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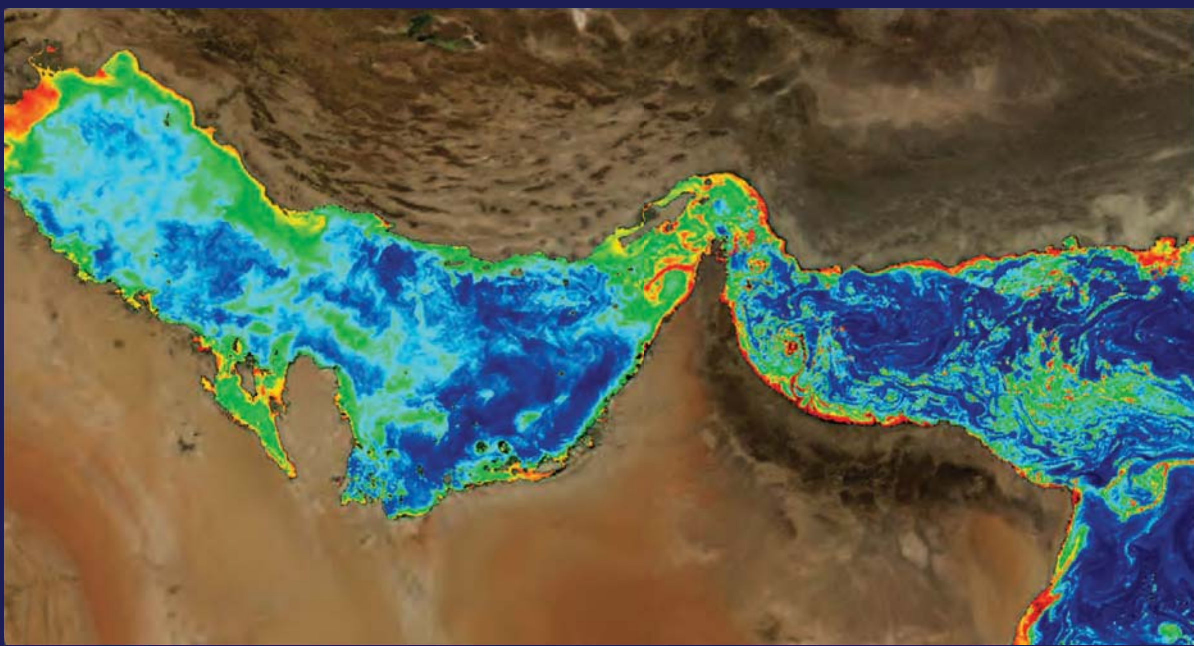
Harmful Algal Blooms (HABs) and Desalination: A Guide to Impacts, Monitoring, and Management



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6 SEAWATER INTAKE CONSIDERATIONS TO MITIGATE HARMFUL ALGAL BLOOM IMPACTS

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6.1 INTRODUCTION

Seawater intakes are a key element in the design, construction and success of desalination plants. Various intake options exist and are generally classified based on their abstraction depth. Surface ocean intakes abstract seawater from the top of the water column or at depth, while subsurface[1] intakes are embedded in the seabed or beach, thereby pre-filtering the abstracted seawater. Location, intake type and depth are important determinants of water quality. Intakes are also the first point of control in minimizing the ingress of algae into a plant or where algal impacts first manifest.

Originally the more robust thermal desalination processes dominated the desalination market where feedwater quality was not the primary driver in determining intake type or location. Instead, feedwater supply was critical, as thermal plants were configured as cogeneration power/desalination plants with common intakes with large volume requirements to generate

[1] Note that 'subsurface' in this context differs from common oceanographic usage, in which the term refers to waters just below the air/water interface.

both power and water. Intake and screening systems were often limited to shallow nearshore intakes with screens sized to meet the necessary seawater quality for power plant, multi-stage flash (MSF) and multi-effect distillation (MED) condenser tubes (Pankratz 2015). Macro-algal seaweed species were initially a significant issue in thermal desalination plants, completely blinding intake screens or clogging settling basins (Figure 6.1). In the mid-1970s, the availability of MSF thermal plants in Libya was dramatically reduced to 100 days/year, with seaweed blockage of the intake pipes the third leading cause for plant outages. At the Zuara plant intake, up to 800 m^3 of seaweed was removed every second day during winter when seaweed became dislodged from the seabed at the end of summer and during storms (Kreshman 1985; 2001). Due to advances in the design of intake systems, the extent of macro-algal intake blocking has been greatly reduced at thermal desalination plants and now mainly results in short term outages.

Figure 6.1. Dry seaweed extracted from the Zwitina desalination plant intake channel in Libya. Photo: Kershman 1985.

Nowadays with seawater reverse osmosis (SWRO) dominating the desalination market, microscopic algal species (phytoplankton) have been more problematic. Occasionally issues have occurred at plant intakes when a high suspended solids load of phytoplankton and debris have overloaded trash racks and/or clogged intake screens (Figure 6.2). In some cases, these impacts have been severe. The notorious 2008/2009 bloom of *Cochlodinium polykrikoides* in the Gulf of Oman resulted in the frequency of cleaning seawater intake screens at Sohar increasing to every 4 hours (Sohar Case Study, Chapter 11). More often adverse impacts are observed in downstream SWRO pretreatment processes or through the promotion of (bio)fouling on membranes as microscopic algae and algal organic matter (AOM) pass through conventional open intakes and screens.

Figure 6.2. Algae and other marine debris blocking the Traveling Water Screens (left) at a SWRO desalination plant in the Indian Ocean and the screens following cleaning (right). Photos: Domingo Zarzo Martinez, Valoriza Agua S.L.

The potential for phytoplankton and AOM to be entrained into SWRO plant intakes, the focus of this chapter, varies greatly. In addition to the intake system design characteristics, prevailing marine conditions, nutrient concentrations at the site, the type, motility and concentration of the algal bloom species play a role. Intake characteristics are recognized to have a significant effect on raw seawater quality and therefore the pretreatment processes required, as well as limiting marine environment impacts which can be a major concern in some projects. Consequently, more attention is given to the selection and location of intake systems in SWRO feasibility studies and during design.

In areas prone to algal blooms, subsurface or open intakes abstracting seawater at depth are often considered a solution to reduce the ingress of floating or surface-concentrated algal blooms into desalination plant intakes. Subsurface intakes offer the advantage that they serve both as a water intake and as pretreatment for a SWRO plant. The seawater is filtered during passage through the strata of the subsurface intake, removing algae and natural organic matter, including components of AOM by both physical and biochemical processes, providing a high-quality feedwater, thereby potentially reducing or replacing conventional pretreatment processes (Missimer et al. 2013; Rachman et al. 2014; Dehwah et al. 2015; Dehwah and Missimer 2016). The effectiveness of these strategies for reducing the entrainment of algae and associated AOM into an intake is discussed below. It should be noted this is a little–studied area in the desalination industry, especially during algal bloom events. Therefore, research on the removal of fractions of natural organic matter (NOM) such as biopolymers produced by both bacteria and algae are examined here, as results may be indicative of what can occur during an algal bloom. Finally, other factors such as engineering constraints, environmental concerns, costs, construction time, and operability may ultimately drive the selection and siting of an intake. A brief overview of approaches to determine the seawater intake for a project is therefore provided in the last part of this chapter.

6.2 INTAKE OPTIONS FOR SWRO DESALINATION PLANTS

Seawater desalination plants require an intake system that is capable of reliably delivering the seawater flow to meet production requirements. Secondly, and arguably equally important for a SWRO plant, the intake ideally delivers water that is high quality and consistent over time - free of pollutants with a low solids and organic load to minimize pre-treatment complexity and chemical consumption. The latter reduces the generation of waste requiring disposal. The key environmental concern in intake design is to reduce the potential for marine life mortality. This may occur through *impingement* of marine life i.e. when larger organisms (typically juvenile and adult stages) are trapped against an intake screen—and *entrainment*—when smaller organisms (typically phytoplankton and early life stages− eggs and larvae) pass through a screen into the process during intake operation.

Intakes therefore, not only play a significant role in SWRO plant capital and operating costs, but designs are highly site-specific—possibly more so than any other aspect of the plant—and have a considerable impact on the operational and environmental aspects of the plant. Additionally, as the initial step in the pretreatment process, the intake effectiveness plays an important role in determining the performance of downstream processes. Intakes are broadly classified based on their abstraction depth and location from shore as described below and are discussed more fully in the following sections with respect to preventing entrainment of microscopic algae:

- Open ocean (or surface) intakes - where seawater is abstracted at the sea surface, within 15 m of the surface (shallow intakes) or at water depths of greater than 15-20 m. Intakes may be located onshore, nearshore or offshore. Feedwater to the plant derived from surface intakes is dependent on the inherent seawater quality and will fluctuate depending on prevailing marine and site conditions; and

- Subsurface intakes which abstract seawater from beneath the seabed or beach. Filtration through the seabed strata generally provides a superior water quality due to the removal of suspended solids, turbidity and some organics.

6.3 SURFACE INTAKE AND SCREEN OPTIONS

Large-scale desalination plants have traditionally employed open ocean surface intakes with associated screening and chlorination plants. In most arrangements, the pump station and screening chamber are located onshore and directly connected to the open ocean by means of a concrete channel or jetty or an intake pipeline (or tunnel) which can extend hundreds of meters into the sea (Figure 6.3). For shallow offshore areas, it is common to locate the raw water intake structure well beyond the surf zone, where it is less vulnerable to damaging wave action and turbidity entrainment. In some instances, it may be up to 500 m or more from shore depending on the bathymetry, to enable the abstraction of water from deeper, less environmentally sensitive areas or to obtain more consistent water quality that is less susceptible to varying debris loads. Note that being far from shore does not necessarily reduce the potential impact from algal blooms, as many originate offshore and are transported to the nearshore waters by winds and currents. Likewise, blooms near the shore can be transported further offshore by what are termed "upwelling-favorable" winds (Chapter 1).

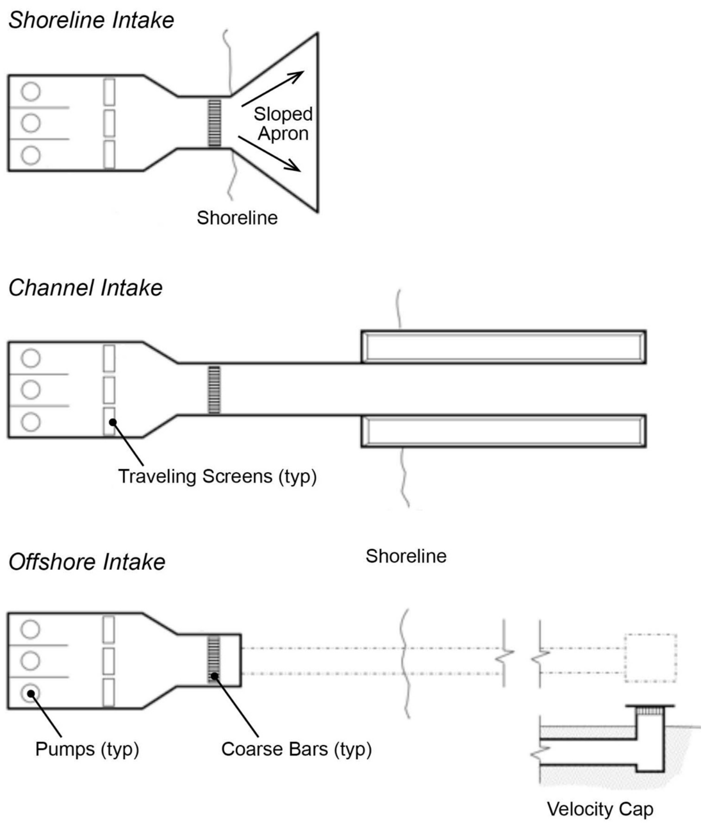

Figure 6.3. Surface seawater intake alternatives – conventional shoreline intake through a lagoon or channel and offshore with typical coarse and fine screening arrangements.

To remove flotsam and larger debris, intake seawater is typically screened at the head of the plant by coarse primary screening (bars) followed by finer secondary screening to protect pumps and downstream processes. Screens need to be cleaned to remove pressure losses caused by debris fouling and ensure flow, and/or sized with larger openings to avoid marine build up. Typical screening options for desalination plants are discussed in more detail in section 6.3.1.

In addition to screening of the seawater, chlorination is often applied at surface seawater intakes to control marine biofouling on screens, piping, and pumps, although the practice varies based on the desalination process employed and whether the intake also provides water for cooling or other purposes. Thermal desalination plants generally add 1 to 2 mg/L of chlorine continuously to maintain a 0.15 to 0.3 mg/L residual, and shock doses of up to 8 mg/L for 15-30 minutes several times a day. Most SWRO plants practice intermittent chlorination/dechlorination with doses of up to 10 mg/L added for up to two hours on a daily, weekly or biweekly basis. Chlorination is now most commonly used on a periodic basis rather than continuously because it is known to cause biofouling of the downstream membranes (Winters et al. 1997). As an additional measure, components of screens e.g. bars may be constructed of alloys with biocidal properties such as cupronickel 90/10 to prevent marine biogrowth.

Historically, large-scale thermal (MSF or MED) seawater desalination plants were coupled to electric power plant to provide a steam source for the distillation process as cogeneration plants. As power plants require large volumes of cooling water to condense power-cycle steam, they are also able to share their seawater intake and screening infrastructure with the desalination plant. Intake arrangements for such cogeneration plants are often open seawater intakes of the channel or lagoon type which are connected to the screening chamber located at the shoreline, or some distance inland. The shared intake system, based on that for power plant cooling water, were developed more than one hundred years ago and typically consist of vertical trash racks (100 mm spacing) fitted with raking machines which can remove floating debris and filamentous algae. This is followed by onshore screening chambers, or wet wells, equipped with mechanically cleaned, traveling water screens or rotary drum screens such as that shown in Figure 6.4.

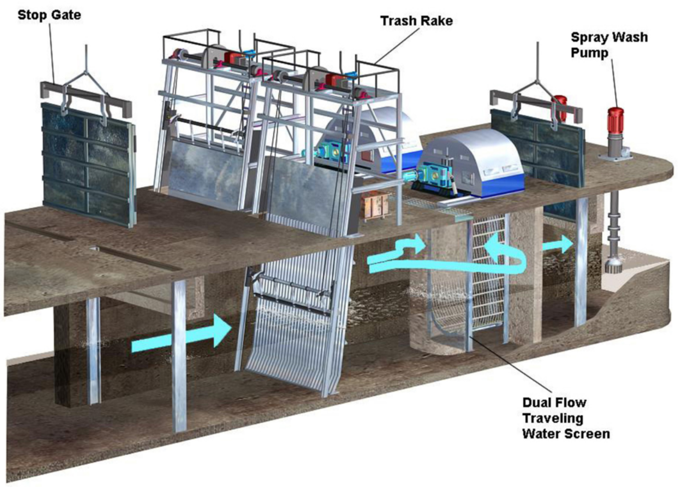

Figure 6.4. Traditional open seawater intake showing primary screening using trash racks followed by fine screening using Traveling Water Screens. Photo: Evoqua.

Initially, large capacity SWRO plants followed this intake arrangement, particularly those co-located with electric power plants or configured as hybrid desalination systems (i.e. thermal combined with SWRO systems). Nowadays, the lower capital and operating costs of SWRO compete favorably with thermal desalination processes, particularly for stand-alone desalination plants where no existing intake or outfall exists. Even if an existing open ocean intake were available, the SWRO process requires feedwater with a much lower level of suspended solids, both in terms of particle size and volume, than thermal processes, which may necessitate a purpose-built intake.

6.3.1 Onshore and offshore intake screening

Screen selection and configuration is influenced by a variety of factors such as type and abundance of marine flora and fauna at site, impingement onto the screen, the risk of entrainment into the intake, the type of pumps and pretreatment proposed downstream and its ability to remove and handle solids. The most common techniques to mitigate the impingement and entrainment of marine life is to lower the velocity of water through the intake screen to less than 0.15 m/s and reduce the size of the screen openings to 1 mm or less, respectively. Algae would be defined as entrainable organisms due to their size. Some entrainable organisms have limited to no swimming ability and therefore lack the ability to avoid the intake flow regardless of velocity (Hogan 2015).

Active (moving) or mechanical screens used for fine screening are located onshore in concrete channels either at the far end of a forebay or a longer channel that extends out beyond the surf zone. Alternatively, the screens may be installed in a wet well or pump station that is connected to the sea by a pipe that extends out into the sea and terminated in a coarse screened inlet head or a velocity cap. Unless the intake terminal of an offshore intake is fitted with a passive (stationary) screen system, the onshore pump station should be equipped with fine wire mesh screens to protect downstream pumps and pretreatment equipment (Pankratz 2015). The mesh size of the mechanical screens generally depends on the desalination process. In thermal MSF and MED plants the mesh openings range from 6 to 9.5 mm with smaller mesh openings of 0.5 to 5 mm sometimes used for MED plants as MED needs finer filtration. For MED the allowable particle size for seawater going through the spray nozzles is <0.5 mm.

6.3.1.1 Traveling Water Screens

Traveling Water Screens, also referred to as Traveling Band Screens, have been employed on seawater intakes since the 1890s. The screens are equipped with revolving wire mesh panels having 6 to 9.5 mm openings, although environmental regulations in the United States—specifically §316(b) of the US EPA Clean Water Act—mandate that many intake screens employ finer mesh screens with openings as small as 1.0 mm to minimize entrainment. The screens are also usually designed so that the maximum water velocity through the screen is less than 0.15 m/s.

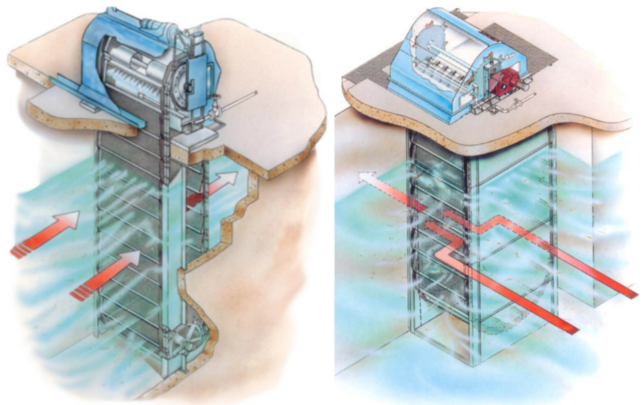

Figure 6.5. Traveling Water Screens: Through-Flow, left and Dual Flow, right. Photos: Evoqua.

As the wire mesh panels revolve out during flow, a high-pressure water spray removes accumulated debris by washing it into a trough for dewatering and further disposal.

There are two distinct types of Traveling Water Screens: a *Through-Flow* screen in which the screening panels are oriented perpendicular to the flow with only the ascending panels utilized as available screening area; and the *Dual Flow* or central-flow type screen in which the screening panels are oriented parallel to the flow, utilizing both the ascending and descending panels as active screening area (Figure 6.5). Besides providing more active screening area per unit, a dual flow traveling water screen virtually eliminates the chances of 'carry over', where debris not removed by the spray system would otherwise fall into the screened water side of the unit and enter the pumps.

6.3.1.2 Rotating Drum Screens

Rotating Drum Screens are an alternative to Traveling Water Screens, and consist of wire mesh panels mounted on the periphery of a large cylinder that slowly rotates on a horizontal axis (Figure 6.6). They are cleaned with a spray wash system similar

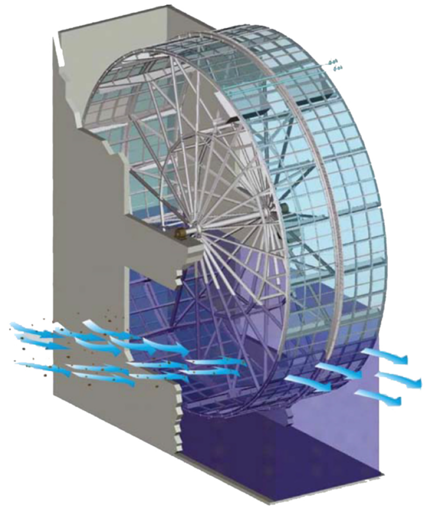

Figure 6.6. Rotating Drum Screen. Photo: Ovivo.

to Traveling Water Screens. Drum Screens may range up to 4 m in diameter and have similar size openings to the traveling water screens.

6.3.1.3 Velocity Caps

As mentioned, above, the most common technique to minimize impingement mortality of marine life is to reduce the through-screen velocity of an intake structure to ≤0.15 m/s allowing fish to swim away from the currents generated at the intake (EPA 2014). An alternative is to fit the vertical riser of an offshore open intake with a “velocity cap” which acts as a behavioral deterrent to guide aquatic organisms away from the intake structure. A velocity cap is a horizontal, flat cover located slightly above the terminus of the riser which provides a narrow opening for the entrance of seawater. Intake water drawn through the openings in the velocity cap is converted from vertical flow to horizontal flow into the pipe. Rapid changes in horizontal flow will provide a physiological trigger in fish inducing an avoidance response thereby avoiding impingement (EPA 2014). Some capped intake risers operate at lower through-screen velocities (≤0.15 m/s), but may not function as an effective fish diversion technology. Most velocity caps operate at higher entrance velocity with the change in flow pattern created by a velocity cap operating at an entrance velocity of over 0.3 m/s, and as high as 0.9 m/s, triggering an avoidance response mechanism in fish. Extending the cap and riser lip by 1.5 times the height of the opening has been shown to result in a more uniform entrance velocity, and improves the ability of fish to react and avoid the intake (Figure 6.7). However, as with all intake configurations, there are many design issues that must be considered, and the performance of a velocity cap may vary in still water versus areas subject to tidal cross-flows. Virtually all velocity cap intakes require some on-shore screening system, usually a Traveling Water Screen or Rotating Drum Screen, to protect downstream pumps and pretreatment equipment.

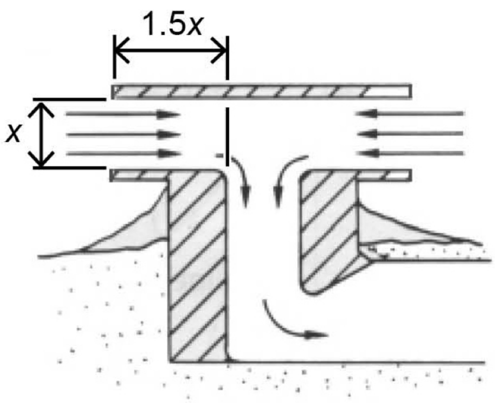

Figure 6.7. Cross-section of a Velocity Cap.

6.3.1.4 Passive Screens

A passive screen intake utilizes one or more fixed cylindrical screens (barrel screens) manufactured of trapezoidal- or triangular-shaped ‘wedge wire’ bars arranged to provide 0.5 to 3.0 mm wide slotted openings. The screens are usually oriented on a horizontal axis with the total screening area sized to maintain a through flow velocity of less than 0.15 m/s to minimize marine life and debris impingement (Figure 6.8). Passive screens are best suited for areas with ambient cross-flow currents that act to ‘self-clean’ the screen face, but may still be impacted by the attachment of organisms such as coral barnacles, or shellfish. Systems may also be equipped with an air backwash system to clear the screens when debris accumulation occurs. In most cases, the screens are located at least one screen diameter from the seabed.

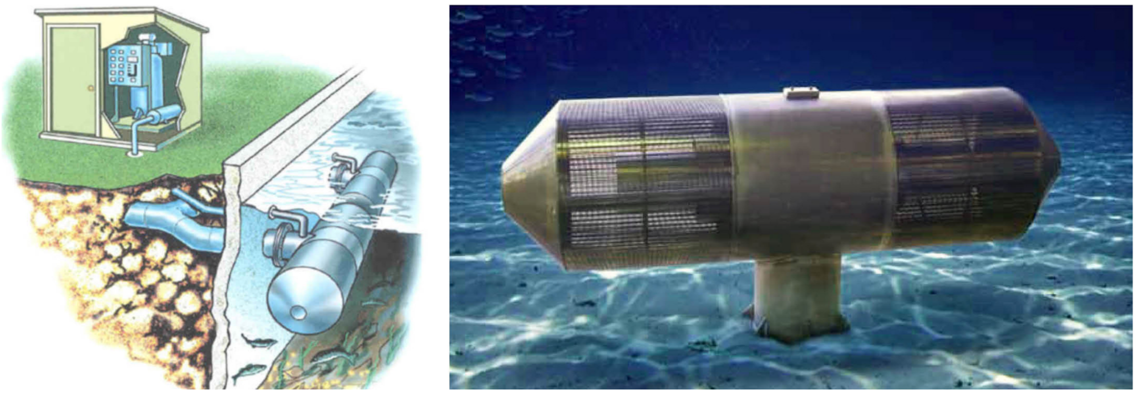

Figure 6.8. Seabed mounted passive screen left and bulkhead mounted passive screen right. Photos: Johnson screens.

Passive screens have been used on many smaller plants around the world, particularly those with nearshore intakes allowing the compressed air plants to be located on shore without undue pressure losses in air transmission pipes. Passive screens have a proven ability to reduce impingement—due to their low through-flow velocities—and entrainment—through exclusion resulting from the narrow slot openings. Tests have shown that 1 mm openings are highly effective for larval exclusion and may reduce entrainment by 80 % or more (Pankratz 2015.)

Figure 6.9. Various offshore intake heads. From top left, clockwise to bottom left, courtesy of WaterSecure, Increa, Water Corporation, and Abengoa.

6.3.1.5 Intake Head

For an offshore intake that does not employ passive screens or a velocity cap, the intake pipe terminus can be fitted with an intake head that is designed with a coarse bar or grid with 25 to 200 mm spacing. Examples are shown in Figure 6.9.

6.3.2 Surface intake strategies to minimize harmful algal bloom (HAB) impacts

Unlike subsurface intakes where seawater withdrawal takes place indirectly, beneath the seabed, open seawater intakes are directly exposed to algal blooms and other natural or anthropogenic increases in debris loading that can inundate an intake facility. Plants with open seawater intakes must therefore develop strategies—whether to change operating tactics, reduce production or shut down entirely—to deal with these inevitable occurrences.

Careful site selection is the first defense against algal blooms. Although occurrences may be sporadic and difficult to predict, historical records that address the frequency and severity of events, and the conditions that led up to the blooms should be considered and factored into siting and operational strategies.

For offshore intakes, it may be possible to choose an intake location or the location of passive screens so as to avoid areas of greatest potential algae concentrations and the entrainment of algal blooms into the intake. For example, the intake point could be located in deeper water and/or farther offshore as discussed in the next section or at a water depth where algae are less likely to accumulate. Whereas, for nearshore intakes that include long approach channels to the intake pumping station, algal concentrations approaching or within the channels could be monitored in order to guide possible management actions such as the addition of coagulation prior to ultrafiltration or modification of chlorination strategy. In situ chlorophyll or optical sensors of several types are described in Chapter 3.

Adopting a lower through-screen velocity of ≤0.15 m/s is expected to have little impact on the type of debris and suspended solids that are entrained into an intake, but this generally results in slower debris build-up, which mechanical screens find easier to handle. Even so, an algal bloom and the high suspended solids associated with it may result in debris loading conditions that are higher than the screen capacity.

A velocity cap will also not be effective in reducing the entrainment of motile e.g. dinoflagellates or non-motile algal blooms as the water flow at velocity caps is far faster than even the strongest dinoflagellates can swim (approximately 0.08 cm/s). Also, algal cells are not able to "sense" the presence of the intake to take evasive action. Thus, there will not be any appreciable decrease in the intake of planktonic organisms during a bloom. At the least, there may be limited reduction in the ingress of HABs as velocity caps limit the zone of influence of the intake to the depth level at which the velocity cap is situated, thus entraining only the algal cells present at that depth (EPA 2014).

The most common technique for mitigating entrainment is to reduce the size of the screen openings, often to 2.0 mm or less such as those of passive screens; however, although screens with fine openings generally allow the ingress of algal cells, the vast majority of which are smaller than 1 mm, they are vulnerable to sudden plugging conditions that may occur when large mats of particulates, such as those produced by algal blooms or flux of polyps (coral spawning), are encountered.

One strategy to deal with algal blooms, should they enter the intake, is to operate all traveling screens or drum screens continuously, (including those incorporated in design for redundancy purposes) during those periods when blooms are most likely to occur. This ensures that screens are kept clean and that a sudden surge of debris will not overwhelm a screen, possibly causing operational problems. For installations anticipating higher debris loads as in areas prone to algal blooms, it is usually possible to add a second spray header to facilitate debris cleaning and/or an additional lifting shelf to accommodate increased debris volumes.

Onshore or offshore intakes equipped with mechanical screens can also select screening equipment that is designed to handle higher debris loads. These options include finer screen mesh, i.e. 2.0 to 3.0 mm versus 6.0 to 9.5 mm, higher pressure spray wash systems, auxiliary lifting shelves on the wire mesh panels, and the addition of mechanical raking mechanisms on coarse screens or trash racks. For locations with a likelihood of encountering large quantities of macroalgae, such as kelp and seaweed, it may be necessary to use auxiliary toothed lifting ledges or 'kelp knives' to ensure that the revolving screens can retain the debris and convey it to the discharge trough.

Finally, chlorination at an intake may be suspended during an algal bloom as chlorination breaks down NOM into easily degradable compounds, also known as assimilable organic carbon (AOC), that serve as nutrients for the regrowth of bacteria and may actually increase growth rates (see Chapter 5 for saline AOC test). In addition, chlorine may result in the lysis of algal cells and release of AOM such as sticky transparent exopolymer particles (TEP) that can promote biofouling or release of intracellular toxins. Retaining toxins inside the algal cells will improve their removal in SWRO pretreatment processes.

6.3.3 Deep-water intakes

Intake depth is an important determinant of water quality. Increasing the intake depth and/or increasing the distance from shore and from coastal influences or discharges is promoted as a means to improve water quality and thereby reduce SWRO pretreatment requirements, filter clogging and membrane fouling. As the total water depth increases, there is typically less turbulence and less suspended solids due to wave action in the water column, and reduced risk of accidental pollution from hydrocarbon spills or leakage from shipping, which typically impact the surface layer. Conversely, strong tidal currents in some areas can create a "benthic nepheloid layer" (BNL), or layer of re-suspended sediment and detritus near the bottom. These can be large (tens of meters thick) and persistent features in areas with strong tidal currents, and could affect water quality for near-bottom intakes. Fortunately, they are easily detected using transmissometers and vertical profiling, and thus can be avoided in the intake design phase.

At greater intake depth the seawater temperature is generally more constant, which is easier for plant operation although exceptions may be found, e.g., the passage of seasonally occurring internal waves can result in rapid significant temperature changes (6˚C) at deep water intakes (Boerlage and Gordon 2011). Furthermore, the seawater may be colder at depth, which necessitates an increase in RO feedwater pressure or more membranes to meet plant capacity than is the case with warmer surface water. Thermoclines can also limit mixing and impact water quality (see La Chimba Case Study, Chapter 11 section 11.6) as can seasonal haloclines which may occur resulting in an increase in salinity of the seawater observed (Boerlage and Gordon 2011). Such temperature and salinity changes may be detected by vertical profiling prior to design.

Deep water intakes, in addition to reducing the suspended solids load, may reduce the organic load of the raw water as seawater drawn from the surface or the upper levels of the water column is where photosynthesis occurs. This is referred to as the photic zone - the depth of which varies with turbidity and season but can extend 50 – 75 m or more. More important is the mixed layer, created by winds, waves, and other surface stresses. The mixed layer is often shallower than the photic zone. Algae, zooplankton, and larvae are often most abundant in this layer and thus shallow intakes or channels may be more prone to algal blooms, as found at the Sohar SWRO desalination plant (see Sohar Case Study, Chapter 11 section 11.2). Therefore, abstracting seawater at depth (e.g. more than 15 to 20 m below sea surface and typically below the mixed layer) is a strategy often put forward to reduce the ingress of algal cells and AOM generated during a bloom into desalination plant intakes. The risk of entraining algal cysts in sediments can also be minimized as screens for deep water intakes are commonly located 1.5 to 4 m above the seabed to reduce sand and sediment entrainment; however, for multiple reasons, dense concentrations of algae (cells) and algal-related detritus including organics can be found well below the water surface as discussed below, causing problems for even deep intakes.

First, the position of the peak algal concentration may vary over time in the water column. For example, motile algal species may display diel vertical movement in the water column whereby they migrate up to the surface during the day to access sunlight for photosynthesis, and swim downwards to more nutrient rich waters late in the day and during night, descending up to 20 m (Chapter 1). This is illustrated in Figure 6.10 showing chlorophyll-*a* measurements for the toxic dinoflagellate *Gymnodinium catenatum* over five days in 20 m water depth. Higher chlorophyll is observed in the top 5 m during the day, but the center of mass of the bloom then moves to more than 17 m depth at night as the algae swim downwards in the water column. Dinoflagellates, the causative species for most toxic HABs, have the ability to swim using their flagella. Speeds of 1 m/h are typical, but *Cochlodinium polykrikoides*, the species that resulted in plant outages in 2008 and 2009 in the Gulf and Sea of Oman regions was shown to reach swimming speeds of 3 m/h (Chapter 1 Section 1.5.9).

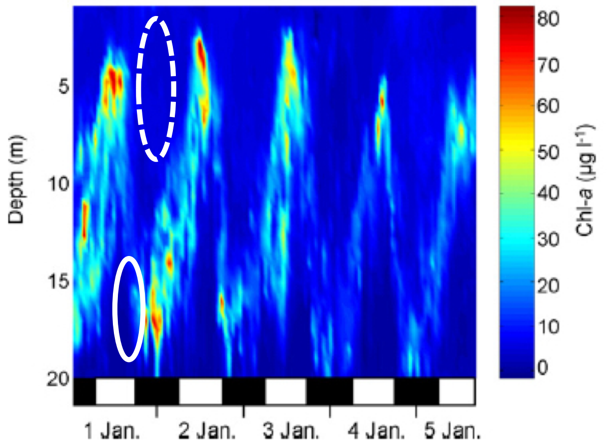

Figure 6.10. Chlorophyll-*a* (fluorescence) profiles from the Huon Estuary in Tasmania, Australia, showing diel vertical migration of the phytoplankton (dominated by *Gymnodinium catenatum*) over the 20 m depth of the water column. On the x-axis, white bars indicate the light period and black bars indicate dark. Data from CSIRO Huon Estuary Study (modified from Doblin et al. 2006). To abstract water with a lower concentration of algae, a shallow intake may benefit from operation at night (dashed line) and for a deep-water intake during the day (unbroken line).

Additionally, algal cells may move through the water column in response to nutrient supply or during various growth stages. Some diatoms, for example, can alter their buoyancy through adjustment of the composition in their vacuoles so that in favorable conditions they are found mainly on or near the surface (Moore and Villareal 1996). When growth slows due to nutrient limitation, they can adopt a "sink strategy" altering buoyance such that the weight of their siliceous cell walls helps them to settle to deeper layers of the water column where nutrients are more abundant. Finally, when algal cells age and die they lose their buoyancy and contribute to the oceanic "snow" that falls slowly to the seabed.

Hence, algal cells (and the oceanic snow) may still be entrained into intake screens depending on the intake depth and the migration or aggregation depth of the algal species that is blooming. Consequently, resultant operational issues downstream in the SWRO treatment process may manifest continuously during an algal bloom event, or only at night for deep water intakes if the bloom species displays diel vertical migration. In the latter case, the opposite would be found for shallow intakes, meaning that the maximum algal cells would be found during daylight hours and lower numbers at night. Hence, monitoring of the RO feedwater for algal cell abundance or proxies such as chlorophyll should be conducted over 24 hours, as discussed in Chapter 1.

Similarly, deep water intakes may or may not result in a reduction of the AOM fouling propensity of a surface water intake as the distribution of AOM in the water column may differ from that of the algal cells negating the effect of lower algal concentration at depth. In other words, a major bloom in surface waters can generate a large amount of organic detritus that falls into the intake zone of deep intakes. AOM comprises not only cell-bound (intracellular) organic matter which may be released through cell lysis or decay but also extracellular organic matter released into the seawater through metabolic excretion by live algal cells (Chapter 2). Organic matter generated by algal blooms varies significantly in its composition and molecular weight ranging in size from small molecular weight toxins to high molecular weight biopolymers which includes the sticky TEP and TEP precursors which may initiate and promote the formation of biofouling of SWRO membranes. This is discussed in detail in Chapter 2.

There are no known studies in the desalination industry examining the distribution of algae and AOM in the water column during an algal bloom. Some research has examined the distribution of algae, bacteria and NOM with depth, where the NOM will include compounds produced by both bacteria and algae such as TEP. TEP present in seawater can be a mixture of those produced by bacteria, algal blooms, and shellfish (Chapter 2). Therefore, results may be indicative of what may occur during an algal bloom. One study (Dehwah et al. 2015) investigating the potential for deep water intakes in the Red Sea off the coast of Saudi Arabia, reviewed the mechanisms leading to movement of TEP in the water column. Key factors include sedimentation of TEP towards the seabed from the abiotic polymerization of dissolved TEP precursors along with TEP in marine snow, whereas the buoyancy of TEP would lead to the upward movement of TEP. Dehwah et al. (2015) conducted water sampling to determine the vertical distribution of algae and NOM. Water samples were collected at the surface and at 10 m intervals to a depth of 90 m at three sites 85 km north of Jeddah and analyzed for total algae and bacteria. Total NOM in these samples was characterized by LC-OCD to provide information on fractions of NOM such as biopolymers. Particulate TEP (size greater than 0.4 μm) and colloidal TEP[2] (with size ranging between 0.1 and 0.4 μm) were also measured. Algal cell counts, total TEP (sum of particulate and colloidal TEP) and the concentration of biopolymers estimated from profiles in this study are presented in Figures 6.11 and 6.12 for the surface, 10 m and 20 m depths to correspond with surface, shallow and deep-water intakes.

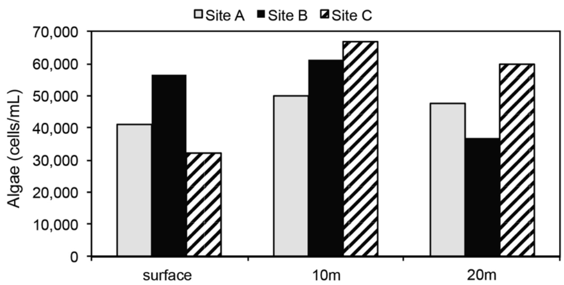

Figure 6.11. Algal cell counts in the Red Sea at the surface, at 10m and 20m water depth at three sites off the coast of Saudi Arabia (estimated from profiles presented in Dehwah et al. 2015).

Although, sampling was apparently not during an algal bloom event, the total algal concentration was relatively high, as was the bacterial concentration (ranging from 350,000 to 450,000 cells/mL at the surface). *Synechococcus*, a marine cyanobacterium (and classed as algae) ubiquitous in the ocean, accounted for more than half of the algal population, and along with the other algal species, varied between the three sites and with depth. With a distance of 3-4 km between sites A and C, the wide range in total algal cell counts between the sites was not unexpected. Algal assemblages are often patchy in nature at the surface due to random horizontal migration or drift produced by winds, shifting currents and tides (Chapter 1). Similarly, algae are not uniformly distributed in the water column as discussed above. *Synechococcus,* the dominant species in the study, is a small picoplankton genus (< 2μm in size), and about 1/3 of its species are motile and move through the water column.

[2] Colloidal TEP in the Dehwah et al. (2015) study was measured using the earlier method developed by Villacorte et al. 2009 using a 0.1 μm test membrane. The latest TEP method employs a smaller 10 kDa membrane which will capture much smaller TEP and TEP precursors present during an algal bloom (see Chapter 5 and Villacorte et al. 2015).

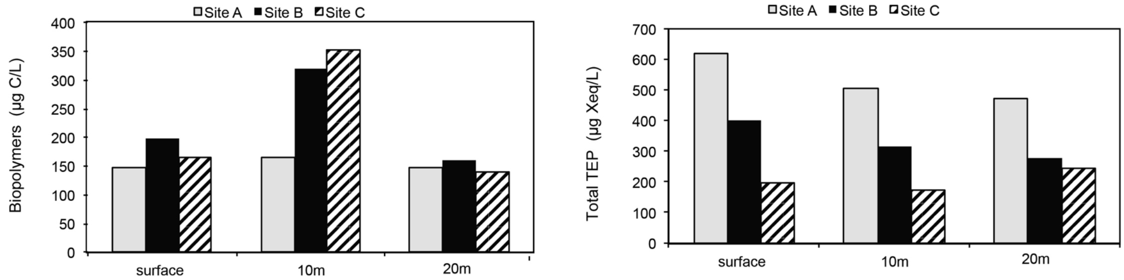

Figure 6.12. The concentration of biopolymers and total TEP in the Red Sea at the surface, at 10m and 20m water depth at three sites off the coast of Saudi Arabia (estimated from profiles presented in Dehwah et al. 2015).

As discussed by the authors of the study, no substantial reduction in algal counts was found between the surface and the first 50 m depth, with peak concentrations observed at 50 m (total algal counts of up to 900,000 cells/ml) instead of in surface layers. Algal concentration did decrease after 50 m, with the minimum concentration found at 90 m. In contrast, although total bacterial numbers also varied between sites, it declined with depth for each site with peak concentrations observed in the first 10 m.

The largest fraction of NOM at the three Red Sea sites was the low molecular weight humic acids; the concentration of biopolymers was substantially lower. The biopolymer fraction of NOM measures proteins and polysaccharides, including TEP, of both algal and bacterial origin (see Chapter 5). During an algal bloom event, spikes in biopolymers and TEP would be attributable to the algae. The biopolymer fraction of NOM and particulate and colloidal TEP varied between sites and with depth but biopolymers to a lesser extent (see Dehwah et al. 2015). The peak concentration of biopolymers was found in the top 10m layer at the three sites in the Red Sea, corresponding to the highest concentrations of bacteria in the water column. Particulate and colloidal TEP concentrations, while quite variable, showed a spike at around 40 m water depth for two of the sites, reflecting the increase in algal concentrations found 10 m deeper at 50 m water depth.

Consequently, Dehwah et al. (2015) concluded that a deep-water intake conferred no clear improvement in water quality compared to a shallow intake. Since measurements in Dehwah‘s study were not made during an algal bloom, algal counts and biopolymer concentration may show a completely different pattern with depth for different algal species during bloom events. In conclusion, there is no simple generalization favoring deep versus shallow intakes in the context of HABs. This issue needs to be determined through sustained, site-specific monitoring prior or during the design phase (Chapters 3 and 5). Furthermore, as with all desalination projects, other factors may drive the intake type, location and depth (see Sections 6.4.3 and 6.5). In the case of the Dehwah et al. (2015) study, a deep-water intake was deemed not feasible due to other factors namely the construction and operational risks for a deep water intake in that area.

6.4 SUBSURFACE INTAKE OPTIONS

Subsurface intake systems can be used to improve feedwater quality for SWRO desalination plants (Missimer 2009; Missimer et al. 2013; Rachman et al. 2015; Dehwah et al. 2015; Dehwah and Missimer 2016). There are several different types of subsurface intakes that can be designed and constructed depending on the local hydrogeology at a given site. These intake types can be subdivided into two categories, wells and galleries.

Well intakes include the following:

- conventional vertical wells (screen or open-hole completions);
- collector or Ranney wells;
- angle or slant wells; and
- horizontal wells (conventional utility type horizontal directional drilled (HDD) systems or Neodren™ systems).

Gallery types include:

- beach galleries: and
- offshore or seabed galleries.

Detailed design methods and examples of subsurface intake utilization can be found in Missimer (2009) and Missimer et al. (2013; 2015b). This includes approaches to borehole completion, screen design, exploration and testing, and general use criteria.

Historically, subsurface intake systems have been employed by small- to medium-size SWRO plants with capacities typically less than 15,000 m^3/d. There are, however, several new plants that are using subsurface intake systems that have higher capacities, and many new plants are considering the use of subsurface intake systems. In fact, in the State of California, where many SWRO projects are being investigated, a regulatory policy requires SWRO plants to use subsurface intake systems unless they can prove that any potential subsurface intake type is not technically feasible as described in Missimer (2015a) (revised California Ocean Plan).

Most subsurface intakes function in a similar manner to river bank filtration used in drinking treatment schemes in Europe and the USA, and dune infiltration practiced in the Netherlands. Such filtration systems use the natural geological properties of sediments and rocks to strain and/or biologically treat the raw water to remove organic matter, suspended solids and dissolved organic matter. With the improvement in raw water quality, pretreatment complexity and operational effort can be reduced. Almost all of the SWRO systems that use surface intake systems utilize conventional pretreatment systems or membrane treatment as shown in Schemes A and B of Figure 6.13, respectively, incorporating dissolved air flotation in areas prone to algal blooms (Scheme C). Yet, despite extensive pretreatment, plants may still encounter biofouling of the membranes. Ideally pretreatment could be reduced to fine filtration and/or simply cartridge filtration with chemical addition limited to acid or anti-scalant (Scheme D) for a well operated subsurface intake system. There is a significant reduction in operational cost accompanying the use of this option, especially when the primary process can be bypassed.

Indeed, there are a number of small to medium SWRO desalination plants in the Caribbean and Malta which require only minimal pretreatment (bag and/or sand filters; WateReuse 2011). The majority of existing SWRO desalination plants using subsurface intakes however, have an additional filtration step prior to SWRO (e.g. the Sur Plant in Oman and the Uminonakamicchi Nata Seawater Desalination[3] Plant in Japan).

Subsurface intakes are expected to attenuate feedwater quality during poor water quality events in the source seawater. Recent studies (Rachman et al. 2014; 2015; Dehwah et al. 2015; Dehwah and Missimer 2016; Dehwah et al. 2016) have investigated the performance of

[3] Commonly referred to as the Fukuoka Desalination Plant in the desalination industry

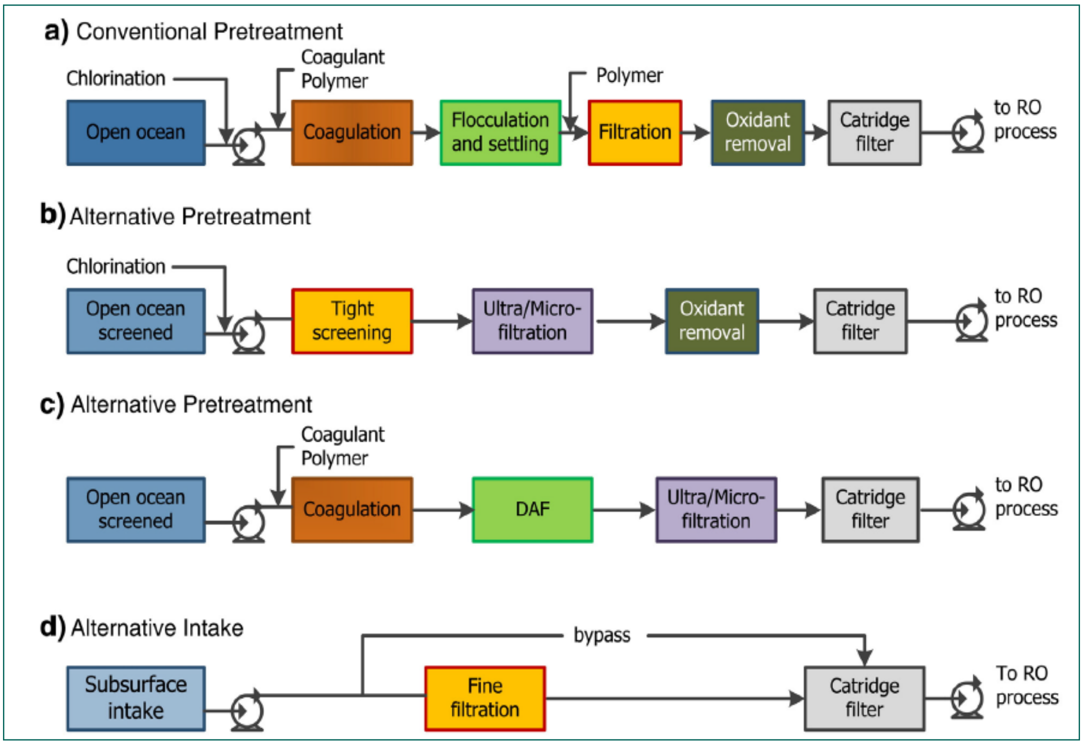

Figure 6.13. SWRO pretreatment schemes for conventional surface-water intake systems with the goal of using alternative "d" with a subsurface intake system.

subsurface intake systems for the removal of natural particulate and dissolved organic matter, such as algae, bacteria, various fractions of NOM and TEP and found them to successfully remove these completely or a large degree, as discussed further in the following sections. This is very important to SWRO plants that are located in regions subject to periodic bloom events. SWRO plants have a history of operational problems or shut-downs during severe blooms (Berktay 2011). Subsurface intake systems can continue to operate during algal blooms depending on the intake type and duration of the event. While no literature has been published for the operation of subsurface intake systems during major algal blooms, well intakes located in area with frequent blooms have been operated during events with no reported shutdown such as the Sur plant in Oman.

Subsurface intakes can be used to provide feedwater for virtually any capacity SWRO system with significant savings achieved in terms of operating costs, which can be reduced by 5 to 35% (Missimer et al. 2013).

Potential cost reductions include:

- Lower capital costs for pretreatment processes due to improved raw water quality;
- Reduction in permitting costs, especially the investigation of impingement and entrainment impacts;
- Elimination of chlorine and coagulant usage translating to a reduction in operating costs;
- Reduction in costs associated with waste disposal e.g. elimination of marine debris disposal from traveling screens and reduction in sludge generated during coagulation; and
- Continuity of supply to meet contract targets.

The capital cost of these intakes can, however, be quite high and the construction complexity can require extensive time periods to complete. In addition, subsurface intake systems require considerable planning prior to development of tender documents in order to reduce contractor bidding risk. Therefore, a full life-cycle cost analysis should be used to assess potential reductions in the cost of water to consumers before a large-capacity subsurface intake system is designed and constructed.

6.4.1 Description of intake types with example installations

Subsurface intakes are subdivided primarily into wells and galleries of varying design. There are some new hybrid designs that also use the groundwater system as a primary filter. Each of these intake types is described and an operating example is provided along with results from research. There is no good example of a beach gallery system of large size, but several small-capacity examples of a similar design are used in the Caribbean.

6.4.1.1 Conventional vertical wells

The most common subsurface intake used to supply feedwater to SWRO systems is the conventional vertical well system. Wells are constructed as close to the shoreline as possible to allow raw seawater to infiltrate through the seabed into the aquifer with flow into the pumped well (Figure 6.14). Well design and capacity are based on the local hydrogeology. Detailed design concepts, such as screen slot size, are based on the specific size distribution of the sediment and is covered in several chapters of Missimer (2009).

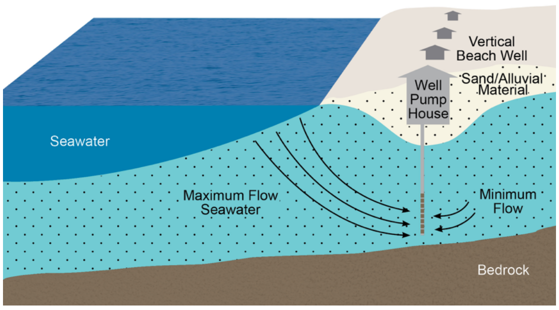

Figure 6.14. Conventional vertical wells located close to the shoreline. The produced water must come predominantly from the sea and not the landward direction. Figure: Missimer et al. (2013).

Use of conventional wells is limited to small- to medium-capacity SWRO systems unless a coastal aquifer containing a very high permeability can be used. Since vertical wells must be located very close to the surf zone on the beach, they may not be a practical intake solution in heavily populated areas because of the visual impacts on coastline or in areas where beaches are eroding; however, well systems located near the shoreline will not clog during algal bloom events due to the self-cleaning nature of the littoral zone wherein breaking waves move filtered debris laterally along the shoreline.

Currently, the largest capacity vertical well intake system in the world feeding a SWRO plant (80,200 m^3 desalinated water/d) is located in Sur, Oman. The wellfield system, located in a highly permeable aquifer, has a design capacity of roughly 160,000 m^3/d, produced from 32 wells split into three clusters, with 5 Ha dedicated to beachwells (Craig 2012). The Sur wellfield system has performed well and no clogging has been observed despite seasonally high concentrations of algae. Water quality produced by the well system is excellent – high SDI_3 up to 27 in the source seawater have been reduced to SDI_{15} of 1.4 and is consistently around 1. These low SDI_{15} results show that the intake indeed functions as pretreatment for SWRO, as after filtration through the aquifer the raw water could be directly fed to the SWRO system as it meets most RO manufacturer guarantee requirements for SDI_{15}. Nonetheless, there is pretreatment at the Sur plant, albeit limited to mono media (sand) filtration with no coagulant addition.

Rachman et al. (2014, 2015) compared the removal of algae, bacteria and NOM at the Sur SWRO plant and at three other plants operating with conventional well intake systems;

Jeddah in Saudi Arabia, Turks and Caicos in the Providenciales and Alicante in Spain. These sites were selected to represent different geographic regions and geologies; the Gulf of Oman, the Red Sea, Caribbean Sea, and Mediterranean.

Algae were almost completely eliminated in all vertical wells; this included small picoplankton species such as *Synechococcus* (see section 6.3.3) irrespective of the concentration in the associated seawater source. The Oman site in particular showed high concentrations of this species, more than 80% higher than the other three sites ranging from 113,000 to 194,310 cells/mL on the two sampling dates. In addition, over 90% and up to 99% of the bacterial population was removed by the subsurface intakes.

Rachman et al. (2014) characterized the NOM in the raw seawater at these sites and following beach well abstraction by LC-OCD into the five fractions: biopolymers, humic acids, building blocks, low molecular weight acids and low molecular weight neutrals. The concentration of particulate TEP was also determined. During algal bloom events, the concentration of biopolymers can peak. Sampling was not reported to coincide with a bloom in the study and indeed the concentration of biopolymers was relatively low. Instead, humic acid was found to be the major fraction in seawater at all sites. Removal of the NOM fractions was found to be selective with the highest removal observed for the larger molecular weight biopolymer fraction with complete to near complete removal at all sites. Substantial removal of the smaller humic acid fraction (>50%) occurred, followed by building blocks, and the light molecular weight organics. The particulate TEP removal rate was, however, variable ranging from 34% up to 92% for the different vertical wells.

In another study, the performance of a beach well in the West Mediterranean was compared to conventional and membrane SWRO pretreatment from other locations in the Mediterranean and North Sea water (see Table 6.1). LC-OCD and the Modified Fouling Index using UF membranes (MFI-UF) were employed to assess the reduction in NOM and the particulate fouling potential, respectively, by beach well filtration and the various pretreatment processes (Salinas-Rodríguez 2011; Lattemann et al. 2012). Both the MFI-UF and the earlier MFI-0.45 test, using larger pore size (0.45 μm) microfiltration membranes, were developed to measure the particulate fouling potential of RO feedwater (Chapter 5). The MFI-0.45 was applied along with AOC to monitor the clogging potential of pretreated river water to be infiltrated in artificial recharge wells due to the deposition of particles and biogrowth, respectively (Schippers 1995). MFI-0.45 and AOC values below a threshold value were expected to prevent clogging. In practice however, these parameters could not reliably predict the clogging rate of recharge wells and low values did not preclude clogging (Schippers 1995). The MFI-UF has not been trialed for predicting clogging of infiltration wells.

LC-OCD results for the beach well showed that humic acid accounted for the major fraction of NOM in the West Mediterranean seawater with biopolymers constituting only 10% of the NOM. As with Rachman et al. (2014) the highest removal after passage through the seabed was found for the larger molecular weight biopolymer fraction (70%) followed by building blocks, neutrals, and finally humic acid, with only 9% removal (Salinas-Rodríguez 2011; Lattemann et al. 2012). The performance of the beach well was superior to that of conventional and membrane pretreatment in terms of removal of biopolymers where removal was variable and ranged from 15% to 51% (see Table 6.1).

For the beach well, the MFI-UF was measured in constant pressure mode using both 30 and 10 kDa molecular weight cut off (MWCO) test membranes thus allowing smaller particles to be captured than in the SDI and MFI-0.45 tests. This is especially so when using the smaller 10 kDa membrane in the MFI-UF test, which has shown a high correlation with the concentration of smaller TEP (TEP_{10kDa}) measured with a 10 kDa test membrane (Chapter 5). Similar to the removal of organics, the beach well appears to be more efficient in removing larger particles, as the removal efficiency was 15% higher for the larger 30 kDa MWCO test membrane than for the 10 kDa membrane (35% removal). Moreover, the $MFI\text{-}UF_{10kDa}$ measured in the beach well discharge was still high (7,300 s/L^2), suggesting that particles, including smaller TEP and TEP precursors, may remain in the seawater and thus could cause downstream fouling.

The beach well achieved a higher removal of biopolymers than conventional and membrane pretreatment. In reducing the particulate fouling potential, the beach well achieved a similar reduction to coagulation and dual media filtration ($MFI\text{-}UF_{10kDa}$ removal of 40%). These results are only indicative however, as the MFI-UF for the beach well was measured in constant pressure mode while the pretreatment options were measured under constant flux mode and therefore may not be directly comparable. A comparison with $MFI\text{-}UF_{30kDa}$ cannot be made as it was not measured for all pretreatment processes.

Table 6.1. Performance of a beach well intake compared to SWRO plant pretreatment processes in removing biopolymer and humic acid fractions of NOM, and reducing the particulate fouling potential measured by the $MFI\text{-}UF_{10kDa}$ test (data from Salinas-Rodríguez 2011 and Lattemann 2012).

Seawater	Pretreatment/ Intake	Biopolymer removal (%)	Humic acid removal (%)	$MFI\text{-}UF_{10kDa}$[2] (%)
West Mediterranean	Beach well	70	9	35
North Mediterranean	In line coagulation (ferric +polymer), dual media filtration	47	30	40
East Mediterranean	Coagulation, mono media filtration	32	6	52
North West Mediterranean	Ultrafiltration (outside-in)[1]	15	1	68
North Sea Water	Coagulation (polyaluminum chloride) Ultrafiltration (inside –out)	51	1	88

[1]Outside in submerged membranes with no coagulation

[2] Although, all MFI-UF were normalized to standard reference values of temperature, pressure and area (Chapter 5), the MFI-UF for the beach well was measured in constant pressure mode (2 bar) while the pretreatment options were measured under constant flux mode (250 L/m^2h) and may not be directly comparable. Results are therefore indicative of MFI-UF removal efficiency when comparing performance of the beach well to other pretreatment options.

As expected, ultrafiltration with or without coagulation yielded the largest reduction in particulate fouling potential (68 to 88%) due to smaller membrane pore sizes than the interstices in dual media filtration or the seabed strata. The results presented in Lattemann et al. (2012) while promising in terms of biopolymer removal by beach well filtration, are limited and indicative only for the MFI-UF results and warrant further investigation.

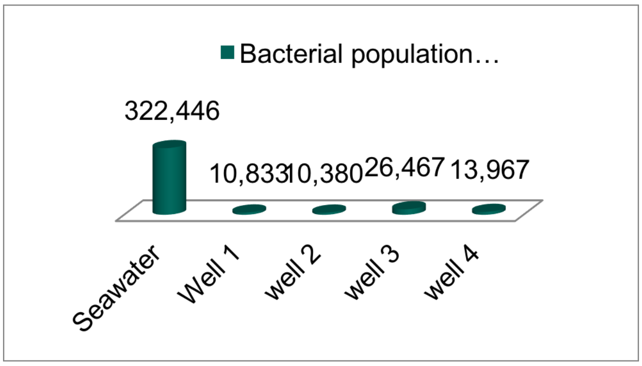

Figure 6.15. Comparison of bacterial concentrations in surface seawater and well discharges for a SWRO plant with a groundwater flow path averaging about 100 m. Figure: Rachman et al. 2014.

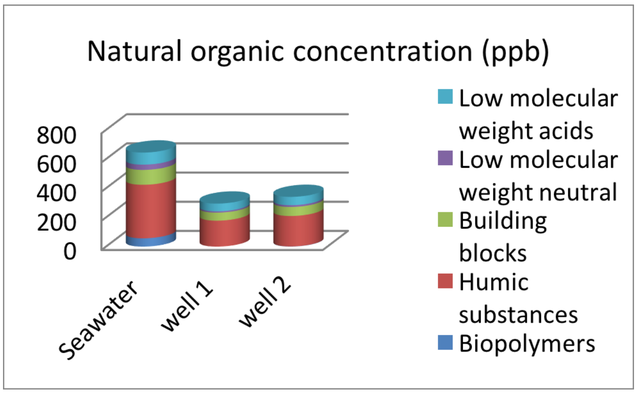

Figure 6.16. Comparison of NOM fraction for surface seawater and well discharges for a SWRO plant in Jeddah in Saudi Arabia with a groundwater flow path of about 200m. Note that the reduction of the biopolymer fraction, which contain sticky polysaccharides and proteins is nearly 100% at this location. Modified from Dehwah and Missimer 2016.

The aforementioned studies show that vertical wells reduce the biopolymer fraction of NOM (which includes TEP that can promote biofouling of membranes), in addition to bacteria, algae, and particulate matter. A typical removal percentage for well intakes is 100% for algae and over 90% for bacteria (Figure 6.15). Biopolymer removal ranged from 70% (Table 6.1) up to nearly 100% in some cases (Figure 6.16).

The degree of treatment provided within a well intake system is based upon a number of factors including the flow path length time, the type of geological media, the hydraulic retention time, the biochemical activity in the aquifer, and the local composition of the seawater. Rachman et al. (2014) found that the geological characteristics of the site and aquifer type (the Oman and the Turks and Caicos aquifers are limestone, while the Jeddah system is siliciclastic) did not have a direct correlation to the removal rate of the organic substances. Instead the flow path length and hydraulic retention time had a greater impact on organic matter removal efficiency compared to the geology of the aquifer or specifically, lithology. Therefore, a careful balance must be achieved wherein the wells are located close enough to the shoreline to have most of the recharge from the sea, but sufficiently far away to remove a significant percentage of the organic matter.

6.4.1.2 Collector or Ranney Wells

A collector well (Ranney well) is a specialized well type with a high unit production capacity compared to most wells. It contains a central caisson with a diameter ranging from ~ 2 to 4 m and a series of horizontal laterals to collect water from the penetrated aquifer (Figure 6.17). Collector wells are commonly used to tap gravel units within aquifers underlying river systems in the Midwest region of the United States and other geographic areas. These wells can have capacities of over 51,400 m^3/d (Missimer 1997; Missimer et al. 2013. There are a

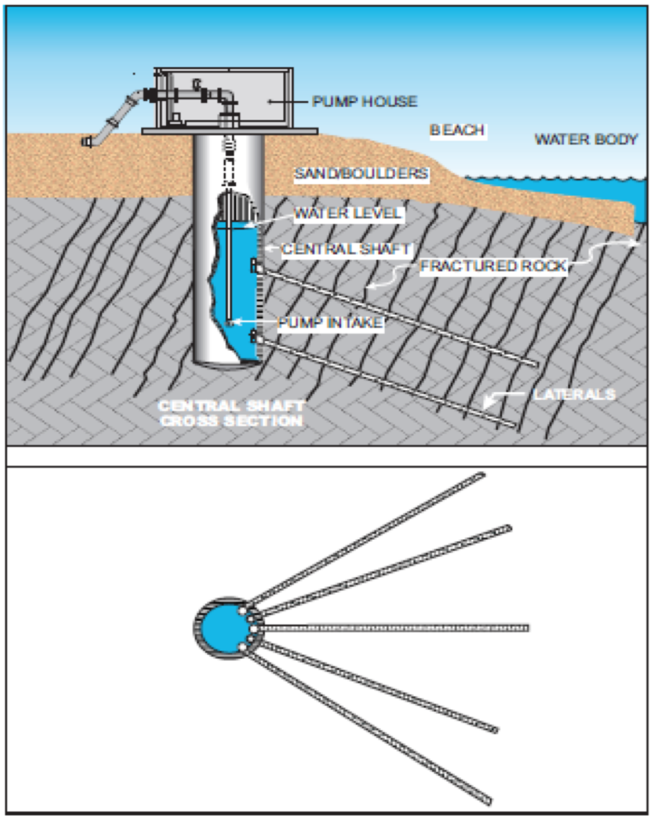

Figure 6.17. Ranney or collector well used to obtain raw water from a fractured rock aquifer hydraulic connected to the sea. Images: Missimer 2009.

few examples of the use of collector wells for SWRO intakes; the largest capacity system being the PEMEX Salina Cruz refinery in Mexico which has three wells with a capacity of 15,000 m^3/d each (Voutchkov 2005).

6.4.1.3 Angle wells

Angle or slant well design and construction is relatively new and is being applied to SWRO plants under design in California (Williams 2015). An angle well is drilled from a location on the beach at an angle so that the screen section of the well is located fully beneath the seabed and seaward of the freshwater/seawater interface (Figure 6.18). While no large-scale slant well intake system is currently operational, a detailed test program over an extended period of time (21 months) has been completed for the Dana Point SWRO plant at Doheny Beach in California (MWDOC 2014). The full-scale wells are expected to have an installed capacity of 113,600 m^3/d (Williams 2015). The filtered water was reported to have a very low SDI and turbidity over the test period (MWDOC 2014). However, the slant well began to draw anoxic water enriched in dissolved iron and manganese which may be challenging for SWRO operation. Aeration of the water during pretreatment could lead to oxidation of the iron and manganese into iron hydroxide and manganese dioxide which may result in fouling of the RO membranes if not removed.

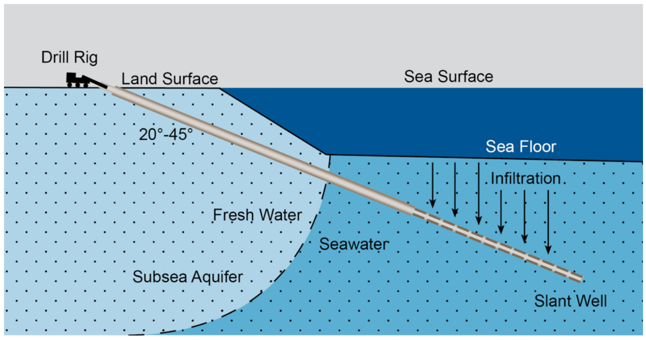

Figure 6.18. Angle well-constructed beneath the seabed and seaward of the freshwater/seawater interface. Figure: Missimer et al. 2013.

6.4.1.4 Horizontal wells (HDD)

Horizontal drilling and micro-tunneling, used to install pipes into the ground with minimal surface disruption, are mature technologies employed in the utility field for over 50 years. Horizontal wells have been designed and constructed in the petroleum field for many years and have also have been used in remediation of groundwater contamination (Delhomme et al. 2005). These early wells have been constructed using conventional drilling technologies and are completed with screens or screens covered with a geofabric.

A newer development of the horizontal technology involves the use of Neodren™ systems (Peters et al. 2007). Construction of this system occurs using a horizontal drilled hole that emerges on the seafloor (Figure 6.19). The well casing with the attached, patented screened assembly is then pulled back through the mudded borehole and set in place. There are several operating examples, the largest of which is Alicante, Spain (Peters et al. 2007). Unfortunately, there is incorrect capacity data contained in the literature regarding the Alicante system which was corrected in Rachman et al. (2014). The current capacity is difficult to assess based upon the combined use of horizontal wells and a water tunnel. It was reported to have a capacity of about 65,000 m^3/d which is now likely to be about 25,000 m^3/d or less.

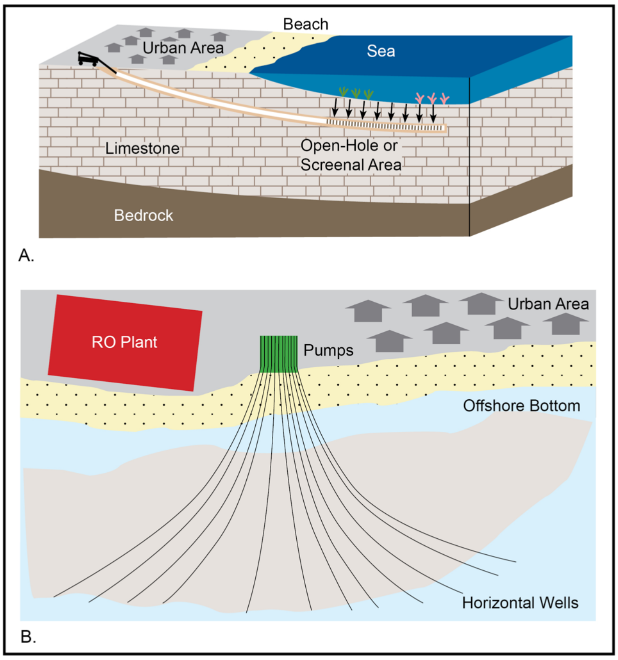

Figure 6.19. Horizontal well drilled beneath the seabed (A). Note that many wells can be constructed from a common pad which saves considerable effort and cost in siting them (B). Figures: Missimer et al. 2013.

Rachman et al. (2104) investigated the removal of NOM, algae and bacteria by the Neodren™ system in Alicante and found a breakthrough of algae, low removal of bacteria (41%) coupled to a higher concentration of the NOM fractions as compared to the source seawater. The poorer-than-expected performance of the Neodren™ system may be a result of the direct inflow of seawater into one or more of the drains or into vertical karst conduits that connect the natural seawater to the drain screens (Rachman et al. 2014).

Large-scale application of horizontal well technology has not yet been developed. While the concept is attractive, there are issues with regard to methods that would have to be employed to maintain the screens. Cleaning and repacking of the gravel could be very difficult based on the distance from the shoreline. The operation of HDD systems have not been documented during algal blooms.

6.4.1.5 Beach gallery systems

Gallery intakes use the concept of slow sand filtration by creation of an engineered filter that can be located on the beach, near or above the high tide line, within the intertidal zone of the beach, or in the seabed.

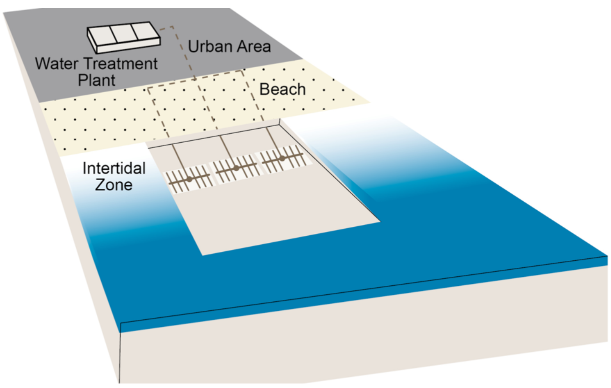

Figure 6.20. Beach gallery intake system directly beneath the intertidal or surf zone. Figure: Missimer et al. 2013.

Beach gallery intake systems involve the placement of an engineered filter beneath the littoral zone of a natural beach (Figure 6.20). Unlike slow sand filters which operate under gravity, beach gallery filters are pumped using a series of collector screens underlying the gravel and sand to abstract seawater filtrate through the filter. Yet, they act like slow sand filters due to their low filtration rates (Maliva and Missimer 2010). Slow sand filtration improves seawater quality by removing particulate matter by straining and organic matter by biological treatment. With low filtration rates and corresponding higher retention time in the filter, the assimilation of organic compounds tends to improve. The advantage of the beach gallery system is that the wave action occurring above the filter tends to keep it clean as particulate matter is removed within the upper sand part of the filter which means the system is essentially self-cleaning.

This type of intake can be used on a moderate energy beach with typical wave heights being between 0.5 and 1 m. A beach gallery intake system was recently explored for technical feasibility at Huntington Beach, California. It was found not to be technically feasible due to extreme rates of beach erosion and great complexity in construction. The construction in this high-energy environment would require use of a tram system and would take 5 to 8 years to complete based upon seasonal prohibitions on beach construction (Bittner et al. (2015).

At present, there are no large-capacity operating examples of a beach gallery intake system; however, a trench intake system was operating in a similar manner on Useppa Island, Florida for more than 20 years for supply to a SWRO plant with a capacity of approximately 400 m^3/d. It is uncertain whether the trench or gallery system is still being used.

6.4.1.6 Offshore or seabed gallery systems

The seabed gallery or infiltration gallery is another intake type that can be used to supply nearly any capacity desired (Missimer 2009). It consists of an engineered filter constructed in the seabed offshore. In concept, it is similar to a beach gallery and operates similarly to a slow sand filter, but it requires pumping and cannot operate under a gravity condition.

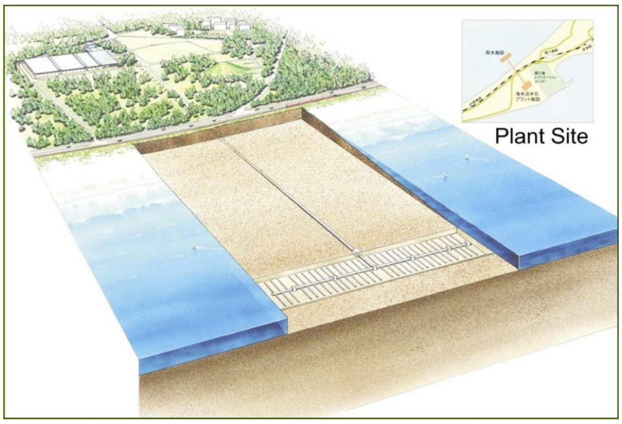

Figure 6.21. Seabed gallery system as constructed at Fukuoka, Japan. Image: Fukuoka District Waterworks Agency, 2015.

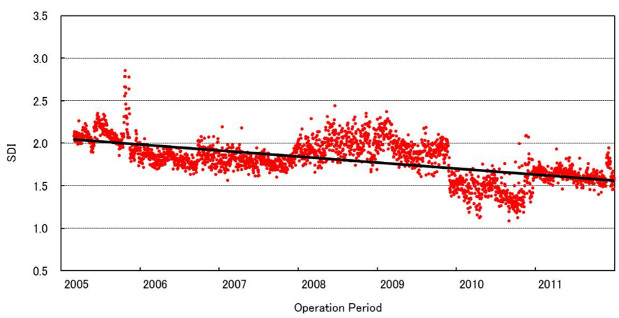

Figure 6.22. SDI_{15} data collected from the seabed gallery intake at the Fukuoka SWRO plant. Figure: Missimer et al. 2013.

The largest capacity seabed gallery system operating today is located at Fukouka, Japan and supplies the Uminonakamicchi Nata Seawater Desalination Plant (Figure 6.21). It has a capacity of 103,000 m^3/d and has operated with minimal maintenance for over nearly 10 years achieving an SDI_{15} consistently below 2.5 in the filtered water (Shimokawa 2012; Figure 6.22).

To investigate performance of a seabed gallery, the Long Beach Water Department designed and operated a demonstration surf zone infiltration gallery referred to as under ocean floor intake system. The gallery comprised an excavated pit filled with engineered sand. Figure 6.23 shows filling of the gallery using a temporary cofferdam and the filtration area of the infiltration gallery. Filtered water was collected through a series of perforated laterals

Figure 6.23. Filtration area of under-ocean floor seawater intake (left hand side) and filling of infiltration gallery with engineered sand during construction using a temporary cofferdam (right hand side) at Long Beach (Zhang et al. 2011). Reprinted with permission from American Water Works Association.

(15 cm diameter V-wires with 0.13 cm slot openings) along the pit's bottom (Allen et al. 2011). Infiltration rates varied between 2.9 and 8.8 m/d (Allen et al. 2009; Zhang et al. 2011). The seabed gallery filtrate then passed through 100 µm or 5 µm cartridge filters. Three algal bloom events occurred during operation of the seabed gallery that increased total organic carbon (measurements available for only two of the events) and turbidity in the source seawater (Zhang et al. 2011) as shown in Figs 6.24 and 6.25, respectively. While infiltration through the seabed gallery appeared to attenuate total organic matter organic carbon (TOC) during these events, turbidity was not always reduced.

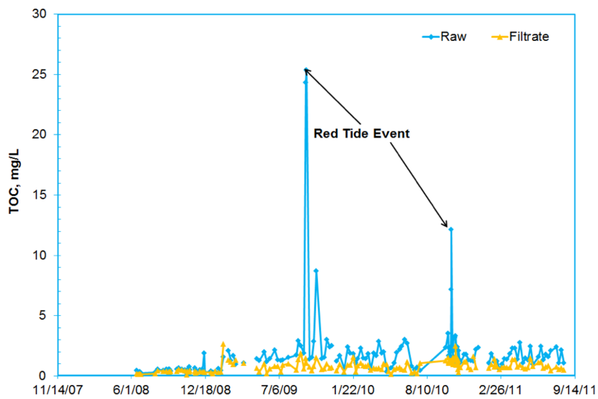

Figure 6.24. TOC in raw seawater and filtrate from the Long Beach under-ocean floor seawater intake (Zhang et al. 2011). Reprinted with permission from American Water Works Association.

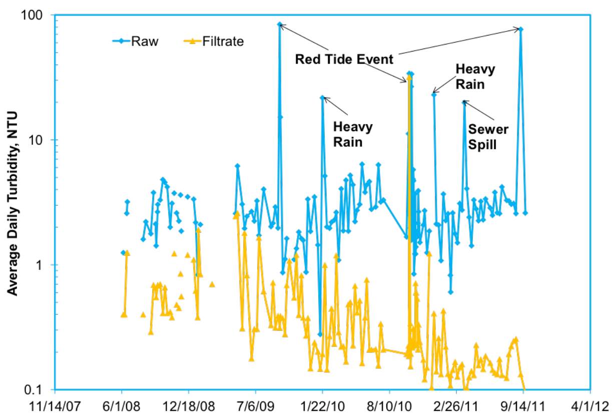

Figure 6.25. Turbidity in raw seawater and filtrate from the Long Beach under ocean floor seawater intake (Zhang et al. 2011). Reprinted with permission from American Water Works Association.

In the previously mentioned study of Salinas-Rodríguez (2011), NOM fractions in the source seawater and filtrate from the Long Beach demonstration gallery were determined by LC-OCD. Although sampling was not reported to coincide with an algal bloom event, Salinas-Rodríguez (2011) found significant removal of biopolymers from the source water (75%) by the seabed gallery and close to 20% removal of humic acids (19%). Cartridge filtration removed biopolymers by a further 13%. This may be due to a combination of adsorption onto the cartridge filters, as biopolymers are noted for being very sticky, followed by degradation by bacteria in the cartridge filters as the filtrate to the cartridge filters was not chlorinated. Indeed, the lifetime of the downstream cartridge filters was reported to be reduced to a week using the seabed filtrate due to biofouling of the cartridge filters (Carollo 2016). Iron and manganese fouling of the cartridge filters also occurred (Zhang et al. 2012). Water quality testing over time at the demonstration facility showed that it would not consistently meet typical SDI_{15} and turbidity membrane guarantee requirements without further pretreatment (Carollo 2016). This may be attributable to the fact that the sides of the seabed filter were not sealed (T. Tseng, personal communication, 2017). Therefore, the engineered sand interfaced directly with the native beach sand which allows native pore water, sediment, dissolved iron and manganese to enter the gallery. Infiltration into the filter was indeed observed from the sides as well as the top and bottom, albeit at different rates (T. Tseng, personal communication, 2017). Sealing the sides and bottom of the filter would prevent this as then the only water entering the system would have been seawater from the top where typically the concentrations of iron and manganese in well oxygenated seawater are low.

A considerable amount of new research has been conducted on design and construction of seabed galleries (Missimer et al. 2015b; Dehwah and Missimer 2017). Recent research investigated the effectiveness of the active layer of a gallery intake system in improving seawater quality in long term bench scale column experiments for two different media (Dehwah and Missimer 2017). Silica and carbonate sand were tested in 1 m columns to evaluate the removal of algae, bacteria, NOM and TEP over 620 days. The infiltration rate was fixed at 5 m/d. The columns required several months to reach an equilibrium state, after which there was a significant improvement in seawater quality. Nearly all of the algae, 87% of the bacteria, 59% of the biopolymers, 57% of the particulate TEP and 32% of the colloidal TEP were removed in the silica sand column in the last 330 days of operation. Within the carbonate sand column in the same time period, removal of biopolymers, particulate and colloidal TEP was higher at 75%, 66%, and 36% respectively but removal of bacteria was lower at 74%. Although the bench scale test simulated a type of slow sand filtration, it was found that a "*schmutzdecke*"[4] did not form at the column surface and the columns did not clog internally which is attributed to biochemical degradation of the organic materials.

6.4.1.7 New subsurface intake designs

Recently, new types of subsurface intake systems have been designed and constructed to achieve a degree of pretreatment. Two systems, the water tunnel and the karst pit, are interesting examples of hybrid subsurface intake systems.

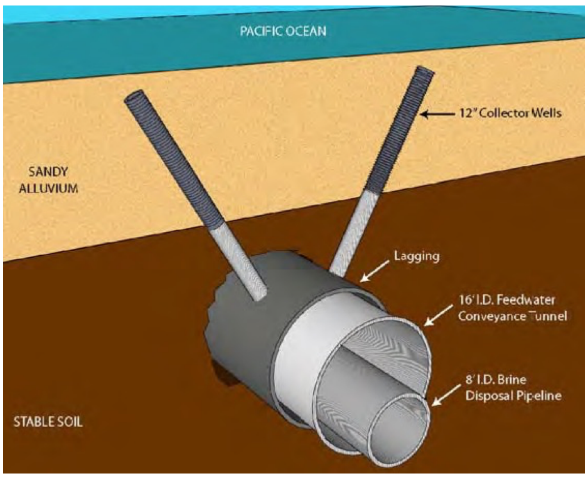

Figure 6.26. Tunnel intake example. Photo: RBF Consulting, 2009.

The water tunnel system consists of a horizontal tunnel that ranges from 2 to 4 m in diameter and contains a series of vertical collector screens that protrude into the roof of the tunnel (Figure 6.26). This general concept was originally developed for a freshwater collection system in Louisville wherein it was used to obtain water from a shallow gravel unit lying beneath a river (Missimer 2009). An example of the system in in place at Alicante, Spain where it is used as part of the intake system for a SWRO plant and has a capacity of 50,000 m^3/d (Rachman et al. 2014; 2015). The tunnel system as described by Rachman et al. (2014) comprises a 3.14 m diameter

[4] thin biologically active top layer in slow sand filtration

tunnel, 1 km in length, oriented parallel to the shoreline located 14 m below sea level. The water intake into the tunnel is provided by 104 pipe laterals, constructed perpendicular to tunnel axis. The performance of the tunnel in removing particulate matter, algae, bacteria, and NOM was compared to conventional vertical wells, including one at the same site in Alicante in the study by Rachman et al. (2015: see discussion in Section 6.3.1.1). As with the vertical wells, the tunnel system was found to be effective in virtually removing all algal cells. The tunnel system also removed 71% of the bacteria and a significant amount of the organic fractions of NOM (e.g. 90% of biopolymers) and TEP (84% and 55% removal of particulate and colloidal TEP, respectively). Overall, the vertical wells gave higher removal of NOM. The lower removal by the tunnel system is most likely caused by a shorter flowpath from the seabed to the tunnel which provides less hydraulic retention time for organic carbon removal (Rachman et al. 2015). The transport distance of seawater to the tunnel intake system was much shorter compared to the transport distance from the sea to the vertical wells.

The karst pit concept was described by Pankratz (2015) for an intake system used in Curaçao (Figure 6.27). The intake consists of a surface excavation located about 100 m from the shoreline. Feedwater is pumped from the excavation which forces seawater to filter through the limestone from the sea into the excavated area. The walls of the excavation contain prefabricated concrete that contain perforations. The intake has a capacity of 52,000 m^3/d to meet the requirements of a 26,000 m^3/d SWRO plant.

Figure 6.27. Karst pit intake system. Photo: T. Pankratz.

6.4.2 Subsurface intake performance for algae and NOM removal

In general, subsurface intake systems are very effective at removing algae, a considerable percentage of bacteria, and some percentage of the biopolymer fraction of NOM, particulate and colloidal TEP (Rachman et al. 2014, 2015; Dehwah et al. 2015, 2016; Dehweh and Missimer 2016; 2017). In the Rachman et al. (2014) study, investigating the performance of conventional vertical wells, the tunnel and horizontal well intake systems at Alicante, all of the subsurface intake types tested were effective with the exception of the horizontal well intake system at Alicante, where there was breakthrough of algae into some of the wells. The well intake systems showed 100% removal of algae during transport through various types of coastal aquifers from the sea to the wells. A comparison of three intake types at Alicante, Spain, showed that the vertical well system removed greater amounts of bacteria, NOM, organic fractions, and TEP in comparison to the tunnel and horizontal drain systems. The aquifer feeding the wells and tunnel did remove all of the algae, but only 70% of the bacteria were removed in flow to the tunnel, and some biopolymers entered the intake as 10% were not removed (Rachman et al. 2014).

The well system located at Sur, Oman, was particularly effective at removing algae. This site is significant in that the Sea of Arabia has a high frequency of algal blooms which have spilled into the Gulf with adverse operational impacts that have affected several SWRO plants (Berktay 2011). During the operational period when the well intakes were being used, it is likely that some type of algal bloom occurred; however, there is no specific documentation concerning the algal concentrations occurring and any changes in the algae occurrence within the production wells during a bloom. Based on the distances of the wells from the sea ranging from 30 to 250 m and the flow path length through the aquifer, it is highly unlikely that algae could actually enter the production wells, even during a bloom. In addition, the removal of algae, biopolymers and TEP should reduce the biofouling potential of a feedwater during a bloom event.

A considerable amount of research has been completed recently on the development of gallery intake systems that can be used to produce a sufficient supply of feedwater even for the largest SWRO plants (Missimer et al. 2015a; Dehwah and Missimer 2017). The seabed gallery, in particular, appears to have the greatest potential number of applications. This system operates similarly to a slow sand filter in which the seawater is filtered through an engineering system with a hydraulic retention time of 6 to 8 hours. Experiments conducted on the use of a slow sand filter as pretreatment for SWRO plants showed no penetration of algae through the system and a very high rate of kainic acid removal which was used as an algal toxin surrogate (Desormeaux et al. 2009). This research can be used as a proxy for the operation of a seabed gallery system and will likely function in the same manner, although no seabed gallery system has been operated throughout a major, high-biomass algal bloom, so the need for additional on-land pretreatment of the gallery effluent is still open to question.

The processes of particulate and dissolved organics removal are similar to those occurring during slow sand filtration with the exception that no biologically active "*schmutzdecke*" layer forms during filtration at the sediment-water interface i.e. on the top surface of the filter (Dehwah and Missimer 2017). Instead, all of the particulate organics and suspended sediments are strained during transport and accumulate within the aquifer or upper 1 m of a seabed filter. No reported clogging has been reported despite operation of wells and seabed filters for up to 30 years (in the case of wells). Therefore, within groundwater systems, it is assumed that biochemical processes are active in reducing particulate organics into dissolved forms that move through the aquifer. This is suggested in some recently collected data by increases in the concentration of light molecular weight neutrals in the well discharges (Dehwah and Missimer 2017). Within seabed filters, the upper layer of the filter undergoes bioturbation whereby organisms that derive nutrition from the sediments such as polychaete worms and some molluscs assimilate organic material and small particulates, leaving beneath rigid fecal pellets that act hydraulically similar to sand grains. The deposit feeders act to prevent the building of a biological clogging layer at the sediment–water interface. Removal of biopolymers and other fractions of NOM are likely caused by a variety of physical (straining and adsorption) and biochemical breakdown processes (bacterial and geochemical). Additional research is being conducted on these mechanisms in column and larger scale experiments.

6.4.3 Planning of desalination plants with subsurface intakes

While subsurface intakes offer many advantages for SWRO plants in areas prone to blooms they are not always feasible. Use of a particular type of subsurface intake is dependent on the hydrogeologic and marine conditions of a specific site. In addition, local infrastructure also plays an important role in the choice of intake type that can be used (e.g., availability of specialized construction equipment, electric power availability to pipe and pump the raw seawater). Since the use of subsurface intakes is related to the site-specific conditions, the specification of an intake type requires planning and a certain level of pre-tender field investigation.

It is prudent to perform feasibility level investigations of the coastal area in regions that are planning use of a subsurface intake system to supply SWRO plants. Dehwah et al. (2014)

developed a coastal geomorphological mapping system that can be used for screening purposes to assess which types of intakes can be used. The method considers both field conditions and the capacity of the plant being considered for the Red Sea shoreline of Saudi Arabia. The factors considered and areas mapped are shown in Table 6.2. The relationship between the mapped coastline and the feasibility of using a specific intake type are given in Table 6.3.

Table 6.2. Geomorphological classifications of the Red Sea coastline (Dehwah et al. 2014).

A. Sandy Beaches

A1 - Sandy beach with corresponding nearshore sand or slightly muddy sand, coral reef complex offshore

A2 - Sandy beaches, restricted, with no reef

A3 - Offshore island with nearshore sandy sediments and reef

B. Rocky shorelines

B1 - Limestone rocky shoreline with corresponding nearshore sand, and offshore coral reef complex

B2 - Limestone rocky shoreline with nearshore muddy sediments

B3 - Limestone rocky shoreline, nearshore deep water, no reef

B4 - Rocky headland with offshore rocky bottom, no reef

B5 - Rocky shoreline, wadi[1] sediments nearshore, offshore reef

C. Wadi [1]intersections

C1 - Wadi sediments (boulders, pebble, and gravel)at shoreline, variable sand, gravel and mud offshore with no reef

C2 - Wadi shoreline sediments, nearshore marine hard ground, minor nearshore sand , coral reef offshore

D. Sabkha[2], lagoons, and mangrove

D1 - Coastal sabkha shoreline and nearshore muddy sediments

D2 - Muddy shoreline with lagoonal muddy sediments, nearshore sand and offshore reef complex

D3 - Muddy shoreline /lagoon/ supra-tidal sabkha with no reef complex

D4 - Mangrove shoreline with nearshore muddy sediments

E. Others

E1 - Shoreline reef complex dropping to deep water in the nearshore off-reef area

E2 - Artificial channels or urban shoreline with artificially filled nearshore dropping to deep water nearshore

E3 - Natural channel

[1]Wadi- an ephemeral stream that flows only during flood conditions in arid regions

[2]Sabkha -a supra-tidal to intertidal area wherein seawater is trapped during storms or high-tide events and the trapped water evaporates to produce hypersaline conditions (salinity often over 250,000 mg/L), commonly with the precipitation of evaporite minerals occurring on the Sabhka plain.

Table 6.3. Correlation between coastal environment and feasibility[1] of using various subsurface intakes along the Red Sea coastline (from Dehwah et al. 2014).

Intake Type Well / Gallery	Subsurface Intake System					
	Well System				Gallery system	
Environments	Vertical	Horizontal	Radial (collector)	Angle	Beach Gallery	Seabed Gallery
A. Sandy Beaches						
A1	1(b)[2]	3	2(b)	2(b)	1(d)	1(d)
A2	1(a)	3	2(b)	2(a)	4	1(c)
A3	1(a)	3	2(b)	2(b)	1(d)	1(d)
B. Rocky shorelines						
B1	1(b)	3	1(b)	1(c)	1(c)	1(d)
B2	4	4	4	4	4	2(c)
B3	4	4	4	3	4	4
B4	4	4	4	4	4	4
B5	1(a)	3	2(b)	2(a)	2(c)	2(c)
C. Wadi intersections						
C1	4	4	4	4	4	4
C2	1(b)	3	2(c)	2(b)	2(c)	2(c)
D. Sabkha, lagoons, and mangrove						
D1	4	4	4	4	4	4
D2	4	4	4	4	4	4
D3	4	4	4	4	4	4
D4	4	4	4	4	4	4
E. Others						
E1	4	4	4	4	4	4
E2	4	4	4	4	4	4
E3	4	4	4	4	4	4

[1]Feasibility factor: 1=Excellent, 2=Possible 3=Questionable, 4=Not feasible

[2]Estimated Capacity (m^3/d): a. Capacity <20,000, b. 20,000-50,000, c. 50,000-100,000, d. Any capacity

Mapping of the Red Sea coastline of Saudi Arabia in the study of Dehwah et al. (2013) showed that the most favorable environments for use of subsurface intakes are;

- sandy beaches containing a low percentage of mud;
- limestone rocky shorelines with corresponding nearshore sand; and
- wadi sediments with low mud content.

Seabed galleries were found to be the preferred subsurface intake type for large-capacity desalination plants based on the geology. Conventional wells or horizontal wells could be used at shorelines containing limestone cliffs and reefs, but the relatively small thickness of these deposits is a limitation on potential system capacity. Nearshore or coastal wadi sediments not associated with a channel can also be used to develop low-capacity well intake systems. Construction of subsurface intakes in environments where there is a high mud concentration in the sediments and no water circulation (sabkha, lagoons, and mangrove) is not desirable due to the potential for clogging of the filter. The high organic content and high evaporation rate produce additional unfavorable conditions. All of the restricted and nearshore muddy shorelines or mangrove coasts are not feasible for development of subsurface intakes (D1, D2, D3 and D4).

This type of method can be applied to any coastal region. Feasibility will require additional field work and preliminary engineering design with an economic analysis.

6.5 SITING OF DESALINATION SEAWATER INTAKES

Larger desalination plants are being built to take advantage of the economies of scale with corresponding increases in intake flow rates. In addition, desalination projects are increasingly being considered and implemented in regions where desalination was not previously considered e.g. Australia and the USA. As a result, the development of these plants face increased scrutiny by Governmental Agencies and the community, such as in California. Consequently, siting of a plant and accompanying marine infrastructure are important components of feasibility studies if the site is not already fixed.

The importance of the intake to the success of a desalination plant, particularly SWRO plants, is sometimes overlooked in the development of a desalination project. Their design and construction may represent one of the major risks to budget and schedule during the delivery phase, especially in fast track projects, while access to good water quality seawater will facilitate subsequent operation of a SWRO plant (e.g., Huntington Beach plant subsurface intake system would require 5 to 8 years to construct; Missimer et al. (2015b)). In areas prone to algal blooms, careful selection of the intake location, depth, and type can play a role in minimizing their impacts and is the first defense against algal blooms. Intake siting studies are ideally coupled to investigations to characterize seawater quality which, in addition to field water quality sampling, includes a review of historical records to identify the frequency; severity and duration of poor water quality events such as algal blooms (see Chapter 5). Source water quality is, however, only one of a myriad of factors that need to be considered in siting the intake. Other factors may drive the site selection process such as plant capacity, seabed geomorphology and ecology, local environmental and marine regulations to name but a few.

To satisfy all factors associated with the development of a new seawater intake (and brine outlet), various approaches, varying in complexity, have been developed to identify and assess the suitability of candidate seawater intake locations. Evaluation criteria can be developed for engineering, environmental, and social aspects in siting studies. Separate sets of evaluation criteria can be developed for the desalination plant and product water conveyance pipeline to those of the intake (and brine outlet) as these project components have different engineering, environment and social criteria and objectives. This also allows comparison of varying arrangements of land based sites with marine infrastructure options. Generic examples of evaluation criteria are given in Table 6.4 and are often more detailed depending on the project requirements and specific to the region.

As site selection is often a challenging process, use of a multi-disciplinary team is recommended to cover all competencies required in siting and designing desalination marine infrastructure and to develop the evaluation criteria. A specialist in HABs is recommended in areas prone to algal blooms, as these individuals can provide guidance on the likely location, frequency, and type of blooms in an area, as well as information on water circulation and transport. Various approaches can then be adopted to assess the criteria and rank sites.

Performance ratings can be assigned to the evaluation criteria ranging from - fatal flaw, poorly suitable, moderately suitable through to highly suitable. If algal blooms are a known water quality concern, considerations to minimize the ingress of cells can be built into the ratings e.g. water depth, expected bloom transport pathways, or distance from shipping activities such as ballast water exchange, which can result in the introduction of HAB and other nuisance species to an area.

Table 6.4. Generic examples of engineering –infrastructure, environmental and social evaluation criteria for siting a seawater intake.

Evaluation Criteria		**Objective**
Infrastructure	Shipping	Activities associated with shipping do not pose a risk to intake structure or impact water quality. Intake structure does not create a navigation risk.
	Interference with infrastructure	Location of intake does not adversely affect current infrastructure or preclude future developments.
	Constructability	Seabed characteristics do not influence construction.
	Distance to shore	Minimize pipe length and thereby cost. Adequate depth to avoid surface algal blooms and ensure water quality.
	Water Quality	Minimize pretreatment requirements and operational costs with high quality intake water.
	Maintenance	Minimize risk associated with marine maintenance activities.
	Dredging activities	Dredging activities do not create a risk to inlet structure or impact water quality. Inlet structure does not restrict maintenance of dredging channels.
Environmental	Marine vegetation	Destruction of marine vegetation is minimized e.g. sea grass.
	Habitat	Minimize loss or degradation of habitat of rare, vulnerable and/or endangered species.
	Conservation areas	Minimize impacts on conservation areas e.g. marine parks, reefs.
Social	Aquaculture	Minimize aquaculture impacts on commercial fishing, fish and shellfish farming etc.
	Recreation	Minimize the impact on marine recreational activities e.g. boating, swimming and fishing.

For instance, locating the seawater intake within 150 m of a shipping lane may be deemed a fatal flaw by the team. Similarly, geomorphological environments that preclude the development of a subsurface intake and only allow the option of a surface intake e.g. rocky shorelines could be rated as poorly suitable (see geomorphological mapping system in Section 6.4.3). Overall, performance ratings allow a comparative assessment of sites.

More sophisticated approaches may use a Multi Criteria Analysis (MCA) approach, whereby the relative importance of each criterion within a set of environmental, social, and engineering criteria is compared to other criteria, weighted and ranked, e.g. water quality may be ranked first amongst the engineering criteria, especially if HABs are known to be a persistent issue. Thereafter, the MCA can be linked to a Geographic Information System (GIS) modelling and analysis platform to overlay geographic data sets to represent each of the evaluation criteria. The GIS models can then be used to compile scores across all the evaluation criteria and identify areas that are more suitable for the location of a seawater intake. While these approaches aim to provide a balanced approach, some subjectivity is unavoidable by the team with some criteria not directly measureable. Instead the experience of the team is relied upon. Moreover, this technique can be limited by the accuracy and currency of data sets used coupled to the fact that not all critical aspects that determine suitability can be represented in a geographic format. Hence, results need to be verified and validated with ground-truthing of sites.

While financial criteria may not be directly included in MCA, some criteria (e.g. engineering) will require financial considerations of some aspects to weight the criteria. The final decision on siting of marine infrastructure will be based on capital expenditure estimates plus operation and maintenance expenditures at suitable sites plus constructability, permitting requirements, construction and delivery schedules.

MCA has proven useful in selecting the location, depth and type of intake for SWRO desalination plants. One such plant is the Gold Coast Desalination Plant which selected a deep water intake (20 m), 1.5 km off shore based on a MCA, considering factors such as; cost, risk, scheduled delivery window, environmental impact, community disruption, visual amenity in addition to water quality (algal blooms were known to occur). The deep water intake has been successful in providing good water quality and preventing the ingress of *Trichodesmium* blooms which are the most frequent blooms found in the area (see Gold Coast Case Study, Chapter 11).

6.6 SUMMARY

Intake channels and screens of surface intakes can be impacted by macroalgal (seaweed) blooms, with the potential loss of plant availability whereas the impact of blooms of smaller phytoplankton species at the intake is rare. Instead the microscopic algae generally pass through intake screens, impacting downstream processes in SWRO plants. There are limited opportunities to minimize the ingress of such blooms into a plant. Careful selection of the location and depth of a surface intake are important considerations in areas prone to algal blooms or the use of a subsurface intake.

Offshore velocity caps, commonly used for open-ocean intakes on large-capacity stand-alone SWRO plants, have been proven successful in decreasing marine life impingement. They will not however, directly reduce entrainment of free floating algae or motile algae as they cannot detect the change in flow direction or swim away fast enough to avoid entrainment. Velocity caps can reduce the likelihood of entrainment if they are located in an unproductive area, or one with a lower number of drifting organisms. In addition, they could limit the zone of influence of the intake to the depth level at which the velocity cap is situated, thereby entraining only the algae and other plankton present at that depth.

Deep water open ocean intakes may be successful in avoiding some bloom-forming species, but some species are motile or display diel vertical migration so that they move within the water column and are not found solely within the surface, mixed layer. In the case of diel vertical migration, a SWRO plant with a deep-water intake could change to intermittent operation during a bloom, albeit reducing production, – operating during the daylight when algae are more likely to be found at the surface. By the same reasoning, shallow intakes may benefit from operating at night. However, the distribution of AOM may not reflect the distribution of algal cells, as AOM can be extracellular and detrital in form, sinking to the seabed or rising to the surface. Once algal cells are entrained into a plant, chlorination at the intake could be suspended during a bloom event to limit the increase of AOC generated through oxidation of organic matter, lysis of algal cells and release of AOM to reduce downstream fouling and retain toxins intracellularly.

In addition to providing feedwater to a plant, subsurface intakes can act as an engineered pretreatment step, performing similar to, or surpassing conventional or membrane pretreatment to eliminate algae, a high percentage of bacteria, and a significant percentage of natural organic matter (dissolved and colloidal), in particular sticky biopolymers from the source seawater which can include AOM. Depending on the local hydrogeology and the concentration of algae and duration of an algal bloom event, it is likely that most of the subsurface intake systems would allow a SWRO plant to operate continuously during a bloom without interruption. There is however a shortage of literature of SWRO plants operating during a bloom examining the removal of algae and AOM.

Finally, intake type and location may be dictated by other project constraints such as environmental regulations, budget or schedule. Various approaches can be used to balance all competing factors in siting the intake for a SWRO desalination plant such as multi-criteria analysis wherein considerations to minimize the ingress of algal blooms could be incorporated.

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