*Penaeus monodon*)
			3.9 km (100% of time)	21/28-d juvenile/adult prawn growth (*Melicertus latisulcatus*)

(*Continued*)

Table 1 (*continued*)

Plant	SPL	SPTV	Diffuser dilution ratio	WET test species
			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	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*)

Regulatory agencies may still be reluctant to require rigorous operational monitoring studies, and project developers are understandably opposed to funding it. 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. As spatial and temporal variability could falsely be interpreted as an impact without sufficient replication, adequate monitoring could therefore be understood as an "insurance" against unwarranted claims [13].

In this context, it is also noteworthy that both operational 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 the operational effects monitoring, or in case that trigger values are exceeded during compliance monitoring.

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

References

[1] S. Lattemann, M.D. Kennedy and G. Amy. Seawater desalination - a green technology? J. Water Supply: Res. Technol.-AQUA, 59(2–3) 134–151.
[2] Committee on Advancing Desalination Technology. Desalination: A national perspective. Water Science and Technology Board, National Research Council of the National Academies, National Academies Press, Washington DC, www.nap.edu/catalog/12184.html (last accessed on 25.02.2012), 2008.
[3] S. Christie, V. Bonnélye, Perth, Australia: Two-year feedback on operation and environmental impact, IDA World Congress on Desalination and Water Reuse, The Palm, 2009.
[4] S. Trousdale, E. Henderson, Sydney's desalination plant: Addressing environmental issues using innovative design, planning and monitoring, IDA World Congress on Desalination and Water Reuse, The Palm, 2009.
[5] N. Cannesson, P. Johnstone, M. Mitchell, S. Boerlage, Community, environmental and marine impact minimisation at the Gold Coast desalination plant, IDA World Congress on Desalination and Water Reuse, The Palm, 2009.
[6] Olympic Dam Expansion Supplementary Environmental Impact Statement, Chapter 17, marine environment. http://www.bhpbilliton.com/home/aboutus/regulatory/Pages/default.aspx, 2011 (last accessed on 25.02.2012).
[7] S. Lattemann, K. Mancy, H. Khordagui, B. Damitz, G. Leslie, Desalination, Resource and Guidance Manual for Environmental Impact Assessments, United Nations Environment Programme (UNEP), Nairobi, 2008.
[8] B. Spigel, R. Davies-Colley, D. Spooner, J. Cooke, Peer review of studies for desalination plant discharge, Cockburn Sound. National Institute of Water and Atmospheric Research (NIWA), prepared for EPA Services Unit, Department of Environment, Western Australia, 2005.
[9] G. Crisp, M. Rhodes, D. Procter. Perth seawater desalination plant. Blazing a sustainability trail. IDA World Congress on Desalination and Water Reuse, Maspalomas, Gran Canaria, 2007.
[10] M. Dube, K. Munkittrick, Integration of effects-based and stressor-based approaches into a holistic framework for cumulative effects assessment in aquatic ecosystems, Human and Ecological Risk Assessment 7(2) (2001) 247–258.

[11] C. Osenberg, R. Schmitt, S. Holbrook, K. Abu-Saba, A. Flegal, Detection of environmental impacts: Natural variability effect size and power analysis, In: R. Schmitt, C. Osenberg (Eds), Detecting Ecological Impacts: Concepts and Applications in Coastal Habitats, Academic Press, San Diego, CA, pp. 83–108, 1996.
[12] A. Underwood, On beyond BACI: Sampling designs that might reliably detect environmental disturbances, In: R. Schmitt, C. Osenberg (Eds), Detecting Ecological Impacts: Concepts and Applications in Coastal Habitats, Academic Press, San Diego, CA, pp. 151–175, 1996.
[13] C. Osenberg, R. Schmitt, Detecting ecological impacts caused by human activities, In: R. Schmitt, C. Osenberg (Eds), Detecting Ecological Impacts: Concepts and Applications in Coastal Habitats, Academic Press, San Diego, CA, pp. 3–16, 1996.
[14] A. Underwood, Beyond BACI: The detection of environmental impacts on populations in the real, but variable, world, J. Exp. Mar. Biol. Ecol. 161 (1992) 145–178.
[15] C. Port, S. Roddy, S. Trousdale, Sydney's SWRO project—working towards sustainability, The International Desalination and Water Reuse Quarterly 18(4) (2009) 21–33.
[16] A. Stewart-Oaten, W. Murdoch, K. Parker, Environmental impact assessment: Pseudoreplication in time? Ecology 67(4) (1986) 929–940.
[17] K. Zic, D. Couton, Field monitoring and oceanographic modeling for a large desalination plant in Australia, in: 5th International Conference on Marine Waste Water Discharges and Coastal Environment, Dubrovnik, Croatia, 2008.
[18] Tampa Bay Water. www.tampabaywater.org.
[19] I. Safrai, A. Zask, Reverse osmosis desalination plants—marine environmentalist regulator point of view, Desalination 220 (2008) 72–84.
[20] CEFAS, Guidance note for environmental impact assessment in respect of FEPA and CPA requirements. Centre for Environment, Fisheries and Aquaculture Science on behalf of the UK Department for Environment, Food and Rural Affairs (Defra) and the Department for Transport (DfT). http://www.cefas.co.uk/publications/files/windfarm-guidance.pdf, 2004 (last accessed on 25.02.2012).
[21] J. Dupavillon, B. Gillanders, Impacts of seawater desalination on the giant Australian cuttlefish Sepia apama in the upper Spencer Gulf, South Australia, Mar. Environ. Res. 67(4–5) (2009) 207–218.
[22] G. Jones, U. Kaly, Criteria for selecting marine organisms in biomonitoring studies, In: R. Schmitt, C. Osenberg (Eds), Detecting Ecological Impacts: Concepts and Applications in Coastal Habitats, Academic Press, San Diego, CA, pp. 29–48, 1996.
[23] A. Niepelt, T. Bleninger, G. Jirka. Desalination brine discharge modelling. Coupling of hydrodynamic models for brine discharge analysis, in: 5th International Conference on Marine Waste Water Discharges and Coastal Environment, Croatia, 2008.
[24] ANZECC/ARMCANZ, Australian and New Zealand guidelines for fresh and marine water quality. Environment and Conservation Council and Agriculture and Resource Management Council of Australia and New Zealand. www.mincos.gov.au, 2000 (last accessed on 25.02.2012).
[25] N. Voutchkov, Salinity tolerance evaluation methodology for desalination plant discharge, Desalin. Water Treat. 1 (2009) 68–74.
[26] US EPA, Methods for measuring the acute toxicity of effluents and receiving waters to freshwater and marine organisms. US Environmental Protection Agency, Office of Water. www.epa.gov/waterscience/methods/wet/, 2002 (last accessed on 25.02.2012).
[27] US EPA, Short-term methods for estimating chronic toxicity to marine and estuarine organisms. US Environmental Protection Agency, Office of Water. www.epa.gov/waterscience/methods/wet/, 2002 (last accessed on 25.02.2012).
[28] R. Ambrose, R. Schmitt, C. Osenberg, Predicted and observed environmental impacts: Can we foretell ecological change? In: R. Schmitt, C. Osenberg (Eds), Detecting Ecological Impacts: Concepts and Applications in Coastal Habitats, Academic Press, San Diego, CA, pp. 345–369, 1996.
[29] A. Stewart-Oaten, Problems in the analysis of environmental monitoring data, In: R. Schmitt, C. Osenberg (Eds), Detecting Ecological Impacts: Concepts and Applications in Coastal Habitats, Academic Press, San Diego, CA, pp. 109–131, 1996.