Management of Brine Discharges to Coastal Waters Recommendations of a Science Advisory Panel

Prepared for:
The State Water Resources Control Board

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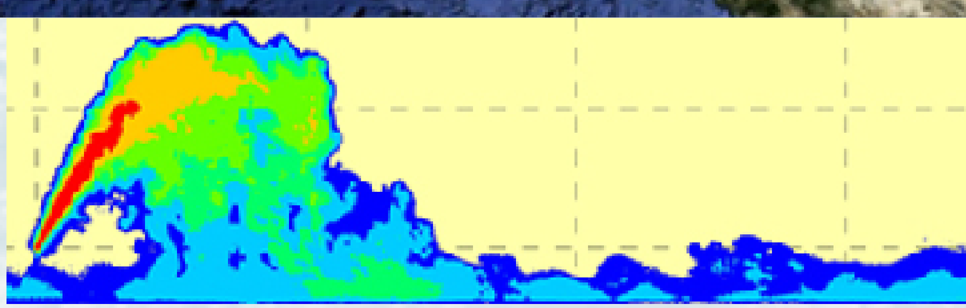

Southern California Coastal Water Research Project

Technical Report 694 - March 2012

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Recommendations of a Science Advisory Panel

Panel Members

Scott Jenkins, Jeffrey Paduan, Philip Roberts (Chair), Daniel Schlenk, and Judith Weis

submitted at the request of the

State Water Resources Control Board

by the

Southern California Coastal Water Research Project

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DISCLAIMER

This document represents a technical review and recommendations of an expert panel regarding brine discharges to coastal waters. The report is intended to describe status of knowledge, identify methods, an propose a revised framework for regulation and monitoring. The recommendations contained in the report represent the opinions of the Panel and are not a statement of State Water Board policy.

ACKNOWLEDGEMENTS

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EXECUTIVE SUMMARY

A panel of five experts in diverse fields related to brine disposal in the ocean was convened to advise the State Water Resources Control Board on best practices for brine disposal in support of the development of an amendment to the Ocean Plan. The brine concentrates can result from desalination of brackish groundwater, recycling domestic wastewater, and especially desalination of seawater. The potential of seawater desalination to provide potable water in the state is growing rapidly, with many plants currently proposed or in the planning stage. The state presently has no regulations on brine discharges and each plant is considered on a case-by-case basis.

The panel reviewed extensive material, including peer-reviewed journal articles, articles in the gray literature, NPDES permits that have been issued, various regulations from around the world, and results of monitoring studies, and heard presentations about experience with operating discharges.

From these reviews it is apparent that concentrate can be disposed of with minimal environmental effects if properly executed. Desirable methods of discharge include co-disposal with heated cooling water from power plants or domestic wastewater, or from a multiport diffuser if "pure" brine is released. Discharges with rapid initial dilution into areas of good flushing result in impacts that extend only a few tens of meters from the discharge. Conversely, poorly implemented disposal schemes with low initial dilution in poorly flushed areas can cause widespread alterations of community structure in seagrass, coral reef, and soft-sediment systems.

Extensive literature on the toxic effects of concentrates were reviewed. The effects (or lack thereof) of desalination concentrate vary widely, depending on the organism, site, the biotic community at the site, the nature of the concentrate, and to what degree it is dispersed. It appears that benthic infaunal communities and sea grasses are the most sensitive; some communities seem to be tolerant of effects of up to 10 psu increases, while others are affected by increases of only 2-3 psu. None of the studies reviewed indicated any impacts of elevated salinity levels less than 2-3 psu. It should be noted, however, that very few peer-reviewed studies have evaluated sublethal effects of desalination discharges either in the laboratory or in the field. It should also be noted that few studies have evaluated "worst-case" embayment scenarios and chronic impacts on demersal vertebrates, particularly those which have significant life history behaviors (i.e., reproduction, migration) driven by salinity variations. For example, embayments with limited flushing may have thresholds lower in anadromous fish such as salmonids or estuarine demersal flatfish, which undergo saltwater acclimation and significant endocrine alterations. Additional and long-term studies are needed on sublethal endpoints such as reproduction and on different types of concentrates and mixtures with antiscalants and other chemicals associated with RO.

We also reviewed regulations and standards that have been applied around the world. These range from salinity increments within 1 ppt, 5%, or absolute levels such as 40 ppt. These limits typically apply at the boundary of a mixing zone whose dimensions are of order 50 to 300 m around the discharge.

Because discharges can be designed to result in rapid initial dilution around the discharge, we recommend that they be regulated by a mixing zone approach wherein the water quality regulations are met at the mixing zone boundary. The mixing zone should encompass the near field processes, defined as those influenced hydrodynamically by the discharge itself. These processes typically

occur within a few tens of meters from the discharge, therefore we conservatively recommend that the mixing zone extend 100 m from the discharge structure in all directions and over the whole water column.

Based on the studies of effects of brine discharges we recommend an incremental salinity limit at the mixing zone boundary of no more than 5% of that occurring naturally in the waters around the discharge. Expressing the limit as a percentage increase allows for natural variability in the background waters. For most California open coastal waters this increment will be about 1.7 ppt; for a typical seawater desalination plant where the brine is concentrated by a factor of roughly two times, this corresponds to a dilution of about 20:1, which should be readily achievable. The dilution is the combination of in-pipe dilution in the case of co-discharges, and near field mixing. In addition to the salinity requirement, the discharge should meet toxicity and other requirements in the Ocean Plan at the edge of the mixing zone.

Co-discharges with power plant cooling water or domestic effluent can be positively buoyant, i.e. less dense than the receiving water. In that case, the regulatory framework of the Ocean Plan should be sufficient for protection of beneficial uses. Near field models should be re-run, however, to account for the increase in effluent density and flow rates on plume behavior.

The preferred methods of discharge are from a multiport diffuser for "raw" effluents, or co-disposal with power plant cooling water or domestic wastewater that results in significant in-pipe dilution. These discharges can be either a shoreline surface discharge (if positively buoyant) or through an existing multiport diffuser. Shoreline discharge of raw effluent is discouraged due to slow near field mixing and potentially high exposures of benthic organisms to elevated salinity.

In computing near field dilutions of negatively buoyant discharges from diffusers, conservative assumptions should be applied: that ocean currents do not increase dilution, and the seabed is flat and horizontal. To account for possible reductions in dilution in areas of poor flushing, estimates of overall flushing of the discharge site should be made to ensure that the dilution requirement at the edge of the mixing zone is still met.

No specific mathematical models are endorsed, but it is recommended that calculations be made using either tested semi-empirical equations available in the literature or by integral mathematical models based on entrainment assumptions. Mathematical models should be validated, and attention should be made to special conditions that occur with typical negatively buoyant discharges such as reduction in dilution due to Coanda effects and jet merging in the case of multiport diffusers.

Because of uncertainties in plume modeling and predicting the biological effects of the discharges, a field monitoring program should be used. Monitoring should include pre-discharge conditions and continue after discharge has begun to evaluate changes in the ecosystem. We recommend that the receiving water monitoring programs be based on Before-After Control-Impact (BACI) monitoring that includes multiple reference locations, samples at various distances from the discharge, and repeated sampling over time. The effluent should also be monitored for specified physical and chemical parameters.

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1. INTRODUCTION

Interest in desalination is high in California, where increasing populations and limitations to existing water supplies have spurred development of alternative sources derived from seawater desalination and reclamation of wastewater and groundwater. A few seawater desalination facilities are currently in operation in California (Table 1-1), but proposals for over 20 additional coastal facilities are under consideration (Cooley et al. 2006) with a potential total capacity approaching 500 mgd in 2030 (Bleninger and Jirka 2010). These include plants in Carlsbad, Camp Pendleton, Huntington Beach, Dana Point, Long Beach, El Segundo, Playa Del Rey, Oceano, Cambria, Marina, Sand City, Ocean View Plaza, Santa Cruz, Moss Landing, Montara, San Rafael, East Bay, and Crockett, with the largest of these proposed plants located in Southern California. The development and operation of these additional facilities will greatly increase the amount of desalination capacity and associated concentrate production in California.

Table 1-1. Desalination facilities located along the California coast (adapted from Cooley et al. 2006).

			Maximum capacity	
Operator/Location	Purpose	Ownership	MGD	Status
Chevron/Gaviota	Industrial processing	Private	0.4	Active
City of Morro Bay	Municipal/domestic	Public	0.6	Intermittent use
City of Santa Barbara	Municipal/domestic	Public	2.8	Decommissioned
Duke Energy/Morro Bay	Industrial processing	Private	0.4	Not known
Duke Energy/Moss Landing	Industrial processing	Private	0.5	Active
Marina Coast Water District	Municipal/domestic	Public	0.3	Temporarily idle
Monterey Bay Aquarium	Aquarium visitor use	Non-profit	0.04	Active
PG&E/Diablo Canyon	Industrial processing	Private	0.6	Not known
Santa Catalina Island	Municipal/domestic	Public	0.1	Inactive
U.S. Navy/Nicholas Island	Municipal/domestic	Government	0.02	Not known
Oil and gas companies	Platform uses	Private	0.002-0.03	Active

Various technologies are utilized to remove salts and other contaminants from water, depending of the characteristics of the source water. The most widely used method is reverse osmosis (RO), where dissolved constituents are removed by passing the water through a membrane under high pressure. In addition to the potable water, reverse osmosis produces a waste stream (concentrate) that contains elevated concentrations of salts (typically double in the case of seawater) and other dissolved constituents. At present, seawater desalination represents a minor portion of the desalination activity within California;most capacity is utilized for the treatment of brackish groundwater or wastewater (Figure 1-1).

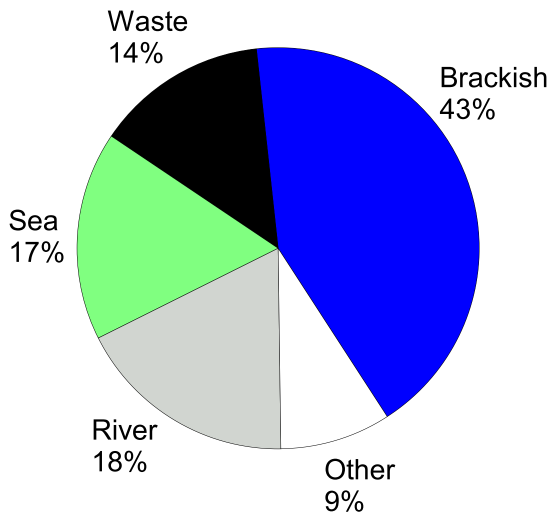

Figure 1-1. California desalination capacity by source water, January 2005 (adapted from Cooley et al. 2006).

The concentrate from desalination (often referred to as brine) varies in composition and volume depending upon the nature of the source water. This concentrate is continuously produced and must be disposed of in a manner that results in minimal environmental impact. In some cases this concentrate is discharged into coastal waters, either through a dedicated outfall or as part of a larger effluent stream from wastewater treatment or power generating facilities. The elevated salinity of the concentrate can cause it to behave differently than traditional wastewater, stormwater and cooling water plumes. When the effluent density exceeds that of the ambient seawater, the plume could settle on the ocean floor and spread as a density current, resulting in increased exposure to bottom-dwelling marine life. The elevated concentration of salts and other constituents in these discharges may result in adverse impacts to sensitive components of the ecosystem.

The regulation of discharges and protection of water quality in California is based on federal and state laws (Section 7 and Appendix A). Discharges to coastal waters must comply with water quality objectives in the California Ocean Plan, as well as regional water quality control plans. Currently, the Water Boards regulate brine discharges from these types of facilities through the issuance of National Pollutant Discharge Elimination System (NPDES) permits that contain conditions protective of aquatic life. However, the Ocean Plan does not yet have an objective for elevated salinity levels in the ocean, nor does it describe how brine discharges are to be regulated and controlled, leading to permitting uncertainty. The Ocean Plan also does not address possible impacts to marine life from intakes for desalination facilities. It is currently left to the Regional Water Boards' discretion to decide what constitutes the "best available site, design, technology, and mitigation measures feasible" for a proposed desalination facility when issuing NPDES permits for plants within their jurisdiction. However, the issues are complex and require significant staff resources and expertise to evaluate the most appropriate technology-based solution. Absent a statewide policy, permits for new desalination plants are likely to be delayed and challenged repeatedly by industrial and citizen petitioners. The planned amendment to the Ocean Plan would provide statewide consistency in controlling impacts from desalination plant intakes. In this report, we address only issues related to discharges, not intakes.

State Water Board staff are presently developing an amendment to the Ocean Plan that would address issues associated with the disposal of brine from desalination facilities and other sources. Desalination facilities and brine disposal was discussed as Issue No. 4 in the 2011-2013 Triennial Review Workplan. The issue has been identified as very high priority for the State Water Board to address. The planned amendment to address potential impacts to aquatic life from the intakes and brine discharges from desalination facilities is scheduled to be adopted by the end of 2012 and would be implemented through individual NPDES permits.

Particular questions need to be addressed in support of the Ocean Plan amendment development, including:

- How can the effects of these discharges be minimized through proper disposal strategies?
- What models should be applied in order to predict how these plumes will behave?
- What cumulative effects are there from multiple sources?
- What are appropriate monitoring strategies for these discharges?

The Southern California Coastal Water Research Project (SCCWRP) was selected by the State Water Board to convene an expert panel and develop recommendations in support of the Ocean Plan amendment. This process was coordinated and facilitated by Mr. Steve Bay (SCCWRP). A panel of experts was convened in 2011 to advise State Water Board staff regarding the above questions. Members of the panel represented expertise in physical oceanography, modeling, ecology, and toxicology (Appendix B). The panel met and received stakeholder input during 2011 and 2012 in order to assess available information, identify data gaps, and develop recommendations for the State Water Board.

This document describes the recommendations of the panel and is organized into several sections. Sections 2 through 4 provide background information on key aspects of the regulation, characteristics, and biological effects of concentrate discharges. Section 5 describes important features of different types of environments that affect the fate and effects of concentrate discharges. Subsequent sections describe the Panel's recommendations regarding the design of concentrate discharges (Section 6), revisions to the regulatory process (Section 7), plume modeling (Section 8), and monitoring (Section 9). Section 10 includes a summary of the Panel's recommendations. This report also includes several appendices, which provide additional background and technical information to support the recommendations.

A diverse collection of information was reviewed by the Panel to prepare this report, ranging from peer-reviewed scientific papers to technical memoranda concerning specific facilities. These sources often used different terms to describe similar discharge parameters and factors, such as salinity. An attempt has been made to standardize several of these terms in this report in order to minimize confusion. The terms "concentrate" and "brine" are used interchangeably when referring to the reject stream from a RO facility, regardless of the source water type. The literature and regulations use various units to describe the salinity of concentrate discharges, usually ppt (parts per thousand) or psu (practical salinity units). In this report we use both units, depending on their use in the original source, but it is noted that salinity in ppt and psu are essentially interchangeable in the context of this report.

2. REGULATORY CRITERIA

This section summarizes regulations for receiving water salinity impacts resulting from dedicated brine discharges, i.e. discharges that are not comingled with other effluent such as municipal wastewater or power plant cooling water. For these discharges, the main water quality concern is elevated salinity in the receiving waters and secondarily the discharge of various chemicals used in the treatment process. The Panel recommends that regulations for salinity be promulgated as applying at the end of a regulatory mixing zone. The mixing zone will generally encompass the near field in which rapid mixing of the concentrate and reduction in salinity occurs. The concepts of mixing zones and near field are discussed in more detail in Appendix D. The recommended salinity limits and mixing zone definitions for California discharges are presented in Sections 4 and 7.

2.1 Existing Regulatory Criteria for Salinity

A few recommendations for regulatory criteria have been proposed based on field and experimental studies of Mediterranean sea grasses, which are highly sensitive to elevated salinity (see Section 4). Sánchez-Lizaso et al. (2008) recommend salinity be less than 38.5 psu for 25% of the time and less than 40 psu (an increment of about 2 psu) for 5% of the time. And Palomar and Losada (2011) quote a Spanish Ministry of the Environment recommendation that the salinity increment be less than 2 psu for 5% of observations. An increment of 2 psu corresponds to an elevation of about 5% over background levels.

There are few actual regulations, standards, or guidelines for brine discharges around the world. Some that have been established and their compliance points for various desalination plants are summarized in Table 2-1. There is substantial variation in the specifics of the regulations, but almost all share two key elements: a salinity limit and a point of compliance expressed as a distance from the discharge. The salinity limit is usually stated as an increment of no more than 1to 4 ppt relative to ambient. However, limits are also less frequently expressed as an absolute salinity or a minimum level of dilution. The point of compliance for the salinity limit is the boundary of the mixing zone, which is usually specified in terms of a fixed distance from the discharge that ranges from 50 to 300 m.

Table 2-1. Regulations and salinity limits for selected desalination brine discharges.

Region/Authority	Salinity Limit	Compliance Point (relative to discharge)	Source
US EPA	Increment ≤ 4 ppt		
Carlsbad, CA	Absolute ≤ 40 ppt	1,000 ft	San Diego Regional Water Quality Control Board 2006
Huntington Beach, CA	Absolute ≤ 40 ppt salinity (expressed as discharge dilution ratio of 7.5:1)	1,000 ft	Santa Ana Regional Water Quality Control Board 2012
Western Australia guidelines	Increment < 5%		
Oakajee Port, Western Australia	Increment ≤ 1 ppt		The Waters of Victoria State Environment Protection Policy
Perth, Australia/Western Australia EPA	Increment ≤ 1.2 ppt at 50 m and ≤ 0.8 ppt at 1,000m	50 m and 1,000 m	Wec, 2002
Sydney, Australia	Increment ≤ 1 ppt	50-75 m	ANZECC (2000);
Gold Coast, Australia	Increment ≤ 2 ppt	120 m	GCD Alliance (2006).
Okinawa, Japan	Increment ≤ 1 ppt	Mixing zone boundary	Okinawa Bureau for Enterprises
Abu Dhabi	Increment ≤ 5%	Mixing zone boundary	Kastner (2008)
Oman	Increment ≤ 2 ppt	300 m	Sultanate of Oman (2005)

3. CHARACTERISTICS OF DESALINATION BRINE AND OTHER CONCENTRATE DISCHARGES

The concentrate produced by the reverse osmosis (RO) process contains multiple chemical constituents in addition to natural seawater components, and the amounts and types of these constituents vary as a function of the source water treated. All RO concentrates contain chemical additives necessary to maintain the treatment system. In addition, RO treatment of wastewater and groundwater will concentrate contaminants and other constituents, which may influence the toxic potential of the concentrate when discharged into the environment. Discharge regulations, monitoring, and future research should take into account differences in the chemical composition of different concentrate types.

3.1 Chemical Additives

Reverse osmosis (RO) membrane systems are widely used for the desalination of water and wastewater. The process itself is relatively simple and involves applying pressure to membranes that are essentially permeable only to water. The membranes reject more than 99.5 % of dissolved salts and suspended contaminants in the feedwater, producing a reject waste stream (concentrate) containing a 2 to 7 fold increased concentration of dissolved and suspended constituents. This can lead to fouling of the membrane and membrane element feed channels by scales, gel-like deposits of coagulated colloidal particles, and biofilms. To control this, various chemicals are continuously added to the system, depending on the feedwater characteristics. In addition, periodic cleaning and flushing of the membranes occurs, which can also alter the composition of the concentrate. Thus, concentrates are complex mixtures of many chemicals. While most studies focus on salinity as the primary cause of biological effects, many chemicals are used in the desalination process (e.g. antiscalants, biocides, etc.), some of which can be toxic.

There is uncertainty regarding the nature and concentrations of chemical additives in RO concentrate, partly because chemical formulations are often proprietary and the concentrations used vary among applications and water types. Antiscalants are aqueous solutions of blended active ingredients chosen from the families of phosphonates, and anionic organic polymers consisting of homopolymers, co-polymers and terpolymers of acrylic, maleic, and related monomers. The total active ingredients in antiscalant products vary from 1 to 40% by weight, the balance being water. Antiscalants typically used in all RO plants that treat groundwater or wastewater are dosed continuously into RO feedwaters at an average dosage of about 3 mg/L, resulting in a concentration of about 5 mg/L of active ingredients in the concentrate. Concentrated sulfuric acid is also frequently added during treatment of wastewater or groundwater. Antiscalants are typically not used in seawater desalination, although other chemicals such as chlorine are added to reduce biofouling or colloidal fouling.

The continuous use of chemical additives can result in relatively large mass loadings of chemicals in concentrate discharges (Höepner and Lattemann 2002). Additional chemical constituents in the discharge result from the concentration of materials occurring in the feedwater. Regardless of source, the discharge of concentrates with high chemical levels has the potential to impair biological communities. Monitoring of water quality around a single Florida desalination plant during the late 1960s found up to 45 kg copper discharged daily (Chesher 1971). Copper concentrations in receiving waters were often at levels above toxicity thresholds for native species (Chesher 1971).

3.2 Concentrate Types

RO treatment of wastewater or groundwater produces concentrates likely to differ in chemical composition from those produced from seawater, due to differences in the composition of the feedwater (Table 3-1). The toxicity of concentrates from groundwater or wastewater treatment should be evaluated separately from seawater concentrates. First, the ionic composition of concentrates from seawater desalination differs significantly from those derived from other feedwater types with the latter being primarily sulfate-dominated and in some cases less toxic than seawater concentrates (Schlenk et al. 2003, Lavado et al. 2012). For example, sulfate is highly regulated biologically and tends to be rapidly converted to molecular forms of sulfur that scavenge metals as well as protect against toxicity resulting from metals (i.e., oxidative stress). Second, the wastewater that is used for recycling and reverse osmosis treatment is typically secondary or tertiary-treated wastewater derived from municipal sewage treatment facilities (Grissop 2009). Consequently, this concentrate may contain excreted hormones, pharmaceuticals or personal care products. There have been few published studies that have evaluated the concentrations of pharmaceuticals and emerging contaminants within the RO concentrate. A concentration factor for pharmaceutical agents typically ranges from 3 to 4-fold (Snyder et al. 2006). While the concentrate is likely diluted upon blending with either wastewater or thermal (i.e., cooling) effluent, no studies have examined the fate of the compounds after blending with effluent, undergoing disinfection and then discharge. Given the possibility that many of these agents target sublethal biological endpoints, it is likely that effects of this effluent may have chronic impacts in addition to the short-term effects typically measured in effluent toxicity evaluations.

Table 3.1. General characteristics of concentrate from different sources. The values shown represent estimated concentrate concentrations for seawater and wastewater (estimated value= source water*100/reject rate). Actual values are shown for groundwater concentrate. Reject rates = 50% for seawater and 15% for wastewater.

Constituent	Seawater [a]				Wastewater [b]				Groundwater [c]			
	N	Min	Med	Max	N	Min	Med	Max	N	Min	Med	Max
TDS (mg/L)	9963	66400[d]	67200[d]	68200[d]	NA	NA	NA	NA	13	25	6320	70000
TOC (mg/L)	NA	NA	NA	NA	239	55	127	724	3	3.4	3.4	4.6
Ammonia-Nitrogen (mg/L)	72	0	0.02	0.46	480	153	207	294	3	0.2	0.21	1.15
Nitrate-Nitrogen (mg/L)	57	0	0.09	2.0	145	0.1	0.7	15	6	0.2	2.65	19.8
Phosphate (mg/L)	65	0.00	0.07	3.1	124	2	12	44	2	0.1	0.1	0.1
Sulfate (mg/L)	-	-	5480	-	136	1087	1630	1967	6	9	710	3400
Arsenic (mg/L)	12	2.46	2.9	3.84	458	0.01	11	45	5	0	0.01	5.1
Bromide (mg/L)	-	-	130	-	NA	NA	NA	NA	2	10	10.4	10.4
Barium (mg/L)	-	-	0.1	-	108	147	240	335	5	0.2	0.39	0.735
Calcium (mg/L)	-	-	820	-	NA	NA	NA	NA	7	53	734	960
Iron (mg/L)	-	-	<0.04	-	148	3	28133	49067	3	20	69	100
Magnesium (mg/L)	-	-	<0.02	-	NA	NA	NA	NA	3	46	63	230
Silica (mg/L)	72	0	0.29	0.93	NA	NA	NA	NA	5	54	100	160
Sodium (mg/L)	-	-	21800	-	4	2053	2357	2613	5	160	210	960

a= Seawater minimum, median and maximum values were calculated from measurements of selected California areas, except where noted. Data obtained from the Bight 2008 Offshore Water Quality Study as part of the Bight 2008 Regional Monitoring Program. Values for seawater sulfate, bromide, barium, calcium, iron, magnesium and sodium represent standard values from Millero et al., 2008.

b= Wastewater minimum, median and maximum values were calculated from measurements of large POTWs as part of SCCWRP Mass Emissions Monitoring Program and from

c= Groundwater minimum, median and maximum values were calculated from measurements of southern California facilities. Data obtained from the Groundwater Desalter and Groundwater Cleanup Treatment Plant Capacity and Discharge Data taken from the USBR Survey on Groundwater Desalters. Data quality reports provided by the South Coast Water District and the City of San Juan Capistrano.

d= Seawater values estimated from salinity measurements as mg/kg.

4. ENVIRONMENTAL IMPACTS OF SEAWATER BRINE DISCHARGE

Peer-reviewed studies on the effects of elevated salinity or brine discharge were reviewed by the Panel. From these studies, it is clear that the effects (or lack thereof) of desalination concentrate vary widely, depending on the organism, site, the biotic community at the site, the nature of the concentrate, and to what degree it is diluted and dispersed. Overall, it would appear that benthic infaunal communities and sea grasses are the most sensitive to the acute effects of concentrate discharge; some communities seem to be tolerant of effects of up to 10 psu increases, while others are affected by increases of only 2-3 psu. However, few studies have evaluated discharges to embayments, where less dispersion of the discharge may occur, and the chronic impacts on demersal vertebrates, particularly those which have significant life history behaviors (i.e., reproduction, migration) driven by salinity variations.

4.1 Biological Impacts

As described previously, effluents from desalination are not merely concentrated salts, but include a variety of chemicals that come from the reverse osmosis process, such as antiscalants and antifoulants, including chlorine and other disinfection by-products that may be toxic, as well as chemicals present in the intake water. Discharge of this mixture into the environment may have adverse effects on marine biota. Ways in which such effects can be measured include 1) laboratory tests (bioassays) of various concentrations of the effluent on different species and 2) field studies focusing on changes in the community of organisms in the receiving environment.

Laboratory and field studies have been conducted throughout the world to better characterize the risks that concentrates pose to the marine environment. Most studies that have evaluated the biological effects of elevated salinity have used concentrate predominately derived from desalination of seawater or concentrated hypertonic solutions from pre-mixed salts.

With regard to laboratory studies, Whole Effluent Toxicity Testing (WET) has primarily been used to evaluate the impacts of desalination concentrates on biota. However, most studies have focused on mortality as an endpoint. This is, of course, the most extreme effect, and the absence of death in exposed organisms does not mean that they are not seriously impaired. There is a great need to learn about sublethal effects, but there have been very few studies that examined effects of long-term exposure on sublethal parameters, such as behavior and reproduction. While the methodology for WET assessments is standardized and evaluates the complete discharge mixture from desalination, reproduction is rarely evaluated as an endpoint. Given the interactions of osmoregulatory alterations with endocrine and reproduction hormone responses, particularly in vertebrates (Avella et al. 1991, Ayson et al. 1994, McCormick 1995), this endpoint may provide more conservatism in discharge areas with little dilution (i.e., embayments).

Wide variations in study design complicate the synthesis of published effects studies. Organisms tested ranged from benthic arthropods, to echinoderms, algae, sea grasses, and mollusks (see Roberts et al. 2010a for review and table in Appendix C). Different studies evaluated different endpoints after exposing organisms for varying periods of time. Furthermore, some studies provide the absolute salinity, while others report the increase over ambient (which varies from site to site), and others refer to the percent dilution of an effluent, so it is not possible to standardize the exposures.

In this section, we review the literature on environmental effects of elevated salinities, through both laboratory and field studies.

Organisms Evaluated in Laboratory Studies:

Mollusks:

A 3 ppt salinity increase reduced size and weight of cuttlefish (*Sepia apama*) embryos, and a 6 ppt increase reduced survival (Dupavillon and Gillanders 2009). Surviving individuals at these concentrations displayed abnormal behaviors such as slow response to stimulation and reduced ink-jet defense responses.

Oyster (*Crassostrea virginica*) survival and reproduction was impaired by 60 day exposure to a 45-55 ppt concentrate, but toxicity was thought to be caused by excessive copper concentrations (Mandelli 1975). In addition to toxicological effects, the altered physicochemical characteristics of the brine appeared to enhance pathogenic fungus infection rates in the exposed oysters. Iso et al. (1994) evaluated the impact of a hypertonic solution on the survival of Japanese clams. Generally, no effects were observed below a 19 ppt increase over control salinity, but survival is the most drastic possible response, and this study did not use whole effluent.

Fish

Iso et al. (1994) evaluated the impact of a hypertonic solution (not whole effluent) on the survival and behavior of sea bream and the survival of flounder embryos and larvae. Generally, no effects were observed below a 19 psu increase over control salinity.

Echinoderms

Effects on echinoids (sea urchins) were observed at 8.5% dilutions of a concentrate from Florida (Chesher 1971). Toxicity was thought to be related to copper concentrations.

Chordates

Effects on ascidians (tunicates) were observed at 5.8% dilutions of a concentrate from Florida (Chesher 1971). Toxicity was thought to be related to copper concentrations.

Sea Grasses

Much experimental research has focused upon the effects of brine upon seagrass (*P. oceanica*) and associated fauna. Laboratory experiments have observed reduced growth, greater occurrence of necrotic lesions and premature senescence in seagrasses at salinities of approximately 39 ppt, which represents only a minor increase above ambient salinity in the study region (Sánchez-Lizaso et al. 2008). Effects on seagrass were observed at 12% dilutions of a concentrate from Florida (Chesher 1971). Toxicity was thought to be related to copper concentrations. Growth of seagrass was impaired 50% following a 15 d exposure to a 5 ppt increase in salinity (Latorre 2005).

California Biota

Data on the effects of elevated salinity and concentrate discharges on California biota are extremely limited, often not peer-reviewed, not readily available, or have flaws in the study design. Only one published study has documented impacts of a concentrate discharge on marine biota of California in the laboratory (Voutchkov 2006). Laboratory studies were conducted on 18 different species encompassing algae, invertebrates and fishes. In contrast to WET, the selection of species included biota from the site of discharge. Salinities ranged from 33.5 (control) to 40 ppt and the duration of exposure was 19 days with survival as the only measure of biological effect reported in the reference. While the author stated growth and fertilization of sea urchins held for 5.5 months were unchanged, it was apparent that the study suffered serious flaws due to a lack of replication and very low individual numbers, which effectively prevent any statistical analysis of results. Overall, the author concluded that no effects were apparent in any of the treatment groups and reported an unreferenced citation that a chronic red abalone threshold derived from WET was 40 ppt. While this author proposed a 10 ppt increase of total dissolved solids as a threshold using WET methods (with lethality as the endpoint), from the other studies reviewed here, it is apparent that a 2-3 ppt increase can produce significant deleterious effects in sea grasses and mollusks.

Benthic Communities

Depending on how it is discharged, as concentrate from desalination plants is more saline than the receiving water, it may settle on the bottom, with potential deleterious effects to the benthic community. Some field studies of desalination concentrates at various sites found adverse effects on benthic biota, while other studies found none. Differences in effects or lack thereof are due to differences in ecosystems and varying concentrate characteristics.

Diatom Communities

Benthic diatom communities were reduced in richness and abundance, and had lower chlorophyll-a in areas receiving desalination concentrate, even though salinity measurements were not different at outfall and reference locations (Crockett 1997).

Sea Grass Communities

Perez Talavera and Quesada Ruiz (2001) found that the discharge from a Canary Island desalination plant was associated with the disappearance of the sea grass, *Cympodocea nodosa,* in areas near the outfall; farther away, the grass was present but in poor condition, but at even farther distances it was in good condition. They found that although the initial dilution was high, a 2 ppt increase of bottom salinity remained over a large area on the bottom, accounting for the effects on the plants. A 1 ppt increase was observed on the surface. Gacia et al. (2007) studied seagrass (*Posidonia oceania*) meadows exposed to concentrate from a reverse osmosis plant, which contained nitrogen from the groundwater source of the intake water. The salinity was 5 ppt above background 10 m from the outlet. Respective increases of 2.5 ppt and 1 ppt were observed at 20 m and 30 m from the outlet. Sea urchins and sea cucumbers, present at the reference sites, were absent from transects that were impacted by the concentrate. These species are highly sensitive to elevated salinity. The sea grass itself showed reduced growth and necrotic tissue, but there was no extensive decline of the meadow. Effects on seagrass meadows may be more apparent when organisms inhabiting depressions in substrata, where pore waters are more saline are studied, since they may be differentially exposed.

In field experiments, discharge from a pilot desalination plant was pumped to experimental seagrass plots for three months (Latorre 2005, Sánchez-Lizaso et al. 2008), elevating salinities from about 37.7 psu, to a range of 38.4 to39.2 psu. These slight, increases in salinity resulted in reduced survival of seagrass, reduced shoot abundance, length and biomass, and presence of necrotic lesions. Monitoring of meadows adjacent to plant outfalls also revealed reduced shoot density, greater abundance of epiphytes, and reduced abundance of epifauna (Latorre 2005, Sánchez-Lizaso et al. 2008). That impacts to seagrasses can occur following increases of only 1 to 2 ppt in salinity highlights the potential sensitivity of these species to desalination brines

Soft Bottom Benthic Infauna

Del Pilar Ruso et al. (2007) found major impacts on benthic communities following the opening of a seawater reverse osmosis plant in Alicante, Spain. The discharge had low initial dilution, and the salinity near the discharge was over 39 psu, and farther away was 37.9. Before the discharge began, the benthic infaunal community was studied along three transects perpendicular to the coast, and was dominated by polychaetes, nematodes, and bivalves. Over time, following the opening of the plant, the diversity and abundance of polychaetes decreased at distances of up to 400 m from the discharge, and the community became dominated by nematodes alone, with the degree of dominance increasing over the two-year time period of the study. This study demonstrates the importance of baseline studies before a plant goes into operation.

Meiofauna

Riera et al. (2011) assessed whether proximity to a brine discharge point in the Canary Islands altered patterns in the abundance and assemblage structure of subtidal, soft-bottom, meiofauna. Samples were collected twice (May 2008 and January 2009) at 0, 15 and 30 m away from the brine discharge point, corresponding to a change in salinity from 45 to 36 psu. Proximity to the brine discharge point affected overall meiofaunal abundance: lowest abundances were observed at 0 m (mean of 64.55 individuals per 10 cm^2) than at 15 (average of 210.49 individuals) and 30 m (average of 361.88 individuals) away from the discharge point. This was particularly notable for the dominant meiofauna: nematodes and copepods. The community structure also differed with varying proximity to the brine discharge point. Multivariate techniques identified changes in salinity as a relevant driver of patterns in community structure, but a shift in particle size composition between the sampling dates also contributed to differences in abundance and assemblage structure with proximity to the discharge point. Hence, meiofauna are suitable for monitoring impacts of hypersaline effluents as long as potential confounding factors, (e.g., changes in particle size composition) are accounted for.

Coral Communities:

In contrast to previous reports of benthic effects, an intensive field study in Antigua found that elevated salinity (sometimes over 40) was present down slope from the brine discharge, but no significant changes were seen in the biotic community including seagrasses, hard and soft corals, algae, other invertebrates, and fishes (Blake et al. 1996, Hammond et al. 1998, cited in NRC 2008). Coral heads exposed to a salinity elevation of 4.5 psu showed no adverse effects after six months. The elevated salinity was distributed well past the 10 m study area, mainly down slope. Unfortunately, although this study did have baseline data, it did not continue after six months, and no reference sites were monitored. The investigators did perform settling plate studies, and seasonal differences were attributed to reproductive seasons rather than to elevated salinity. However, they did not perform settling plate studies before the discharge commenced, so it is not known if the discharge prevented any species from settling. This study was not published in the refereed scientific literature. In contrast, massive losses of coral, plankton and fish in the Hurghada region of the Red Sea have been attributed to desalination discharges, although the data supporting this claim were not presented by the authors and the impacts must be considered anecdotal (Mabrook 1994).

Raventos et al. (2006) found no significant impacts of discharges from a small seawater desalination plant in the northwest Mediterranean, using visual census techniques 12 times before and 12 times after the plant began operating. Levels of salinity were equivalent to background within 10 m of the outlet. The lack of effects was attributed to high natural variability and to rapid dilution of the concentrate upon leaving the discharge pipe, which had a diffuser with 43 ports to facilitate rapid dilution. The use of visual census precludes an analysis of benthic infauna, which may have been affected.

4.2 Overall Summary of Effects:

Roberts et al. (2010a) reviewed previous work on the environmental impacts of discharge from desalination plants. Experiments in the field and laboratory demonstrated the potential for acute and chronic toxicity, and small-scale alterations to community structure after exposures to concentrations of concentrate likely to be present near the discharge site. It was clear from the reviewed articles that the method of discharge and site selection are the main factors determining the extent of ecological impacts. Ecological monitoring studies have found variable effects ranging from no significant benthic impacts to widespread alterations of community structure in seagrass, coral reef, and soft-sediment systems when discharges are released in poorly flushed areas. In most other cases, environmental effects appeared to be limited to within 10s of meters of outfalls. The greatest impacts in the field have occurred around older multi-stage flash (MSF) plants discharging to water bodies with little flushing, a discharge scenario not relevant in California. Such sites show substantial increases in salinity and temperature, along with accumulation of metals, hydrocarbons, and anti-fouling compounds in receiving waters. Environmental issues associated with older desalination plants have often been linked to excessive copper content of the concentrate (Chesher 1971), an issue that is now largely avoidable.

Roberts et al. noted that a large proportion of the published work is descriptive and provides little quantitative data that can be assessed independently. Many monitoring studies lacked sufficient details of study design and statistical analyses, making interpretation of results difficult. They concluded that greater clarity and improved methodologies are required in the assessment of the ecological impacts of desalination plants, that it is necessary to employ Before–After, Control-Impact (BACI) monitoring designs with adequate replication, and that multiple independent reference locations are needed to assess potential impacts adequately. Such studies using robust experimental designs are currently underway in Australia (e.g., Perth and Sydney desalination plants) and are expected to substantially add to our understanding of field effects of desalination concentrate discharge. Detailed results from these studies are not yet available for review.

4.3 Recommendations

Based on published studies, a salinity increment of less than about 2 to3 psu would seem to be protective of local ecosystems. However, as we have noted, there is a surprising paucity of studies focused on sublethal impacts, including effects on biota in California waters. In addition, this value does not include site specific aspects of water quality or bathymetry which could substantially affect threshold determinations. For example, in embayments with limited flushing, thresholds may be lower in anadromous fish such as salmonids or estuarine demersal flatfish, which undergo saltwater acclimation and significant endocrine alterations.

Additional and long-term studies are needed specifically on sublethal endpoints such as reproduction, endocrine disruption, development, and behavior, and on additional taxa including benthic invertebrates and fish (i.e., demersal flatfish). Studies are also needed on different types of concentrates and mixtures with antiscalants and other chemicals associated with RO. Such studies are strongly encouraged, and their results may warrant more conservative levels.

Studies are also needed to evaluate the potential impact of concentrate discharge into estuarine waterways that have hydrodynamic issues which may limit discharge dilution and dispersion. For example, the combination of freshwater removal and climate change-induced hypersalinization combined with significant concentrate discharge into embayments such as the San Francisco Bay Delta may substantially alter species that utilize salinity gradients for critical life history segments (i.e., salmonids; delta smelt). It is likely WET would greatly underestimate the impacts of these combined stressors.

5. DISCHARGE SITE SCENARIOS

This section discusses the key features of the discharge site that can influence the fate and ecological risk of concentrate discharges. The geomorphic, depth, and habitat characteristics of the discharge site are important factors to consider, as well as variability in salinity, temperature, and waves. These factors should be considered in determining the suitability of a site for discharge and in developing effluent limits and monitoring requirements. Receiving water salinity can vary naturally by as much as 8% to 71%, depending on location along the California Coast. A statewide water quality objective for salinity should not be expressed only as an absolute salinity value, as such an objective would likely not be appropriate for all discharge scenarios. Discharge sites with high ambient mixing and advection (typical of exposed, open-ocean, collision-coastlines) are preferred, due to their greater ability to dilute and disperse the discharge. Discharge sites with bathymetric depressions (hollows) or barriers (offshore rocky outcrops) should be avoided with negatively buoyant discharges. Such sites have an increased potential for accumulation resulting in degraded water quality in the near-bottom. Discharge sites in estuarine embayments present greater potential ecological risk from concentrate discharge, due to accelerated flocculation of the sediment in the receiving water or reduced dispersion of the plume.

5.1 Key Site Characteristics

The dilution, dispersion, and biological impacts of a concentrate discharge are determined by the interaction of multiple factors specific to the site and discharge. From a plume modeling perspective, there are three types of factors to be considered: 1) effluent characteristics and type and mode of discharge, 2) boundary conditions that represent the topography and other physical and biological characteristics of the receiving environment (i.e., the far field), and 3) forcing functions such as tides, waves, and currents. All of these factors interact on a site-specific basis and should be considered when assessing the ecological risk of the discharge (Figure 5-1). The Panel recommends that these factors and their interaction, which constitute the discharge site scenario, be considered in evaluating the impacts of concentrate discharges and establishing permit conditions.

Effluent Type

As discussed in Section 3, the physical and chemical composition of the concentrate can vary greatly, depending on the characteristics of the feedwater and additives. The type of concentrate considered in this report includes brine from ocean desalination facilities, as well as concentrates from treated municipal wastewater or brackish water. Each of these effluent types are likely to vary in salinity and the concentration of potentially toxic constituents from chemical additives or wastewater contaminants. These characteristics, in addition to the volume discharged and temperature, will influence the initial dilution of the discharge, its interaction with other constituents in a combined discharge, and the overall toxic potential to marine life.

Discharge Mode

The mode of discharge controls the physical properties of the discharge plume, the most significant of which is the net buoyancy of the plume. There are three principal discharge modes to be considered: 1) positively buoyant combined discharges that blend concentrate with thermal or wastewater effluents using existing infrastructure, 2) negatively buoyant discharge using dedicated brine discharge infrastructure, and 3) sometimes or weakly-negatively buoyant combined discharge when brine is the predominant effluent constituent. Each of these discharge modes will interact differently with the receiving environment, producing different near field different characteristics and dimensions in both the near and far fields. Discharge modes are discussed further in Section 6.

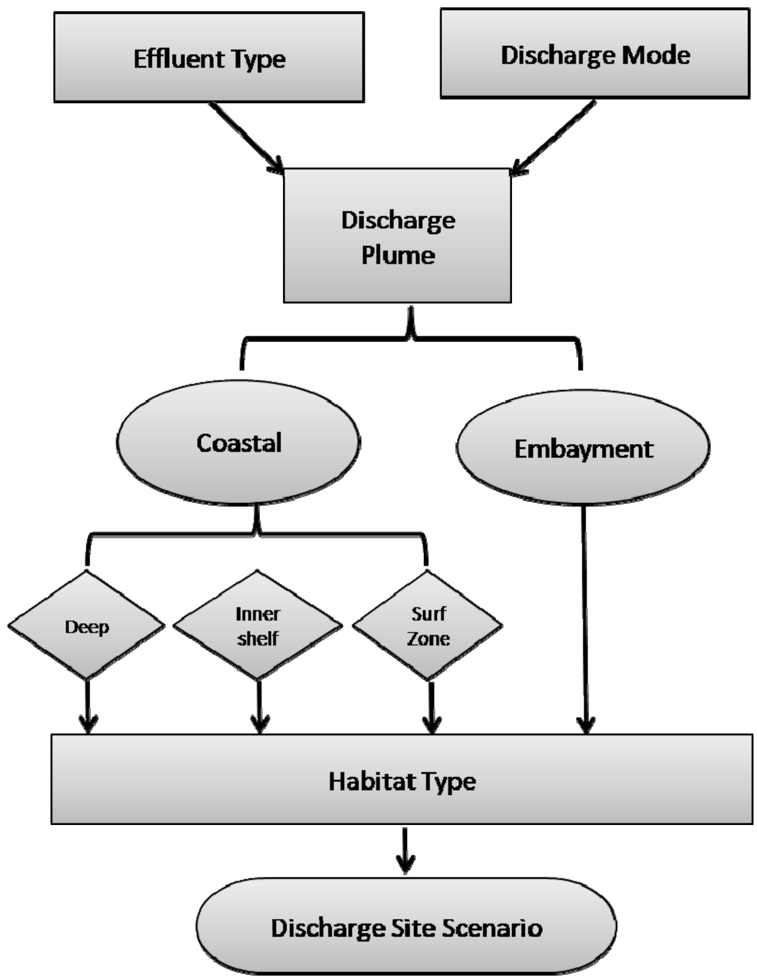

Figure 5.1. Factors influencing discharge site scenarios.

Receiving Environment

The physical boundaries, oceanography, and geomorphology of the receiving environment affect the fate of the plume and also determine the nature of the biological communities potentially affected by the discharge. Key boundary conditions that should be considered include coastal type, bathymetry and coastal structures, sediment properties, and water mass properties (salinity and temperature). The coastal type includes collision coasts, which are exposed open coastlines that are accompanied by either sandy or rocky intertidal and subtidal habitats. The geomorphology of both the sandy and rocky collision coastal types creates high-energy coastal environments with vigorous ambient mixing and advection that contributes to rapid dilution that limits dispersion and accelerates extinction of brine discharge plumes.

Concentrate releases into the open ocean will be influenced by different currents as a function of the depth (i.e., location) of the discharge. The three primary circulation regimes that can be expected in the coastal setting are shown in Figure 5-1: 1) surf zone, 2) inner shelf, and 3) deep waters. These regimes are distinguished by the different processes that dominate their currents. The surf zone is that shallow water province at the shoreline in which currents are dominated by the effects of breaking waves. The inner shelf is that region from the surf zone to the offshore location where incident waves begin to refract and shoal and where surface and bottom boundary layers (also known as Ekman layers) first intersect. Finally, the deep water is the offshore region for which surface and bottom boundary layers exist but the bulk of the flow is dominated by the geostrophic acceleration balance between horizontal pressure gradients and the effect of Earth's rotation (i.e., the Coriolis acceleration). The exact depths that determine the boundaries between the three offshore regimes vary from place to place and, at one location, in time. The boundary between the surf zone and the inner shelf is around 10 m and between the inner shelf and the deep water is around 30 m.

Tidally influenced bays and estuaries represent another important coastal type in California. This coastal type has three subsets: 1) estuarine embayments at the mouth of a major coastal river/watershed (eg. San Francisco Bay), 2) marine embayments that no longer have significant fresh water input, due usually to river diversion/navigation projects (e.g., San Diego Bay), and 3) man-made embayments (e.g., Los Alamitos Bay). All types of embayments in California are low energy depositional environments. The predominant mixing and advection processes in these environments are tidally driven, although the interaction of river flow with the tidal transport produces salt wedge stratification and flocculation dynamics that can have a controlling influence on concentrate dilution and dispersion.

Geomorphology also influences the resident biological communities of a particular coastal environment. Open coasts with sandy environments support soft-bottom habitat species (particularly benthic macroinvertebrates), while rocky coasts provide substrate for kelp-based marine communities. Rocky coasts support tide pool environments, and kelp beds and sea grasses in the offshore environments, both of which are often protected in California as designated Areas of Special Biological Significance (ASBS). The dispersion of concentrate discharges near any ASBS should be avoided as discharges into ASBS of any kind are prohibited by State law. Estuarine embayments are generally complex and highly productive ecosystems, likely to have tidal marshes in the intertidal zone, which is another sensitive habitat type protected under the Clean Water Act. The subtidal areas of embayments are generally nurseries for a variety of juvenile fish species and are considered to be sensitive habitats under the California Coastal Act.

Forcing Functions

Forcing functions affect the strength of ocean mixing, ventilation and available dilution volume in shallow water, including: waves, currents, ocean water levels (tides and sea level anomalies), and winds. Movement of material in inner shelf and surf zone, including the average and low-frequency movements, is controlled by the waves approaching the beach and the shape of the bottom. Waves striking the beach at an oblique angle drive mean currents up or down the beach (cross-shore currents) and parallel to the beach (long-shore currents) as a function of the incidence angle. When waves approach the beach straight on, local rip currents are more common, which are associated with complex circulation cells that control the amount of mixing with waters outside the surf zone.

Hence, any assessment or monitoring of a shoreline (i.e., surf zone) discharge must include local wave statistics and bathymetry data. Appropriate theory, models, and data for the assessment of dispersion in the surf zone can be seen in a number of textbooks on coastal processes and sedimentation, including the ones by Dean and Dalyrmple (2002) and Arnott (2010).

Of the three current regimes, the inner shelf has received the least amount of study. By definition, it is a transition region between the surf zone and deep waters. In recent years it has been recognized that the inner shelf is a critical region with regard to cross-shore exchange of material and better characterization of inner shelf dynamics has been shown. A good review of the important dynamics in the inner shelf is given by Lentz and Fewings (2012).

The deep water currents outside the inner shelf are a combination of flow driven by local winds and geostrophic currents. Accurate modeling of velocity statistics for a given location can be difficult because, in all cases, flow is highly dependent on the conditions at the boundary of any local model. (For a review of numerical ocean circulation models see Miller 2007). Direct measurements in the region may be adequate to describe the velocity variability if the data records are appropriately long. Care must be taken, however, to focus model results and observations on the local bottom currents when assessing the fate of concentrate in a negatively buoyant plume. In particular, deep water bottom currents will include the effects of the frictional bottom Ekman (boundary) layer (see: Csanady 1982 and Cushman-Roisin and Beckers 2011).

5.2 Discharge and Site Variability

None of the initial conditions, forcing functions or the boundary conditions of the far-field are constants over time, and consequently the dilution and dispersion of concentrate discharge can have considerable variability, which complicates the determination of "natural" conditions and prediction of discharge dispersion. There may be short-term or seasonal changes in RO operations resulting in variations in the concentrate discharge rates, salinities and temperatures. On the other hand, the temporal variation in boundary conditions and forcing functions of the far field receiving water can vary over a vast range of time scales that are related to geophysical, atmospheric, and climatic processes, including: diurnal variations related to tides, solar heating and coastal winds, monthly variations related to tidal spring/neap cycles, semi-annual variability related to summer/winter equilibrium transitions, and longer term variability related to climate (e.g., El Nino/Southern Oscillation (ENSO)). Variations in salinity, climate, and bathymetry are discussed here to illustrate some of the most important factors. Additional discussion of variability is included in Appendix E.

Salinity

Ocean salinity variation exerts a modulating effect on the impact of sea salts discharged from a desalination plant. The RO process produces a concentrated sea water reject (brine) that is a fixed multiple of the instantaneous source-water salinity (generally 1.8 to 2 times ambient). However, the ambient ocean salinity has considerably different degrees of variability throughout California.

Figure 5-2 shows the variation in daily mean salinity in the coastal waters off Huntington Beach in Southern California from 1980 until mid-2000. These data indicate ocean salinity varies naturally by 10% between summer maximums and winter minimums, with a long term average value of 33.53 parts per thousand (ppt). Maximum salinity was 34.34 ppt during the 1998 summer El Nino when southerly winds transported high salinity water from southern Baja up into the Southern California Bight. Minimum salinity was about 31.02 ppt during the 1993 winter floods.

Ocean salinity can be much more variable in other regions, such as is shown for Northern California in Figure 5-3. Here, long term variability about the mean is 71.7 %, with a long term average value of 33.39 ppt. Maximum salinity was 35.6 ppt during the 1998 summer El Nino when southerly winds transported high salinity water from southern waters. Minimum salinity was about 13.6 ppt during the 1993 winter floods. Considerably greater salinity depression occurs in the coastal waters of Northern California because the climate is wetter and the rivers discharge proportionally greater volumes of fresh water during floods (Inman and Jenkins 1999).

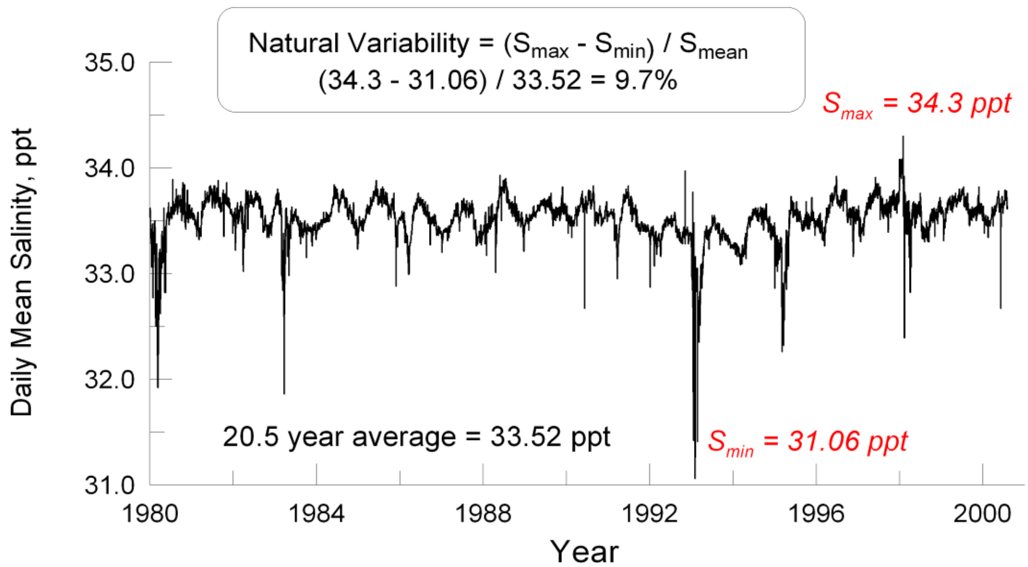

Figure 5-2. Long-term salinity variation typical of the Southern California Bight. Data from NPDES monitoring reports for AES and OCSD outfalls in Huntington Beach.

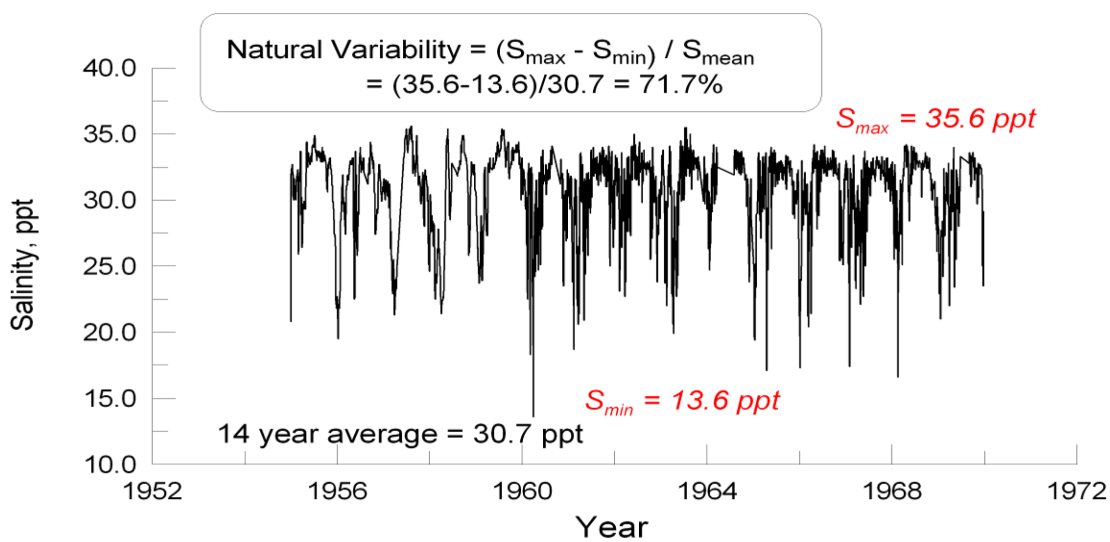

Figure 5-3. Long-term salinity variation typical of Northern California. Data for Crescent City.

Because receiving water salinity can vary naturally over the long term by as much as 71%, along the California Coast, a water quality objective for salinity should not be a fixed limit in terms of absolute salinity units. Rather, a water quality objective should be stated in terms of some relative measure of deviation from natural background, such as % deviation from background or a minimum initial dilution producing equivalent results.

Ocean climate

The plume dilution and dispersion processes in the far field are influenced by ocean temperature, salinity and the wave climate. These features vary as a result of seasonal weather cycles and can also be severely modified by global ocean climate events, such as the Pacific Decadal Oscillation (Inman and Jenkins 2004).

The effect of climate on plume dispersion is illustrated using the proposed Santa Cruz Seawater Desalination Project (SCSDP) in Monterey Bay as an example. This example assumes brine from the proposed SCSDP would be blended with treated wastewater and the combined effluent would be discharged through the existing ocean outfall one mile (1.6 km) offshore in about 110 feet (30.5 m) of water at an initial dilution of 139:1 (NPDES Permit No. CA0048194, 2005).

There are three well known hydrographic climate periods of Monterey Bay circulation, namely: 1) the wind-induced *upwelling period*, 2) the *oceanic period* dominated by *relaxation states*, and 3) the *Davidson Current period*. During upwelling, a front typically forms across the mouth of Monterey Bay with a cyclonic gyre inshore of the front inside the Bay (the Monterey Bay Gyre); and an anti-cyclonic eddy offshore of the upwelling front that is influenced by the California Current meander system, (Paduan and Cook 1997, Ramp, et al. 2009, Tseng et al. 2009). The Monterey Bay Gyre (Tseng et al. 2003) produces westward flow in the neighborhood of Santa Cruz , causing the net drift of the discharge plume to spread along shore toward the west (Figure 5-4a). Circulation patterns change and become more variable when upwelling subsides during the relaxation state. The transport of the discharge plume under this condition is less consistent, but is expected to have a slight eastward trajectory bringing the plume closer to shore (Figure 5-4b). The influence of the Davidson Current and other circulation patterns during the Davidson Current Period are very weak at the nearshore location of the SCSDP. Consequently, there is very little net mean motion in any direction from the Davidson Current circulation pattern and dispersion of the discharge plume is governed by balanced east-west tidal oscillations (Figure 5-4c).

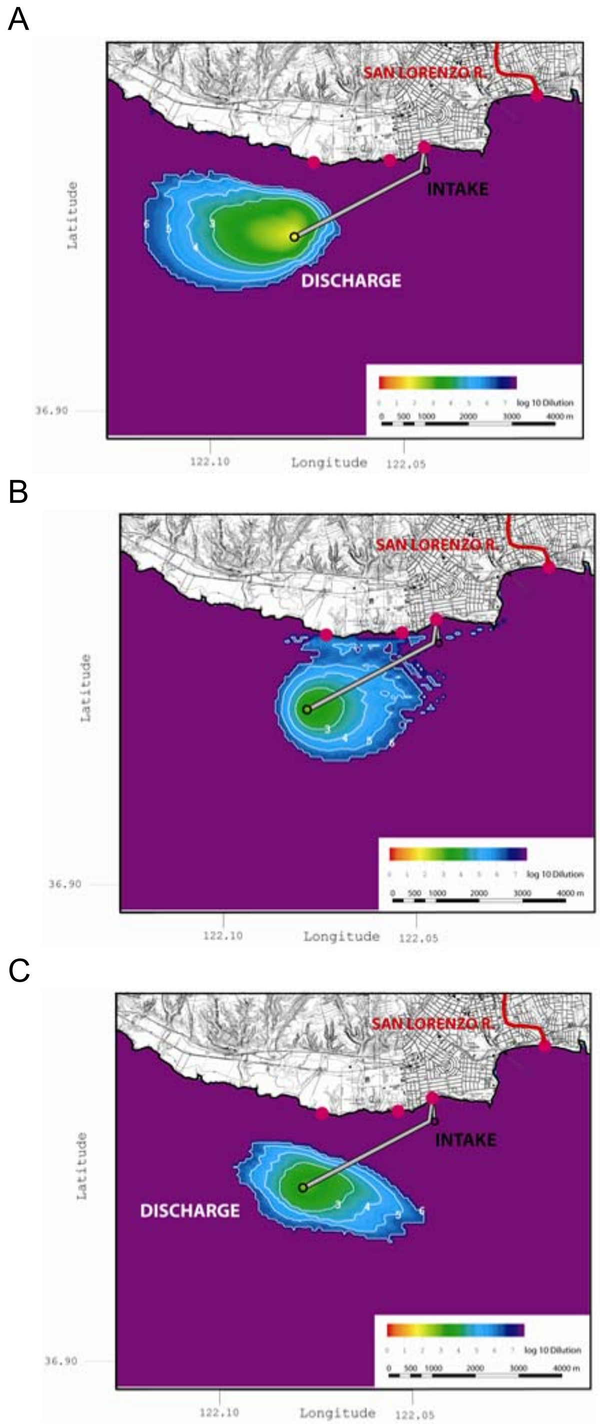

Figure 5-4. Example of effect of ocean climate on plume dispersion. Simulations based on proposed discharge off the coast of Santa Cruz, CA (from Jenkins and Wasyl, 2009).

5.3 Bathymetry and Gravity Currents

The dynamics of negatively buoyant plumes are fundamentally different that those of positively buoyant plumes. The fate of positively buoyant plumes is primarily controlled by background currents, density stratification, and wave or wind-induced mixing. They will either reach the water surface or be trapped by ambient stratification. Negatively buoyant plumes, on the other hand, will generate density (i.e., gravity) currents along the seabed by virtue of their density anomaly compared with the ambient bottom waters. The magnitude of these density currents will depend on the magnitude of the density anomaly and the bottom slope (Simpson 1997). Most of the coastline of California has favorable bottom gradients for offshore dispersion of dense plumes, because the narrow continental shelf geomorphology provides steep shelf and nearshore bathymetry. Therefore, residual high density waters at the edge of the near field in the case of negatively buoyant plumes will move away from the zone (and shoreline) under the influence of background bottom currents and self-induced density currents.

There may be environmental concerns with respect to density currents, however. The presence of rocky outcrops and reefs offshore from the discharge site may block the offshore dispersion of brine by gravity. Therefore, discharge sites with bathymetric barriers (offshore rocky reefs and outcrops) should be avoided with negatively buoyant discharges. Depending on the mixing rates with ambient waters outside of the density layer, the dissolved oxygen (DO) supply to the density layer may not meet the net oxygen demand of the benthic fauna within the layer. In this case, DO will decrease over time and, if the layer persists long enough, hypoxia or anoxia within the bottom layer can produce lethal effects in the far field well away from the discharge. This is unlikely to occur with a well-designed discharge, however.

Many factors control the development of hypoxia or anoxia, including the stratification between the ambient waters and the density layer, the thickness of the layer, the water depth, the slope of the bottom, the strength of the wind, the vertical velocity shear across the layer, and the height of the surface waves. The general situation and many of these factors are addressed in the excellent study by Hodges et al. (2011) using observations from a natural proxy to an anthropogenic negatively buoyant discharge created when high-salinity, dense waters flow out from Oso Bay into the larger Corpus Christi Bay along the Texas Gulf Coast. The potential for such a situation occurring in California can be minimized by avoiding shoreline discharges of dense undiluted concentrate.

Other far field bathymetric features to be avoided for the siting of a negatively buoyant brine discharge are bathymetric depressions (hollows). These are not generally features found along the exposed open coast of California, but can be common in embayments, either from natural shoaling effects or from man-induced activities such as the dredging of navigation channels and berthing areas. When such features are located in embayments with low mixing, a bathymetric depression can fill with brine and displace the lighter ambient seawater from the depression. This situation can result in stratification and stagnation of the bottom layer, leading to hypoxia and increased exposure of the benthos to the plume contaminants. Sites with topographic depressions should be avoided as locations for negatively buoyant discharges.

6. DISPOSAL STRATEGIES AND NEAR FIELD EFFECTS

The Panel reviewed the discharge technologies either in use, or likely to be used, in California with respect to their ability to achieve the level of dilution needed to minimize ecological risk. For a direct discharge of brine, the use of a diffuser is preferred. For flows typical of major desalination plants, a multiport diffuser will probably be required that results in high dilutions and rapid reductions of salinity in the near field. The diffuser should be designed so that the jets do not impact the water surface and the effects of jet merging should be carefully modeled (see later discussion of modeling techniques). For co-discharges with power plant cooling water, existing shoreline surface discharges, multiport diffusers, or single-port risers can probably be used. In most cases, however, near field dilution alone may not suffice to meet water quality standards and in-pipe dilution will also be needed. If the discharge is negatively buoyant, the dilution from horizontal nozzles must be carefully evaluated to ensure adequate initial dilution. Small amounts of concentrate can probably be discharged through existing municipal wastewater outfall diffusers. However, the dilution must be reevaluated to account for the change in effluent density and flow rates, and carefully evaluated if negatively buoyant.

6.1 Introduction

It is important to understand the distinctions between near field, mixing zones, and other related terms that are often associated with wastewater discharges. These are discussed further in Appendix D. The near field is a hydrodynamic, or physical, concept. It is the region where mixing of the effluent is influenced and affected by discharge parameters. The physical processes are primarily entrainment caused by shear between the buoyant jet (either positively or negatively buoyant), an internal hydraulic jump where the plume impacts a boundary (e.g., sea floor) or water surface and transitions to horizontal flow, and entrainment in the horizontally spreading layer. The near field ends where the self-induced turbulence collapses under the influence of the induced density stratification. The layer then spreads as a density current of some finite thickness. Ultimately, ambient diffusion due to oceanic turbulence is responsible for most mixing and dilution; this region is known as the far field. The rate of mixing and dilution in the far field is much slower than in the near field. A mixing zone is a regulatory concept that will generally encompass most, or all, of the near field.

The near field characteristics of negatively buoyant discharges are primarily determined by the orientation of the discharge port or nozzle to the horizontal, the jet exit velocity, and the density difference between the effluent and receiving water. Flowing currents will generally increase the dilution in the near field. For larger discharges a multiport diffuser consisting of many nozzles will be needed. In that case, an additional parameter is the port spacing and orientation of the diffuser axis to the prevailing currents.

6.2 Disposal Alternatives

Examples of common concentrate discharge scenarios are shown in Figure 6-1 (after Bleninger and Jirka 2010). Concentrates can be disposed of in several ways. They can be discharged as a surface stream at the shoreline, co-mixed (and pre-diluted) with other effluent such as municipal wastewater or power plant cooling water, or directly into the ocean as a “pure” brine stream.

For shoreline surface discharges (Figures 6-1a and b), the near field results primarily from entrainment into the surface layer (for a positively buoyant flow), or the bottom density current (for a negatively buoyant flow). This entrainment is dependent on the source velocity, as entrainment due to the spreading density currents is quite slow. Also, the density stratification reduces vertical mixing in the far field. Because of these effects, near field dilution is quite small, of order 5 times or less.

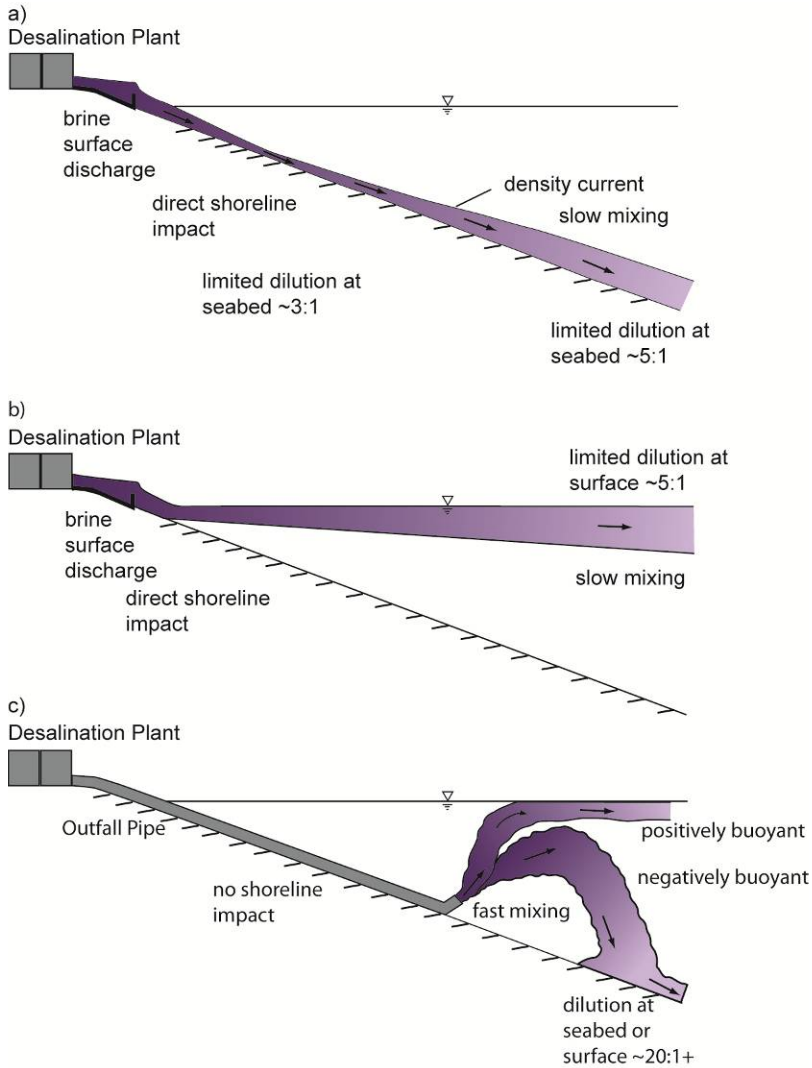

Figure 6-1. Mixing characteristics and substance distributions for various brine discharge configurations and effluents (after Bleninger and Jirka 2010). *a*) RO plant (dense effluent) shoreline discharge via channel or weir, *b*) Thermal plant (dense effluent mixed with buoyant cooling water) shoreline discharge via channel or weir, *c*) submerged discharge (dense effluent) via pipeline and nozzle or diffuser.

Shoreline discharge of raw (negatively buoyant) concentrate (Figure 6-1a) will result in a density current that runs down the bottom slope. Because the resulting density stratification inhibits vertical mixing, dilution is relatively small and benthic organisms could be exposed to relatively high salinities. Shoreline disposal of pure concentrate by this means in California is discouraged.

Co-discharge is another disposal strategy that involves diluting the concentrate to below potentially toxic levels prior to discharge into the receiving water body. This strategy involves blending brine with an existing effluent stream to achieve what is referred to as *in-the-pipe dilution* or *in-plant dilution*. Co-discharge is permitted by California water quality regulations and is currently used by several facilities. Shoreline discharges are practical if co-discharged with a much larger flow for pre-dilution, such as power plant cooling water. In this case, the effluent is likely to be positively buoyant because of the elevated temperature of the cooling water (Figure 6-1b).

There are two common means for achieving in-plant dilution: 1) co-locating the desalination plant with a wastewater plant, in which the dilution water is generally of very low salinity; or 2) co-locating the desalination plant with a power plant where the dilution water is cooling water taken from the receiving water body, typically the ocean. Dilution with wastewater produces a discharge salinity lower than ambient seawater, even at relatively low wastewater discharge rates because the treated effluent is fresh water. This is a means of reducing or eliminating hypersalinity impacts on marine life from brine discharge (Jenkins and Wasyl, 2005).

Concentrates that are blended with other effluents are typically discharged though existing ocean outfalls and diffusers (Figure 6-1c). Discharge through an existing outfall and diffuser will generally be at “low” pressure, i.e. the jet exit velocity is relatively low and the jet momentum flux will be quite small. For thermal discharges from power plants this would be either through a multiport diffuser (such as San Onofre) or a large single riser (such as Huntington Beach). In the former case, and for a municipal wastewater diffuser, the nozzles are generally horizontal. If the effluent is positively buoyant as a result of the elevated effluent temperature, the jets will ascend towards the surface. If the ambient stratification is strong enough the plumes will be trapped below the water surface, if not the plumes will reach the water surface. The near field is primarily the rising plume region and has dispersion characteristics similar to other buoyant plumes currently addressed in the Ocean Plan. Because multiport diffusers for positively buoyant effluents are predominantly horizontal, they may not be suitable for a negatively buoyant discharge and will have to be carefully evaluated. A possible solution is to open more ports on the diffuser and fit the ports with variable-area check valves which give higher velocity at low flow rates. Again, the dilution must be carefully modeled and evaluated.

For co-discharge though a single large vertical riser (such as used for some power plants) the exit dimensions may be very large, such as a square opening 25 ft on side, which is comparable to the local water depth. In that case, the initial dilution can be quite small and mixing in the spreading layer should be incorporated into the near field. These types of discharges should include in-pipe dilution of the brine with larger flows of seawater in order to achieve adequate dilution of the brine within the mixing zone.

The use of seawater to achieve in-plant dilution requires a much larger volume, relative to municipal wastewater effluent, to achieve a comparable level of reduction in the salinity of the brine discharge. The intake of seawater used for in-plant dilution (e.g., as power plant cooling water) causes additional mortality to marine organisms through velocity shear and turbulence in the confined flows through pumps and impellors of the (older design) once-through sea water circulation systems (Marcy et al. 1978, Bamber and Seaby 2004). However, recent work on hydro-electric turbines by Cada (2001) and Cada et al. (2006) has shown pump-induced turbulence mortality can be reduced by employing low speed impellors after the *Kaplan* turbine and Archimedes screw pump that reduce the shear stresses on entrained organisms to levels they can tolerate. Low-stress water wheel technologies are also being considered as alternatives to seawater circulation pumps of legacy power plants to reduce impacts on marine life. The practicality of these technologies for the applications considered here remains to be demonstrated, however.

The final case is direct discharge of negatively buoyant brine concentrate by means of high velocity jets inclined upwards. This could be either a single jet for a small discharge or a multiport diffuser for larger discharges. Multiport diffusers are used for the Perth and Sydney (Australia) desalination plants. The high jet velocities result in entrainment of ambient seawater into the jets and rapid dilution and reduction of salinity. The processes are illustrated in Figure 6-2. Dilutions exceeding 30:1 can be readily accomplished by such a diffuser.

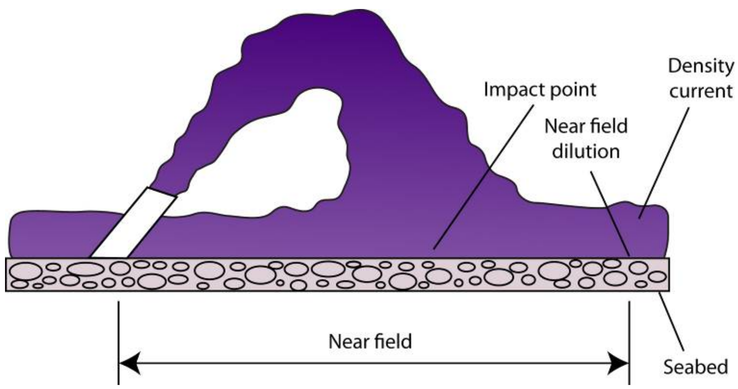

Figure 6-2. Schematic depiction of brine discharge as inclined jet.

A multiport diffuser with multiport “rosette” risers is shown in Figure 6-3. In this example, the rosettes each consist of four nozzles. Other diffusers may have the nozzles distributed uniformly along one or both sides of the diffuser.

Figure 6-3. A rosette multiport brine diffuser.

In turbulent environments, physical damage can occur to delicate eggs and larvae. The effect of turbulence on larval mortality was studied in the field by Jessopp (2007), who found that even turbulent tidal flows produce significantly increased mortality to thin-shelled veligers of gastropods and bivalves. While there is presently no known published evidence of mortality to marine species for diffuser jets, the cause and effect relations demonstrated by prior studies certainly raises that possibility. Threshold shear stress tolerances of marine organisms to diffuser discharges could be established by combining data from laboratory tests, computational fluid dynamics modeling, and field studies of diffuser systems.

7. CONCEPTUAL FRAMEWORK FOR REGULATION AND MONITORING

The Panel developed a revised regulatory framework that accommodates the varying concentrate types and discharge scenarios. The current regulatory framework is appropriate for concentrate-containing discharges that are buoyant relative to the receiving water. However, the initial dilution of the discharge and mixing zone should be reevaluated and the permit conditions modified accordingly. The initial dilution and dispersion of discharges that are negatively buoyant should be assessed using models appropriate for the discharge and receiving environment. A monitoring program consisting of both laboratory and field measurements is needed to confirm model-based predictions regarding plume dilution, fate, and effects.

7.1 Existing Regulatory Approach

Currently, there are no water quality objectives in the Ocean Plan that apply specifically to concentrate discharges from seawater desalination plants, wastewater reclamation plants, or groundwater desalting facilities. Operating seawater desalination plants discharge either directly to nearshore waters or blend the concentrate with higher volume seawater discharges. Regional Water Quality Control Boards have established permit requirements on a site-specific basis, and have applied variable effluent limits.

Concentrate discharges from wastewater and groundwater treatment facilities are not usually discharged directly into coastal waters, but rather are discharged into the influent stream of wastewater treatment systems or combined with treated wastewater effluent prior to discharge. In such cases, no special effluent limits are assigned to the concentrate discharge by the regulatory agencies; the final combined discharge must meet the water quality objectives for toxicity and chemical characteristics specified in the Ocean Plan at the boundary of the zone of initial dilution (ZID) or mixing zone. Examples of this situation are found in the NPDES permit for the Monterey Regional Water Pollution Control Agency, which receives concentrate from groundwater desalination and treatment systems, and also for the Oceanside Ocean Outfall NPDES permit, where groundwater desalination concentrate is comingled with treated wastewater effluent from several other facilities prior to ocean discharge. However, variations from the application of discharge limits have occurred. For example, concentrate from the South Coast Water District's groundwater recovery facility is currently required in their permit to meet certain Ocean Plan objectives prior to its blending with other effluents before discharge through the San Juan Creek Ocean Outfall.

7.2 Revised Regulatory Framework

New or revised permits involving concentrate discharge will need to consider a number of environmental factors that may influence the behavior and impacts of the plume. The first level determination that should be addressed is whether the discharge plume will be always positively buoyant, always negatively buoyant, or possibly positively or negatively buoyant under the range of operating conditions. The type of optimal discharge and the amount and extent of the initial dilution will depend on which situation applies. Regulations and precedent exist for the first case in which the plume is always positively buoyant. This report is focused on the second case in which the plume is always negatively buoyant. The third case in which the plume may be either positively or negatively buoyant depending on the particular operating parameters is, obviously, more complex. It is recommended that this case be evaluated and monitored for impact based on the requirements and expectations for both positively and negatively buoyant plumes.

The regulation of any new or modified discharge will follow one of the two main pathways diagrammed in Figure 7-1 as a function of its buoyancy relative to ambient receiving waters. In the case of plumes that remain positively buoyant even following the introduction of a brine component, the evaluation of the discharge will follow the existing regulatory framework for ocean discharges. The exception to this statement is the need for modified discharges to undertake a new determination of their zone of initial dilution based on the modified discharge parameters. This revision may require adjustments in the monitoring program specified by the initial permit. However, the engineering and environmental assessment guidelines for this case are covered by the existing regulatory framework.

In the case of a negatively buoyant plume, the regulation and monitoring of a new or revised discharge should follow the alternative pathway outlined in Figure 7-1 and discussed throughout this report. Both best-case engineering practices and the methods of environmental impact assessment will vary depending upon the location of the discharge - within an embayment or into the open ocean - and, in the open ocean scenario, depending upon the depth of the discharge. In all cases, the goal of the assessment is to understand both the initial and long-term fate of the concentrate discharged into the environment. The initial dilution and the expected footprint of the zone of initial dilution can be estimated using mixing models as described elsewhere in this report. Furthermore, the ultimate fate of the concentrate materials will depend on the background circulation, the local topography, and, possibly, flocculation effects. It is these processes that the revised regulatory framework recommended here is intended to address.

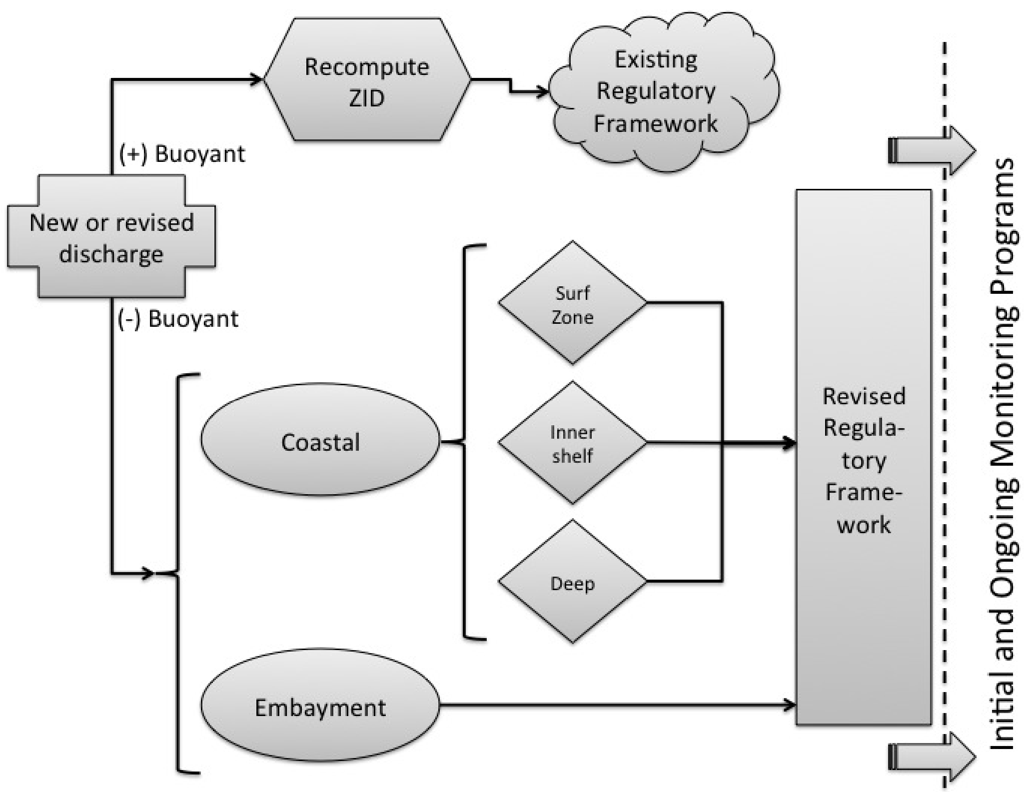

Figure 7-1. Proposed regulatory approach.

The revised regulatory framework should include three major elements (Figure 7-2). The first element consists of determining the near field characteristics of the discharge, which are used to evaluate the effectiveness of the discharge for minimizing ecological impacts. Plume models are applied to the discharge characteristics, the mixing characteristics of the site, and the proposed discharge technology to determine the initial dilution and verify that receiving water objectives will be met at the edge of the mixing zone.

The second element of the recommended framework is specification of the water quality conditions to be met at the mixing zone boundary (Figure 7-2). Establishment of these limits is expected to follow the existing regulatory guidelines. In addition, for dense discharges it will be necessary to establish effluent limits for additional constituents, e.g., salinity. A salinity objective of no more than a 5 % increase relative to background is recommended.

The final element of the regulatory framework consists of a monitoring program designed in consideration of the site-specific discharge scenario. The monitoring program should contain both laboratory and field components, be designed to have adequate statistical power, and include both biological and chemical parameters relevant to the discharge. The mixing zone is a defined region around the discharge that should be equal or larger than the near field (Figure 7-3). Monitoring locations should include locations within, at the edge of, and outside of the mixing zone.

The actual dimensions of the mixing zone will continuously vary as a function of discharge and environmental characteristics, and can be estimated using modeling approaches (see Section 8). For practical purposes and most discharge situations, a fixed mixing zone extending 100 m from the discharge point is suggested for compliance monitoring. Such a zone will encompass the near field of well-designed discharges. Dense plume discharges may require monitoring of the sediments inside the regulatory mixing zone over time to address potential build-up of hazardous chemicals in the vicinity of the discharge due to flocculation followed by rapid sedimentation. Other constituents may require monitoring far outside the regulatory mixing zone, such as dissolved oxygen (DO), in order to address long-term or cumulative effects.

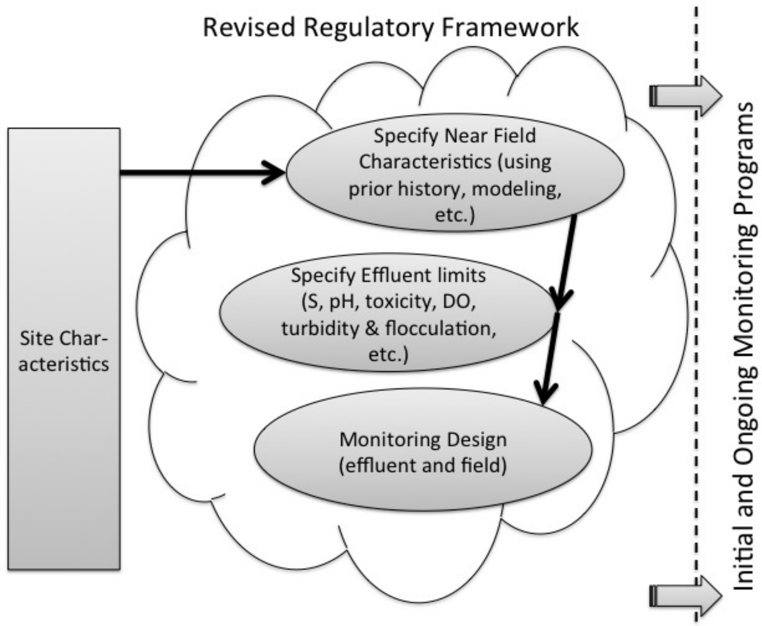

Figure 7-2. The three major elements of the expanded regulatory framework with respect to dense plumes.

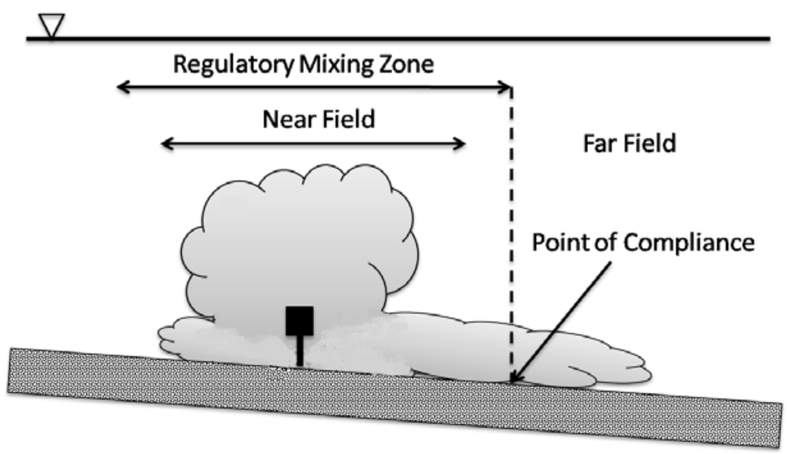

Figure 7-3. Relationship of regulatory boundaries to plume features.

7.3 The Various Discharge Site Scenarios

Depending upon the design of a given discharge, the concentrate may be released into very different oceanographic regimes and habitats (Section 5). The processes that drive the currents vary fundamentally across the different regimes, which means that models used and data required to predict or monitor the fate of the plume also vary widely. Any assessment or monitoring program should identify the dominant circulation processes associated with the discharge site scenario and choose appropriate models and data sources.

The contrasting situation of discharge into an embayment versus the open ocean is a clear first-cut distinction. Coastal open ocean discharges provide the best opportunity for plume dispersion and are preferable to embayments as sites for concentrate discharges. In most cases, discharge into an embayment is expected to restrict the long-term dispersion of the concentrate compared with a similar discharge into the open ocean. It will be critical to determine the residence time of water within the embayment, particularly for bottom water. Cases where bottom waters in an enclosed bay have little or no exchange with the ocean have long residence times (i.e., measured in weeks or months) and represent very poor choices for receiving water with respect to negatively buoyant plumes. Other cases exist, however, in which the residence time of bottom waters in an enclosed bay are very short (i.e., measured in hours or days). This can occur, for example, in enclosed bays with a strong tidal connection to the ocean. If the tidal prism is large relative to the volume of the bay then bay waters are mixed quite extensively with ocean water. If concentrate were discharged into the strong tidal flow, then its long-term dispersion would be relatively high.

7.4 Monitoring Concentrate Dispersion

Monitoring programs represent an important aspect of the recommended permitting process for negatively buoyant discharges. Modeling and background observations conducted prior to commissioning can and should be used to predict plume behavior but monitoring after commissioning is essential to validate those predictions. Monitoring should be used to confirm predictions about the near field. i.e., its size and near field dilutions. This includes physical properties and brine constituent concentrations. Monitoring should also be used to confirm predictions about far field effects, including potential hypoxia generation resulting from bottom trapped density layers as described in this section.

Monitoring should be short term and long term. Because the near field dilution modeling and the far field distribution modeling is complex and site specific, the initial monitoring should be much more comprehensive in both time and space. Critical diurnal variations, such as oceanographic tides, atmospheric sea breezes, and discharge cycling should be resolved during the initial monitoring phase. Similarly, spatial sampling grids must be dense enough within the near field to confirm predictions about the zone of initial dilution and they must extend far enough into the far field to validate predictions about the movement and breakdown of any bottom-trapped density layers. In addition, local two-dimensional bathymetry data must be incorporated into the sampling design so that all observations are placed into the proper context in terms of sloping versus flat bottoms and any local depressions.

The types of observations taken during the various monitoring phases will also be site dependent. Various chemical constituents known to exist in the initial concentrate at levels exceeding environmental standards must be shown to have been reduced to acceptable levels at the edge of the regulatory mixing zone. In terms of far field effects, sampling with commonly available conductivity, temperature, and depth plus dissolved oxygen (CTD+O2) sensor suites, as was illustrated in the study of Hodges et al. (2011), is a good alternative. In addition to the relationship of the sampling grid to the local topography, it will be essential to collect observations that extend very close (i.e., within centimeters) of the sea bottom to identify and track potential thin layers along the bottom.

8. DISCHARGE MODELING AND ASSESSMENT

The Panel reviewed the key issues involved with modeling the dilution and fate of negatively buoyant discharges. Different modeling approaches are needed to assess plume fate at the three scales of importance to receiving water impacts: near field, far field, and overall flushing. Near field modeling may involve physical or numeric approaches. For numeric modeling, the use of entrainment models is recommended. Several entrainment models are available for use with dense plumes. Different models and more extensive data are needed to assess far field plume fate. Both near field and far field model results must be coupled to predict overall plume behavior. The use of mass-balance box models is recommended for assessing the long-term buildup of contaminants in the vicinity of the discharge.

8.1 General Considerations

The fate and transport of concentrate discharged into the ocean depend on processes that operate over a very wide range of length and time scales. Shortly after discharge, turbulent entrainment dominates, the plumes may then impact the sea bed if negatively buoyant, or rise and be trapped by ambient stratification or impact the water surface if positively buoyant. After reaching their terminal levels, the flows becomes primarily horizontal, may undergo an internal hydraulic jump, and entrain further seawater, but eventually the turbulence collapses under its own induced density stratification. All these processes are commonly referred to as near field processes, i.e. determined by the discharge itself under parameters under the control of the outfall designer. Beyond the near field, the plume drifts with the ocean currents and is diffused by oceanic turbulence; this region is referred to as the far field. The rate of mixing in the near field is generally much greater than in the far field. In addition, there may be a region of lateral spreading as dynamical density current. This is sometimes referred to as a mid-field.

Near field processes typically operate on time scales of minutes and over length scales of tens of meters. Far field processes operate under time scales of hours to days and length scales of tens of meters to kilometers. Finally, large scale ocean currents and other processes such as upwelling determine the long-time flushing of contaminants and build-up of background levels over time scales of weeks to months.

Mixing in the near field is very rapid and high dilutions are readily obtained that rapidly reduce contaminant levels. Because of this, regulatory agencies allow a small mixing zone and water quality regulations are met at the mixing zone boundaries. The distinctions between near and far field and mixing zones are discussed in more detail in Appendix D.

The modeling challenges are to predict water quality at the mixing zone boundaries to ensure that water quality regulations are met and to assess the longer term fate of the effluent. Because of the very wide range of length scales involved it is generally not feasible to capture all of them in one model. Instead, separate models are employed for each phase and the models are coupled. In this section, we discuss some of the essential issues involved in modeling. First we discuss near field models, then far field models, and finally a simple box model to assess overall flushing. For a more detailed discussion, see Appendix F.

8.2 Discharge Configurations

Depending on the discharge area characteristics and the dilution needed for regulatory compliance, various concentrate disposal modes are possible, including co-discharge with power plant cooling water or municipal wastewater, and direct discharge. Depending on the flows and densities, effluents from co-disposal may be positively or negatively buoyant. Here we mainly consider negatively buoyant discharges, as modeling positively buoyant discharges are covered extensively in other publications and are accommodated by the current Ocean Plan.

Shoreline surface discharges of negatively buoyant effluent (Figure 6-1a) will result in a density current that runs down the bottom slope. Because the resulting density stratification inhibits vertical mixing, dilution is relatively small and benthic organisms will be exposed to relatively high salinities. This mode of disposal is therefore not recommended and shoreline disposal of pure brine by this means in California seems unlikely.

Shoreline surface discharges are sometimes employed with co-disposal of power plant cooling water. In that case, the mixed effluent is probably positively buoyant and forms a surface jet (Figure 6-1b). Near field mixing is also quite slow for this case.

In the absence of a co-located facility, offshore submerged diffuser systems are used to maximize brine dilution, Figure 6-1c.

8.3 Characteristics of Negatively Buoyant Diffuser Discharges

In order to effect high dilution of negatively buoyant effluent it will be necessary to discharge it as high velocity jets through a diffuser (Figure 6-2). These diffuser systems effect rapid mixing and dilution by entrainment into the jet. Because the jet is dense, it falls back to the seabed where it then spreads as a density current. The highest seabed salinity occurs where the centerline of the jet impacts the seabed. Additional dilution occurs beyond this point before the flow collapses under the influence of the induced density stratification. The point where this collapse occurs is the end of the near field, and the dilution at this point is the near field dilution. The processes in the near field operate over small scales: distances of order tens of meters and times of order minutes.

Figure 6-2 shows details of the different flow regions: the ascending jet phase, terminal rise height, descending jet phase, seabed impaction and transition to horizontal flow, mixing in the density current, and finally into the far field. For multiport diffusers, such as the one shown in Figure 6-3, merging of the individual jets and the concomitant reduction in dilution must also be considered. The degree of dilution depends on the exit velocity and jet diameter, the effluent and receiving water densities, and ambient currents. It can be estimated in stagnant environments by semi-empirical equations as discussed below.

The far field is located farther away from the discharge point, where the brine becomes a gravity current that flows down the seabed slope or horizontally in the case of a flat seabed. Mixing depends primarily on ambient (oceanic) turbulence and is affected by currents, breaking waves, etc. The difference in density between the spreading layer and receiving waters results in a density stratification that reduces vertical mixing. Because of these effects, the rate of mixing is much slower than in the near field. Flow and mixing characteristics are dominated by larger scales: distances of order hundreds of meters to kilometers, and times of order hours.

8.4 Near Field Modeling

Introduction

Modeling positively buoyant discharges from submerged diffusers has been discussed in many publications, for example, Roberts et al. (2011) and Davis (1999). The near fields are usually simulated by entrainment-type models. Examples of models that are widely used include CORMIX (CorJet) module, UM3 module of Visual Plumes, VisJet, and NRFIELD. Shoreline surface buoyant discharges are also often modeled by entrainment models. Well known models include Cormix2 and PDS, a component of Visual Plumes. For a review of surface buoyant jet modeling, see Jirka (2007ab) and Davis (1999). Because positively buoyant discharges are extensively covered in the above publications and elsewhere, we do not consider them further here.

There are three main techniques for predicting the near fields of negatively buoyant concentrate discharges: 1) Physical modeling using scaled laboratory models, 2) Semi-empirical equations, and 3) Numerical modeling.

If mathematical models are used it is recommended that entrainment-type models are used. They should be verified, however, as the jets do not always correspond to the symmetrical Gaussian distributions assumed in these models nor do they account for Coanda effects that result in reduced entrainment and dilution as discussed further in Appendix F.

Physical Modeling

Physical modeling consists of laboratory experiments using scale models that simulate the particular case being tested at a smaller scale. Tests can be carried out on any effluent, discharge configuration, and ambient conditions. For discussions of physical modeling, see Ettema et al. (2000) and Appendix F.

Physical modeling is particularly useful where mathematical models are not verified or uncertain, such as merging multiple jets, discharges from multiport rosettes (for example, Figure 6-3), or the effects of ambient currents. Their disadvantages are that they may be relatively expensive and it is less easy to simulate a wide variety of alternatives. Examples of physical modeling of concentrate diffusers are given in Miller and Tarrade (2010), Tarrade et al. (2010), and Miller (2011).

Semi-Empirical Equations

Semi-empirical equations (see Appendix F, Section 3.2) have been obtained from experimental studies of dense jets with the common design of a 60° orientation that are widely used for diffuser design with single (or non-merging) jets. These equations have been widely used in brine diffuser designs and validated in various field studies.

Numerical modeling

Numerical (computer) modeling is now often employed for near field predictions. They are used particularly for complex cases, such as merging jets, effects of currents, or effects of bottom slopes.

Near field predictions are usually made by entrainment models or computational fluid dynamics models (CFD). However, as will be discussed, present numerical models cannot accurately simulate all flow features, especially the effects of currents and jet merging.

Entrainment Models

Entrainment models are the most common tool for engineering analyses of jet and plume-type flows such as brine discharges. The concept is illustrated in Figure F-2. The rising plume entrains external fluid that then mixes with and dilutes the plume fluid. The entrainment hypothesis is that fluid is entrained into the plume at a rate proportional to the local centerline velocity. Because the conservation equations are integrated over the jet cross-sections, entrainment models are also known as integral models.

Although entrainment models can be used for predicting dense jets, they are subject to a number of limitations. Experiments in stationary and flowing currents reveal complex flows in which different phenomena can occur and predominate at different locations in the same jet and at different current speeds. These include shear-induced entrainment modified by a crossflow, buoyancy effects, a sharp radius of curvature near the top, bottom impingement, turbulence collapse, gravitational spreading, and bifurcation, among others. Tracer cross sections do not show axial or self-symmetry, either within the same flow at one current speed, or between flows at different current speeds. Fluid can detrain from the plume. These factors probably lead to the common refrain that entrainment models can predict trajectories (with suitable choice of entrainment coefficients), but significantly underestimate dilutions.

Merging jets from multiport diffusers result in further complications. In particular, the jets entrain, or attract, each other, sometimes called the Coanda effect. If the jets are too close together, the supply of entraining water is restricted resulting in reduced dilution. In general, entrainment models cannot predict the Coanda effect, which reduces jet rise height and dilution. For these cases, physical modeling will be more reliable.

Some common models that have been are widely used for predicting jet and plume-type flows, including dense brine discharges are Cormix, Visual Plumes (UM3), and VisJet. For a recent extensive discussion and comparison of these models for simulating dense jets in stationary environments, see Palomar et al. (2012ab).

Computational fluid dynamics models

Computational fluid dynamics (CFD) modeling is being increasingly applied to a wide variety of turbulent flows in nature and engineering. In CFD computations, the equations of continuity and momentum are solved numerically with some turbulence closure assumptions. There are several major CFD techniques; for a review, see Sotiropoulos (2005) and further discussion in Appendix F.

There have not been many applications of CFD to jet and plume-type flows such as dense jets. This is because of the geometrical complexity of realistic multiport diffusers, the large difference between port sizes and the characteristic length scales of the receiving waters, buoyancy effects, plume merging, flowing current effects, and surface and bottom interactions. CFD models of brine discharges have been reported by Muller et al (2011) and Seil and Zhang (2010).

Although promising, the complexity of CFD models and the effort required to set them up and run and long computation times suggests that entrainment models will continue to be used for many years.

8.5 Far Field Modeling

Far field hydrodynamic models are being increasingly used to predict the fate and transport of coastal discharges in the far field. Most models have been two-dimensional (depth-averaged) and this may be adequate for fairly shallow (unstratified) waters. But in deeper waters, three-dimensional models are needed. The models should be combined with field studies to ensure reliable results.

In contrast to near field models, far field hydrodynamic models require extensive data. Boundary conditions in open coastal waters must be specified, requiring detailed information on the variations of currents and water level, stratification, and other parameters and their temporal variations. Due to computational restrictions, it is usually not practical to model an area large enough that the area of interest is independent of these boundary conditions. Therefore, a common approach is to model a large area with a coarse grid and to embed a finer-scale model within it. The grid size of the smaller model is small enough to resolve scales of interest to outfall dispersion. The fine-grid model derives its boundary conditions from the larger model and is said to be nested within it. Frequently used models include Delft3D from Deltares, Mike3 from DHI, ROMS, Elcom, and many others.

8.6 Model Coupling

The separate near and far field models must be coupled to predict overall plume behavior. The problem is illustrated in Figure 8-1.

Entrainment induces a current whose magnitude is typically a few cm/s and decreases with distance from the diffuser. Therefore, typical outfalls do not significantly affect coastal circulation patterns (this may not be true for large cooling water discharges from power plants). The coupling is therefore usually considered to be one way, i.e. local currents affect the discharge, but not vice versa.

The main question is how and where to introduce the effluent flow and its pollutant mass into the far field model. This can be accomplished (see Appendix F) by making the contaminant fluxes computed at the end of the near field be mass input fluxes to the appropriate grid cells of the far field model. The height of the input cells may vary with varying current speeds; for a dense discharge they will be the cells closest to the seabed.

Suitable coupling between the near and far field models is essential for reliable prediction of impacts. If near field dilution is not accounted for, predicted far field dilutions will be much too low, leading to considerable overestimates of environmental impacts. The concern that biologically thin density layers may persist along the bottom for time periods long enough to create hypoxic conditions represents a unique aspect of the dense plume problem that could be addressed in this way given accurate stratification data near the bottom and accurate bathymetry.

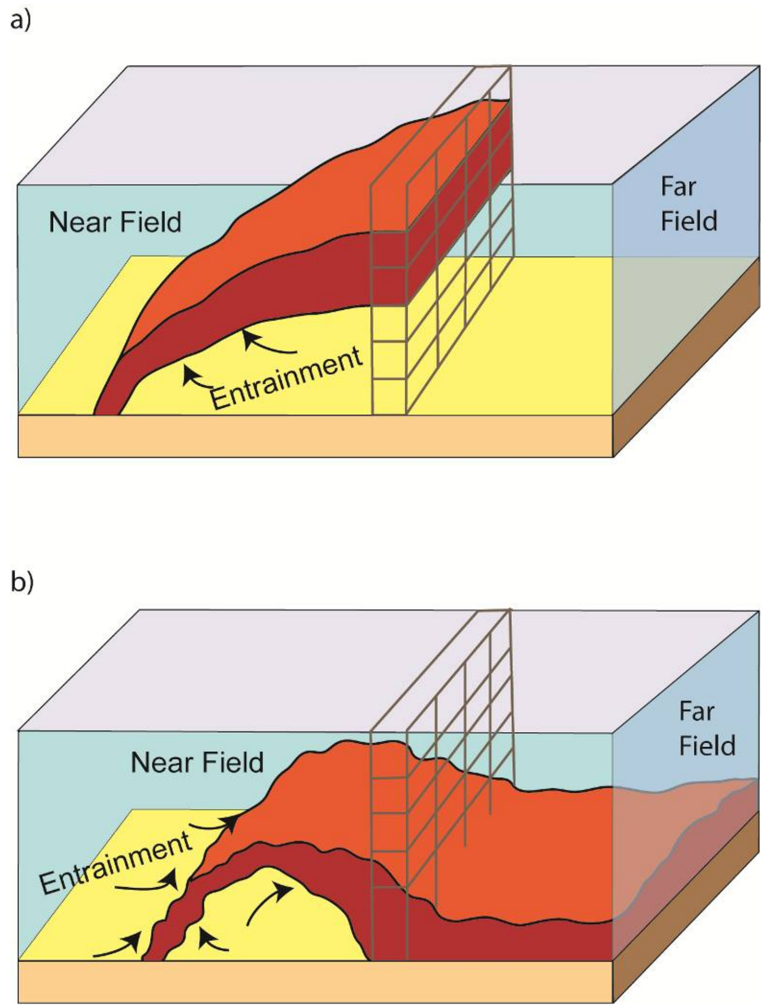

Figure 8-1 Model coupling (after Bleninger 2006). a: positively buoyant discharge, b: negatively buoyant discharge.

8.7 Box Models

Mass-balance box models (Figure 8-2) are a useful, but simple, way to assess the long-term buildup of contaminants in the vicinity of the discharge, or coastal "flushing" which occurs on long time scales. The "background" mean concentration field in the vicinity of the diffuser is governed primarily by flushing due to the mean drift, horizontal diffusion (and, for non-conservative substances, chemical and biological decay). It can reduce the near field dilution due to re-entrainment of previously discharged effluent and should be accounted for in near field dilution predictions.

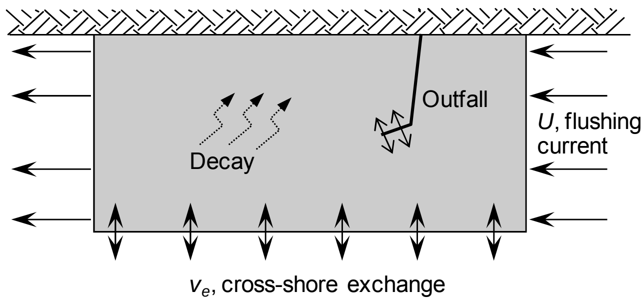

Figure 8-2 Box model for estimating long-term buildup of contaminants (after Csanady 1983).

Tidal currents distribute the effluent over an area, or "box" whose dimensions are approximately equal to the tidal amplitude. Long-term average current speeds are usually much slower than instantaneous values. Long-term average dilutions due to mean flushing currents, cross-shelf diffusion (and decay processes for non-conservative constituents) can be obtained by mass conservation using equations suggested in Appendix F. These methods give only approximate order of magnitude calculations, but they are very useful for screening and estimating long-term impacts. They can be applied to other substances such as toxic materials to estimate their potential for accumulation.

9. DISCHARGE MONITORING STRATEGIES

Concentrate discharge sites can vary in terms of physical structure, hydrology, and biological communities Consequently, any monitoring strategy should be site-specific and a "one-size-fits-all" approach will not be effective. A monitoring strategy should be based upon what the State of California wishes to protect and its policies, and should be revisited in 3-5 year increments. Monitoring of physical and chemical parameters of the influent, effluent and receiving water are required. Methods should be conducted with standard Quality Assurance/Quality Control guidelines as required under typical NPDES permitting guidelines. Monitoring should include laboratory analyses of influent and effluent as well as field components for effluent and receiving water.

9.1 Influent

Incoming water to the desalination facility should be monitored as in any other treatment facility, for the purposes of informing plant operation and maintenance decisions. Constituents would include pH, total residual chlorine, salinity, temperature, ammonia-nitrogen, suspended solids, and priority metals and other contaminants of local concern. Measures of harmful algae may also be needed if blooms are apparent at the time of sampling.

9.2 Effluent

Periodic chemical and toxicological effluent testing should be done in accordance with NPDES testing parameters, with site-specific caveats mentioned below (i.e. site-specific species etc.). In addition to standard toxicity endpoints of embryonic development, survival and growth, reproduction endpoints (particularly in vertebrates) should be added if discharge occurs within embayments where extensive dilution does not occur.

WET testing is currently used to evaluate biological impacts of discharges for NPDES permitting. This approach has provided consistency as well as standardization, and the use of biological testing provides a means to evaluate the impact of chemical and physical mixtures at the site of discharge. However, the species that are used in WET are in some cases not relevant to the sites and primarily depend upon short-term effects on survival, and in some cases growth. Consequently, care should be given with regard to species and endpoint selection. We strongly recommend the inclusion of sublethal endpoints, especially reproduction. Reproduction should be evaluated especially if the concentrate is derived from wastewater recycling, which would likely contain concentrated micropollutants and constituents of emerging concern that have undergone disinfection (typically with chlorine) and treatment with descalants. Site specific thresholds for biologic effects need to be determined for each concentrate and each discharge site. Until the impacts of these constituents and degradates have been clearly shown to not impair biota, monitoring should be employed.

From a constituent perspective, the panel has already discussed the unique aspects of concentrates with regard to potential hazardous compounds (i.e. metals, excess solutes). It is probably not necessary to include the standard list of "priority pollutants" that are normally evaluated unless the concentrate is blended with WWTP discharge or discharge in which those constituents occur. Contaminants of emerging concern should only be evaluated if concentrate from municipal wastewater or contaminated groundwater is discharged. Chlorine residuals, ammonia-nitrogen, antiscalants and other chemicals used in the reverse osmosis process should also be measured (if methods for chemical analysis exist). The same characteristics monitored for influent: pH, salinity, temperature, and suspended solids should also be included on a standard list for monitoring. The effluent should also be measured for all constituents for which effluent limits are established in the permit.

WET tests should emphasize benthic species and/or species most relevant to the site. At any of these site scenarios, the benthic habitat is of primary concern for effluents that are denser than seawater and sink to the bottom. Benthic habitats may be soft bottomed (sand or mud) or may be a hard substrate. Each substrate has unique biota that may be affected. Laboratory tests should focus on those unique species. For example, an abalone test would likely be a better species if the discharge was expected to contact hard substrate and a sand dollar test would be appropriate for sand/mud. Since field studies have indicated algal impacts at 1-2 ppt salinity increases, testing a site-specific algal species is also recommended.

9.3 Receiving Water

An important conclusion of the review by Roberts et al. (2010a) is that many published ecological monitoring programs do not appear to be scientifically defensible assessments of the impacts of concentrates. Thus, there is a general lack of empirical evidence supporting conclusions on effects of desalination concentrates in receiving systems, a fact that is recognized in almost all regions that operate large plants. Much of the research into the environmental effects of desalination plants is in the grey literature (i.e. unpublished technical reports produced by consultants and government bodies).

We recommend receiving water monitoring programs include two major design elements:

- Use of field experimentation, such as settling plates, to examine the effects of desalination concentrates under field conditions.
- Before-After Control-Impact (BACI) monitoring design that includes multiple reference locations, samples at various distances from the discharge, and repeated sampling before and after plant operation.

For California, an ecosystem monitoring program should be set up to assess potential impacts of any proposed project that would discharge concentrate. Monitoring of physical factors in the water such as salinity, pH, DO, turbidity, and high resolution near field bathymetry should be conducted concurrently with biological monitoring. Similar monitoring data has been used by Perth Australia to manage operations of the Cockburn desalination plant in order to reduce risk of hypoxia.

Limited sediment chemistry monitoring should be conducted to assess flocculation and deposition of effluent chemical components, especially with negatively buoyant plumes. Measurements in the near field are needed to assess whether effluent limits are sufficient to prevent accumulation of harmful constituents. Particular concern should be taken when concentrate is combined with effluent from sewage treatment plants, since they may contain toxic materials (e.g. metals, industrial contaminants, pharmaceuticals). These can accumulate in benthic animals, which could be a route of trophic transfer to fishes and other larger organisms. In addition, sewage treatment plants often chlorinate their effluent, and the chlorine may combine with organic materials to produce trihalomethanes and other organochlorine compounds that are toxic, bioaccumulative, and persistent. The high salinity of the concentrate may cause flocculation, and promote the movement of the toxicants towards the bottom, where they accumulate in biota. Under these circumstances it would be important to monitor the potential accumulation of these toxicants in the benthic biota.

In hard bottom environments, environmental monitoring should include the use of settling plates. These plates can give an early indication of possible effects on the recruitment of sessile organisms long before changes can be detected in the resident assemblage. The plates are placed in the water for a period of time and then removed for quantification of the abundance of the various species that have settled on them. Settling plates should be deployed at various seasons of the year, since different species have different reproductive seasons. It is most useful to deploy settling plates initially before construction of the plant, to determine which species normally settle during different times of the year. This baseline data can later be compared with results after operation of the plant has begun.

For both hard and soft bottoms, the resident benthic community should be assessed at sites along transects radiating out in different directions from the discharge. The same sites should be used for the physical/chemical measurements. The sites chosen for sampling along the transects should include both near field and far field sites. Standard techniques for community analysis should be used to quantify the abundance of various species and species richness, and to calculate a diversity index (e.g. Shannon-Wiener) from replicate samples. These data can be analyzed by parametric and multivariate statistics to gain greater insights. On hard substrates, the percent cover is another metric of interest.

The data should be used to calculate an index of biological condition. There are many such indices developed for different ecosystems and regions, and several are available for use in California. Benthic indices evaluate the ecological condition of a sample by calculating scores based on various community attributes (metrics) and comparing them to reference values expected under non-degraded conditions in similar habitats. The expected values may be different during different seasons of the year. Multiple types of metrics may be used in benthic indices, including: abundance, species diversity (richness), diversity index, and abundance/biomass of pollution-tolerant or pollution sensitive taxa. Other metrics that may be used in soft bottom environments include percent abundance of carnivores and omnivores, percent abundance of deep-deposit feeders, percent biomass and percent number of taxa found >5cm below the sediment-water interface. Benthic indices synthesize this complexity into an overall score that can be used to evaluate the overall ecological condition of a site.

All of these analyses require scientists who can identify the various organisms resident in the soft and hard bottom environments subject to the effluent. It is important that multiple reference sites be identified that are similar in nature (same benthic fauna) to the site of the future discharge..

The frequency of field monitoring is an important design element. Site-specific factors such as the size of the discharge, potential for impacts, and uncertainty in data used to derive effluent limits should be considered. For example, the Huntington Beach desalination plant permit specifies that the field monitoring frequency shall be quarterly for the 1st and 5th year of the permit and semiannually during the 2nd, 3rd and fourth year of the permit.

For monitoring to be most useful, the benthic community should be characterized and monitored at multiple phases of plant development: 1) before construction of the plant, 2) after construction but before the plant is discharging concentrate, and 3) after the plant is in operation. This will establish the baseline conditions, demonstrate before/after effects, and separate out effects of plant construction from effects of the discharge itself. Knowledge of baseline conditions will also provide guidance for the selection of appropriate reference sites for ongoing monitoring.

A power analysis study should be performed to assess the statistical power of the monitoring (e.g. how much of a change in abundance or species richness would be needed in order for the data to be statistically significant, given the natural variation found in the benthic environment). Development of the monitoring program for the Sydney Australia desalination project provides an example of this process. The process used for developing monitoring programs in Australia should be considered as a template for the design of brine discharge monitoring programs in California.

10. CONCLUSIONS AND RECOMMENDATIONS

10.1 Environmental Impacts of Discharges

- Based on existing information, a salinity increase of no more than 2 to 3 ppt in the receiving waters around the discharge appears to be protective of marine biota.
- When concentrate is blended with municipal wastewater, chemical/physical interactions of the concentrate with municipal wastewater constituents may produce toxic effects that cannot be detected using traditional WET test methods.
- A monitoring program of both the effluent and the receiving environment should be required for all discharges having potential for environmental impacts. Laboratory toxicity testing of effluent using local species and sublethal endpoints should be included. Field monitoring should include analysis of benthic community condition and employ a study design having adequate statistical power to detect changes of concern.

10.2 Discharge Strategies

- Different discharge strategies can be used, depending on site-specific considerations. There is no single discharge strategy that is optimum for all types of anticipated scenarios.
- Multiport diffusers provide the highest dilution of dense discharges. This technology is preferred when developing a new discharge containing only brine.
- Discharge sites with high ambient mixing and advection are preferable.
- Discharge sites with nearby bathymetric depressions or barriers should be avoided with negatively buoyant discharges.
- Blending or co-location with existing discharges can be effective in achieving high dilution of the discharge.
- Use of augmented seawater intake to achieve high dilution can be effective, but may result in adverse impacts due to impingement or entrainment. Clarification of whether this discharge strategy is permissible is needed in the revised regulatory framework.

10.3 Regulatory Approach

- For a blended concentrate discharge that results in a positively buoyant plume, the current process for establishing effluent limits and monitoring (i.e. the regulatory framework) is adequate.
- For negatively buoyant plumes, such as those arising from dedicated seawater desalination brine discharges, a revised regulatory framework is needed. This framework should include a revised definition of the regulatory mixing zone and a field monitoring component.
- The regulatory mixing zone should include the near field.

- Water quality objectives must be met at the edge of a regulatory mixing zone that extends vertically through the water column up to 100 m from the discharge structure in all directions.
- In addition to toxicity and other limits contained in the Ocean Plan, excess salinity at the mixing zone boundary should not exceed 5% (or an absolute increment of no more than 2 psu, whichever is less) of that occurring naturally in the receiving waters. This reduction can be achieved through a combination of in-pipe dilution and near field hydrodynamic mixing that results in an overall dilution not less than 20:1.

10.4 Discharge Modeling and Dilution Calculation

- Deterministic process-based models should be used for describing near field plume dynamics. The models must be calibrated and verified.
- Near field dilution calculations should be made using either tested semi-empirical equations available in the literature or by integral mathematical models based on entrainment assumptions. Physical modeling may be needed for complex diffuser geometries.
- In computing near field dilutions of dense discharges, conservative assumptions must be made: that ocean currents do not increase dilution, and the seabed is horizontal. The possible reduction of near field dilution due to reentrainment caused by limited overall flushing must also be accounted for.
- Discharges near areas of special biological significance should be avoided.

10.5 Data Gaps and Research Needs

- Additional research is needed on the sublethal and chronic effects of elevated salinity to sensitive life stages and locally relevant species. Emphasis should be given to effects on benthic species likely to be exposed from negatively buoyant plumes.
- Insufficient toxicology data are available to evaluate the potential ecological risk of RO chemical additives and interactions between brine and municipal wastewater constituents. Especially lacking are studies of reproductive and behavioral effects that evaluate the final effluent mixture discharged to the environment.
- Studies are needed to investigate the impacts of turbulence from high velocity diffusers on plankton. Threshold tolerances of marine organisms to free-stream turbulent shear could be established by combining data from laboratory tests, computational fluid dynamics modeling, and field studies of diffuser systems using methods previously applied to hydro-electric turbines.

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APPENDIX A. WATER QUALITY REGULATION IN CALIFORNIA

Applicable State and Federal Water Quality Law

In 1972, Congress enacted the federal Clean Water Act (CWA) to restore and maintain the chemical, physical, and biological integrity of the Nation's waters[1]. Under the CWA, the states are primarily responsible for the adoption and periodic review of water quality standards for all waters within their boundaries.

The California Porter-Cologne Water Quality Control Act[2] (Porter-Cologne) of 1969 is the primary water quality law in California. The State Legislature, in adopting Porter-Cologne, directed that California's waters "shall be regulated to attain the highest water quality which is reasonable". Porter-Cologne addresses two primary functions: water quality control planning and waste discharge regulation. Porter-Cologne is administered regionally, within a framework of statewide coordination and policy.

Porter-Cologne authorizes the State Water Resources Control Board (State Water Board) to adopt statewide water quality control plans and directs each of the nine Regional Water Quality Control Boards (Regional Water Boards) to adopt water quality control plans that provide the basis for protecting water quality in each Region[3]. When the State Water Board adopts a water quality control plan, the state plan supersedes regional plans for the same waters, to the extent of any conflict[4]. All water quality control plans must list "beneficial uses" of waters which need to be protected; establish "water quality objectives" necessary to achieve protection for those beneficial uses; identify areas where discharges are prohibited, and set forth a program of implementation to ensure that water quality objectives are met. The program of implementation describes the actions necessary to achieve objectives, includes a time schedule for these actions to be taken, and describes the monitoring to be performed to determine compliance with the objectives[5].

Both statewide and regional plans are subject to review every three years, which may lead to periodic updates[6]. Triennial reviews are comprehensive and include a public hearing to identify issues to be addressed. The State or Regional Water Board evaluates all available information at the hearing to determine whether revisions to the plans are needed and the nature of any necessary revisions.

Amendments to a statewide or regional plan are initiated by the appropriate Water Board, and follow state and federal requirements for public participation and for environmental and economic consideration. Regulatory provisions of amendments must further be approved by the State Office of Administrative Law (OAL). Any amendments to surface water quality standards must also be approved by the U.S. Environmental Protection Agency (USEPA) in order to be effective.

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1 See 33 United States Code (U.S.C.) §1251 et seq.
2 See Wat. Code, §13000 et seq.
3 See Wat. Code, §13240.
4 See Wat.Code § 13170.
5 See Wat.Code § 13242.
6 See CWA §303(c)(1).

Under Porter-Cologne, the Water Boards regulate waste discharges that could affect water quality through waste discharge requirements, waivers or prohibitions[7]. In addition, the Water Boards are authorized to issue federal National Pollution Discharge Elimination System (NPDES) permits to point source dischargers of pollutants to navigable waters. Issued NPDES permits must implement all applicable state and federal standards, whether numeric or narrative. Permits contain technology-based effluent limitations (reflecting the pollution reduction that is achievable through technology) and any more stringent limitations necessary to meet water quality standards. NPDES permits are usually renewed (and expire) on a five-year schedule. Regional Water Boards are generally responsible for issuing the NPDES permits, which include self-monitoring and reporting programs. Consideration of the terms and conditions of NPDES permit requirements must occur at a public hearing. Regional Water Board staff also conducts periodic inspections of each permitted discharge to monitor permit compliance.

The California Ocean Plan

Porter-Cologne specifically requires the State Water Board to formulate and adopt the California Ocean Plan[8] to protect the State's ocean waters. The Ocean Plan designates ocean waters for a variety of beneficial uses, including rare and endangered species, marine habitat, fish spawning and migration and other uses (including industrial water supply), and establishes water quality objectives to protect those beneficial uses. The Ocean Plan provides the basis for regulation of wastes discharged into California's coastal waters. The State Water Board, in conjunction with the six coastal Regional Water Boards, implements and interprets the Ocean Plan. Coastal Regional Water Boards consist of the North Coast, San Francisco Bay, Central Coast, Los Angeles, Santa Ana and San Diego Regions.

The State Water Board first adopted the Ocean Plan in 1972, and has since periodically revised the Plan. The Ocean Plan was last updated in 2009[9].

[7] See Wat. Code, §§13263, 13377.
8 See Wat. Code, §13160 et seq.
[9] See http://www.waterboards.ca.gov/water_issues/programs/ocean/docs/2009_cop_adoptedeffective_usepa.pdf

APPENDIX B. PANEL MEMBER BACKGROUND

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Scott A. Jenkins received his B.S. in Chemistry and Physics from Yale University in 1972 and his Ph.D. in Physical Oceanography from Scripps Institution of Oceanography in 1980. He has been a researcher in nearshore physical oceanography and coastal engineering for 30 years, with experience in field measurements, experimental design, and theoretical modeling. He has worked on a broad range of problems related to coastal processes, including: estuarine and littoral transport; beach and shoreline erosion; scour and burial of structures on the seafloor; hydrodynamic and hydraulic modeling of estuarine and nearshore circulation involving point and non-point source pollution; wastewater and thermal effluent discharges from engineered outfalls; and brine discharged from co-located desalination plants.

Dr. Jenkins has performed hydrodynamic modeling for desalination projects by the cities of Carlsbad, Huntington Beach, Long Beach, and Santa Cruz, CA; the Los Angeles Department of Water and Power; the West Basin Municipal Water District; and the San Diego County Water Authority. Dr. Jenkins is an author of over 60 scientific papers on coastal processes, numerical hydrodynamic modeling, sediment transport, sedimentation control, and underwater glider technologies. He has given scientific presentations on brine dilution and source water modeling before the National Research Council and their Committee on Advancing Desalination Technology and before the Workshop on Environmental Issues with Desalination in California hosted by the University of California at Santa Cruz. He has also been an invited speaker at a number of national conferences on desalination, including those hosted by the Multi-State Salinity Coalition, the American Membrane Technology Association, the South Central Membrane Association, and the Association of California Water Agencies.

Dr. Jenkins holds four United States patents for coastal flow control devices. He received the 1985 Inventor of the Year Award from the Patent Law Association and was co-recipient of the 1988 Best Special Project Award from the American Council of Consulting Engineers. He was recently inducted into the Who's Who in America 64th and 65th editions (2010, 2111).

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Jeffrey D. Paduan received his B.S. in Engineering from the University of Michigan in 1982 and his Ph.D. in physical oceanography from Oregon State University in 1987. His background involves study of upper ocean currents and air-sea interaction. As a research scientist at Scripps Institution of Oceanography, his investigations focused on larger-scale current structures as measured by satellite-tracked surface drifters. In 1991, he joined the faculty of the Department of Oceanography at the Naval Postgraduate School (NPS) where his research has focused on the application of high frequency (HF) radar systems in coastal oceanography. In 1997, he co-edited a special issue of the Oceanography Society's journal (Oceanography, Vol. 10, #2) devoted to this topic. In 1999, as keynote speaker, Dr. Paduan presented an overview of this research area at the IEEE 6th Working Conference on Current Measurement Technology. In 2001, he co-founded the International Radiowave Oceanography Workshop (ROW; http://radiowaveoceanogrphy.org), which continues to be an important focal point for this growing branch of marine science.

Dr. Paduan has been principal investigator for a series of projects around Monterey Bay that have brought together observations, modeling, and data assimilation of circulation and ecosystem responses; these projects include: the ICON project, which was a Monterey Bay area component of the National Ocean Partnership Program; the NOAA/COTS Center for Integrated Marine Technology program based at UC Santa Cruz (CIMT); and the state-funded Coastal Ocean Currents Monitoring Program (COCMP). He has also designed and conducted a series of environmental assessments to characterize the thermal plumes produced by the Moss Landing and Morro Bay power plants. In addition, Dr. Paduan has co-authored 49 publications and numerous technical reports related to the physics of the upper ocean.

Dr. Paduan is a member of the American Geophysical Union, the Oceanography Society, and the American Meteorological Society (AMS). He has served on the AMS committee for Meteorology and Oceanography of the Coastal Zone, on the steering committee for the Ocean.US community workshop on ocean observing systems, and as chair of the Ocean.US steering committee for the national Surface Current Mapping Initiative. Dr. Paduan has also served as President of the Monterey Bay Crescent Ocean Research Consortium, a member of the Monterey Bay National Marine Sanctuary's Integrated Monitoring Network science steering committee, and a member of the Science Advisory Team for California's Ocean Protection Council.

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Philip J.W. Roberts received his B.S. in Mechanical Engineering from Imperial College of Science and Technology University, London in 1968 and his Ph.D. in Environmental Engineering Science from California Institute of Technology in 1977. Dr. Roberts' research interests focus on environmental fluid mechanics, particularly in terms of application to the engineering design of water intakes and ocean outfalls for disposal of wastewaters and desalination brine, and density-stratified flows in lakes, estuaries, and coastal waters. His research includes investigation of mixing and dynamics of natural water bodies, mathematical modeling of water quality, field studies, and laboratory studies of turbulent mixing.

Dr. Roberts is an authority on the fluid mechanics of outfall diffuser mixing and the development and application of mathematical models of wastewater fate and transport. He is an author of 49 scientific articles and 7 books related to this subject. He has extensive international experience in marine wastewater disposal including the design of ocean outfalls, review of disposal schemes, numerical modeling, and the design and analysis of oceanographic field study programs. In addition, he has lectured widely on outfall design and is presently Co-Chairman of the IAHR/IWA Committee on Marine Outfall Systems.

Dr. Roberts' mathematical models and methods have been adopted by the U.S. EPA and are widely used around the world. He is a regular lecturer at workshops for the U.S. EPA on mixing zone analyses and on the use of mathematical models and outfall design for the Pan American Health Organization. He has developed innovative experimental techniques for research on diffuser mixing processes using three-dimensional laser-induced fluorescence and has published extensively in this area. For this research he was awarded the Collingwood Prize of ASCE in 1980 and was UPS Foundation Visiting Professor at Stanford University in 1993-94. Dr. Roberts is presently one of only two Distinguished Scholars in the National Ocean and Atmospheric Administration (NOAA) Oceans and Human Health Initiative (OHHI) in which he is conducting research on the hydrodynamic aspects of bacterial and pathogen transport in coastal waters.

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Daniel Schlenk received his B.S. in Toxicology from Northeast Louisiana University, Monroe in 1984 and his Ph.D. in Biochemical Toxicology from Oregon State University in 1989. From 1989 to 1991 his studies were supported by a National Institute of Environmental Health Science postdoctoral fellowship at Duke University. He was a visiting Scholar in the Department of Biochemistry at the Chinese University of Hong Kong, in 1995, 1998, and 1999; a visiting scholar at the Instituto Del Mar - Venice, Italy, in 1999; a visiting scientist at the CSIRO Lucas Heights Laboratory - Sydney, Australia, in 2003; and a Distinguished Fellow of the State Key Laboratory for Marine Environmental Science in Xiamen University of China. His initial studies focused on the impacts on pesticides within estuarine systems. Today, the overall focus of Dr. Schlenk's laboratory is to evaluate mechanisms of action of chemicals in aquatic and marine organisms.

Dr. Schlenk's professional interests include impacts of hypersaline acclimation on the biotransformation and toxicity of xenobiotic chemicals in anadromous and catadromous fish. He is an author of more than 175 scientific publications related to this subject. His research in California has focused on the impacts of hypersaline acclimation on organophosphate insecticides and organoselenide compounds that are biomagnified in hypersaline waterways, such as the Salton Sea and the Central Valley. Current studies are underway to evaluate the impacts of climate change on hypersaline conditions in San Francisco Bay and the role it may have in enhancing or diminishing the toxicity of current-use pesticides, such as pyrethroids. It is his goal to understand the modes of action of these compounds, alone and in mixtures, to determine the interactive roles each may have in endocrine disruption.

Since 2007, Dr. Schlenk has served as a permanent member of the USEPA FIFRA Science Advisory Panel and will serve as chair during 2012. In addition, he was elected as a Fellow of the American Association for the Advancement of Science (AAAS) in 2009, and he served as a member of the Board of Directors for the North American Society of Environmental Toxicology and Chemistry from 2003 to 2006. He was the co-Editor-in Chief of *Aquatic Toxicology* from 2005 to 2011, and now serves on the editorial board for this journal and the editorial boards for *Toxicological Sciences*, *The Asian Journal of Ecotoxicology*, and *Marine Environmental Research*. He has also participated in proposal review panels for the USEPA, NOAA, and the National Institute of Environmental Health Sciences.

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Judith S. Weis received her B.A. in Zoology from Cornell University in 1962 and her Ph.D. in Biology from New York University in 1967. Much of her research has been focused on estuaries in the NY/NJ Harbor area, but she has also done research also in Indonesia and Madagascar.

Dr. Weis is interested in stresses in estuaries (including pollution, invasive species, and parasites), and their effects on organisms, populations and communities. Her research focuses mostly on estuarine ecology, and she has published over 200 refereed scientific papers, as well as a book on salt marshes ("Salt Marshes: A Natural and Unnatural History") in 2009 and a book on fish ("Do Fish Sleep?") published in 2011. She served for two years as a Program Director at the National Science Foundation and has been a visiting scientist with the US Environmental Protection Agency.

Dr. Weis is a Fellow of the American Association for the Advancement of Science (AAAS), was a Congressional Science Fellow with the U.S. Senate Environment and Public Works Committee, and was a Fulbright Senior Specialist in Indonesia in 2006. She serves on the editorial board for BioScience, and is one of the editors of the on-line Encyclopedia of Earth. She has also served on numerous advisory committees for USEPA, NOAA and the National Research Council and is currently chair of the Science Advisory Board of the NJ Department of Environmental Protection. She was the Chair of the Biology Section of AAAS, served on the boards of the Society of Environmental Toxicology and Chemistry (SETAC), the Association for Women in Science (AWIS), and the American Institute of Biological Sciences (AIBS), of which she was the President in 2001.

APPENDIX C. STUDIES OF BRINE IMPACTS IN MARINE SYSTEMS

Table C-1. Biological impacts of concentrate discharges. Table modified from Roberts et al., 2010a.

Species	Study Type	Condition/ Location	Observed Biological Effects	Reference
Seagrass				
Posidonia oceanica	Lab exposure	15-d exposure to 38-43 ppt	Decreased growth after exposure to salinities ≥ 40 ppt; 50% mortality at 45 ppt	Latorre 2005
Posidonia oceanica	Lab exposure	15-d exposure to 23-57 psu	Reduction of vitality and mortality at salinities ≥ 39.1, at 45 psu 50% of plants died	Sánchez-Lisazo et al. 2008
Cympodocea nodosa	Field study	Barranco del Toro Beach, Canary Islands	Decreased presence near outfall discharges. Farther away from the outfall discharge the seagrass improved condition	Perez and Ruiz 2001
Caulerpa prolifera	Field study	Barranco del Toro Beach, Canary Islands	Decreased presence near outfall discharges. Farther away from the outfall discharge the seagrass condition improved	Perez and Ruiz 2001
Posidonia oceanica	Field study	Formentera, Spain	Increased leaf necrosis and decreased carbohydrate storage near discharge site, relative to control locations	Gacia et al. 2007
Posidonia oceanica	Field study	Key West, Florida	Seagrass photosynthesis inhibited after exposure to 12% brines for 24 h	Chesher 1971
Posidonia oceanica	Field study	Shark Bay, WA	Increased mortality and senescence at salinities of 50-65 ppt	Walker and McComb 1990
Posidonia oceanica	Field study	Alicante, Spain	Exposed to brines in the field for 3 months. Exposures raised salinity to 38.4-39.2 ppt in experimental plots and caused mortality, surviving plants had reduced shoot and leaf abundance	Sánchez-Lizaso et al. 2008
Posidonia oceanica	Field study	Balearic Islands, Spain	Reduced growth and presence of necrotic tissue in seagrass from transects impacted by brine, but there was no extensive meadow decline	Gacia et al. 2007
Plankton				
	Field study	Key West, Florida	Reduced abundance in water surrounding brine discharge area. Majority of effects attributed copper levels in brine	Chesher 1971

Table 1. Continued.

Species	Study Type	Condition/ Location	Observed Biological Effects	Reference
Ascidians				
	Lab exposure	Key West, Florida	Relatively more sensitive than other invertebrates exposed in the study, 50% mortality after exposure to 5.8% effluent	Chesher 1971
	Field study	Key West, Florida	Reduced abundances in areas surrounding brine discharges. Majority of effects attributed to copper levels in brine	Chesher 1971
Mysids				
Leptomysis posidoniae	Lab exposure	15 d exposure to 23-57 psu	Mortality observed at salinities > 40 psu and it was temperature dependent	Sánchez-Lisazo et al. 2008
Echinoderms				
Paracentrotus lividus	Lab exposure	15 d exposure to 23-57 psu	Mortality observed at salinities > 40 psu and it was temperature dependent	Sánchez-Lisazo et al. 2008
	Field study	Alicante, Spain	Disappeared from meadow in front of desalination plant, lower vitality observed in seagrass in the same area	Fernandez-Torquemeda et al. 2005
	Field study	Key West, Florida	Reduced abundances in areas surrounding the effluent discharge area. Majority of effects attributed to copper levels in brine	Chesher 1971
	Lab exposure	Key West, Florida	Reduced survival after exposure to 8.5% dilutions	Chesher 1971
	Field study	Key West, Florida	Died within 2-3 d of exposure, survival improved when copper emissions were reduced following plant maintenance	Chesher 1971
Paracentrotus lividus	Field study	Balearic Islands, Spain	Sea urchins and sea cucumbers absent from transects impacted by brine	Gacia et al. 2007
Mollusks				
Sepia apama (squid embryos)	Lab exposure	99-d exposure to 39-55 ppt	Total mortality observed after exposure to 50 ppt. Egg hatching decreased at 45 ppt. Reduced growth after exposure to 45 ppt	Dupavillon and Gillanders 2009
Crassostrea virginica (juveniles and adults)	Lab exposure	60-d exposure to 45-55 psu	Brines contained high Cu concentrations. Effects in juveniles and adults observed at Cu levels between 19 -43 ug/L. Effects included, reduced reproduction and increased fungal infections	Mandelli 1975
Tapes philippinarum (clams)	Lab exposure	0.5-72 h exposure to 31-100 ppt	Mortality found at 60 ppt after 48 h, sluggish behavior observed after 24 h at 60 and 70 ppt	Iso et al. 1994

Table 1. Continued.

Species	Study Type	Condition/ Location	Observed Biological Effects	Reference
Fish				
Pagrus major (juveniles)	Lab exposure	0.5-72 h exposure to 31-100 ppt	Mortality observed at 50 ppt after 24 h, body coloration changed at this salinity after 0.5 h of exposure	Iso et al. 1994
Pleuronectes yokohumae (eggs/ larvae)	Lab exposure	0.5-144 h exposure to 31-100 ppt	Larvae mortality at 55 ppt after 140 h of exposure; egg hatchability was delayed at concentrations $\geq$ 50 ppt after 73 h	Iso et al. 1994
Benthic Communities				
	Field study	Alicante, Spain	Communities close to outfall discharges were dominated by nematodes (up to 98%); polychaetes, mollusks and crustaceans more abundant with increasing distance from discharge	Del Pilar Ruso et al. 2007
	Field study	Alicante, Spain	Reduced polychaete abundance and diversity adjacent to outfall. Ampharetidae and Paraonidae were the most and least sensitive families (respectively)	Del Pilar Ruso et al. 2008
	Field study	Antartica	A study of diatom communities found reduced richness and abundance in areas receiving brine, even though salinity measurements were not different at outfall and reference locations D46	Crockett 1997
	Field study	Grand Canaria, Canary Islands	A study of meiofauna communities found lower abundance of copepods and nematodes near outfall discharge, abundances increased away from the discharge point. A shift in particle size also contributed to the changes in abundance	Riera et al. 2011
	Field study	Tampa, Florida	No changes in the abundance of the benthic community including sea grasses, algae, hard and soft corals, and other invertebrates despite salinity increases of up 40 times higher than baseline data	Blake et al. 1996
	Field study	Hurghada, Egypt	Many fish species declined and even disappeared, as well as many planktonic organisms and corals, near the area around the plant	Mabrook 1994
	Field study	Blanes, Spain	No significant impact found by discharges after visual census. Lack of effects attributed to high natural variability and to rapid dilution	Raventos et al. 2006

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APPENDIX D. MIXING ZONES

Central to understanding the environmental impacts of an ocean discharge and how they are regulated is the concept of a mixing zone. The mixing zone is a region of non-compliance and limited water use around the diffuser. Water quality criteria must be met at the edge of the mixing zone. Within this zone the discharge undergoes energetic mixing that rapidly reduces the concentrations of most contaminants to safe levels. The mixing is caused by the turbulence generated by the high velocity of the jets issuing from the diffuser ports and by the effluent buoyancy (positive or negative) that causes it to rise or sink through the water column. These mechanisms entrain substantial quantities of ocean water that readily dilutes the effluent within a few minutes after discharge and within a few hundred meters from the diffuser.

This rapid and very substantial contaminant reduction is recognized by the concept of a regulatory mixing zone. For example, the US EPA regulations for toxics (USEPA 1991), defines a mixing zone as:

"*An area where an effluent discharge undergoes initial dilution and is extended to cover the secondary mixing in the ambient water body. A mixing zone is an allocated impact zone where water quality criteria can be exceeded as long as acutely toxic conditions are prevented.*" (Water quality criteria must be met at the edge of a mixing zone.)

Thus, water quality requirements are specified at the edge of the mixing zone rather than by end-of-pipe requirements for conventional and toxic discharges.

There is much terminology associated with wastewater mixing processes and the regulations that cover them. Unfortunately, there do not appear to be universal definitions of these terms and they are often used interchangeably and imprecisely. As summarized in Table D-1, they include zone of initial dilution, regulatory and hydrodynamic mixing zones, and near and far field mixing. This report will use the definitions given in Table D-1.

Table D-1. Outfall mixing and mixing zone terminology

Term	Definition	Comments
Mixing zone	A limited area where rapid mixing takes place and where numeric water quality criteria can be exceeded but acutely toxic conditions must be prevented. Specified dilution factors and water quality requirements must be met at the edge of the mixing zone	
Allocated impact zone (AIZ)	Same as a mixing zone	
Regulatory mixing zone	As defined by the appropriate regulatory authority	Can be a length, an area, or a volume of the water body
Legal mixing zone (LMZ)	Same as a regulatory mixing zone	
Near field	Region where mixing is caused by turbulence and other processes generated by the discharge itself	Near field processes are intimately linked to the discharge parameters and are under the control of the designer. For further discussion, see Doneker and Jirka (1999), Roberts (1999), and Roberts et al. (2010).
Hydrodynamic mixing zone	Same as near field	Near field and hydrodynamic mixing zone are synonymous with these definitions
Far field	Region where mixing is due to ambient oceanic turbulence	Far field processes are not under control of the designer
Toxic dilution zone (TDZ)	A more restrictive mixing zone within the usual mixing zone	
Initial dilution		A general term for the rapid dilution that occurs near the diffuser
Zone of initial dilution (ZID)	A region extending over the water column and extending up to one water depth around the diffuser.	A regulatory mixing zone, as defined in the U.S. EPA's 301(h) regulations (USEPA 1994)

The mixing zone may not correspond to actual physical mixing processes. It may fully encompass the near field and extend some distance into the far field, or it may not even fully contain the near field. Mixing zones can be defined as lengths, areas, or water volumes. An example is contained in the guidelines for the US National Pollutant Discharge Elimination System (NPDES) permits for the discharge of pollutants from a point source into the oceans at 40 CFR 125.121(c) U.S. Federal Water Quality that defines a mixing zone for federal waters as:

"…*the zone extending from the sea's surface to seabed and extending laterally to a distance of 100 meters in all directions from the discharge point(s) or to the boundary of the zone of initial dilution as calculated by a plume model approved by the director, whichever is greater, unless the director determines that the more restrictive mixing zone or another definition of the mixing zone is more appropriate for a specific discharge*."

The California Ocean Plan (discussed below) defines initial dilution (which is therefore a regulatory mixing zone) as:

"*…the process which results in the rapid and irreversible turbulent mixing of wastewater with ocean water around the point of discharge. For a submerged buoyant discharge, characteristic of most municipal and industrial wastes that are released from the submarine outfalls, the momentum of the discharge and its initial buoyancy act together to produce turbulent mixing. Initial dilution in this case is completed when the diluting wastewater ceases to rise in the water column and first begins to spread horizontally.*"

Clearly, application of these regulations require much judgment, such as which oceanographic conditions, currents, density stratification, flow rates, and averaging times are used. These must be carefully chosen and explicitly specified in the outfall design documentation.

Mixing zone water quality standards are usually limited to parameters for acute toxicity protection (sometimes determined by bioassays) and to minimize visual impacts. They are not usually applied to BOD, dissolved oxygen, or nutrients. Bacterial standards are also not normally imposed within or at the boundary of mixing zones unless the diffuser is located near areas of shellfish harvesting or recreational uses.

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APPENDIX E: DISCHARGE SITE CONSIDERATIONS

E.1 General Boundary Conditions and Forcing Functions of *Collision Coasts*

Collision coastal environments are the predominant geomorphic coastal environment in California. These are exposed, open-coastlines that are intrinsically erosional, with steep coastal topography and narrow continental shelves formed by the collision of oceanic techtonic plates with continental plates (Figure E-1). The natural boundaries of these coastal environments are referred to as *littoral cells*, of which there are two general categories based on the amount of sediment cover over the bed rock. *Sandy littoral cells* have abundant sediment cover because they are nourished by coastal streams and rivers, with sandy beaches and moderately sloping shelves, and are bounded in the longshore direction by headlands and submarine canyons. The other collision coastal type is referred to as *rocky littoral cells*. These are nourished by sea-bluff erosion that form pocket beaches, accompanied by tide-pools, rocky reefs, steeply-sloping shelves with limited sediment cover, and are bounded in the longshore direction by headlands, bluffs and rocky out-crops. The geomorphology of both the sandy and rocky collision coastal types creates high-energy coastal environments with vigorous ambient mixing and advection that contributes to rapid dilution and limited dispersion of brine discharge. The high energy in these collision coastal environments in California arises from shoaling ocean waves produced by North Pacific frontal cyclones and Southern Hemisphere storms, wave and wind driven currents and weakly damped tidal currents and internal waves exhibiting numerous high amplitude harmonics arising from trapped oscillations over the shelf bathymetry.

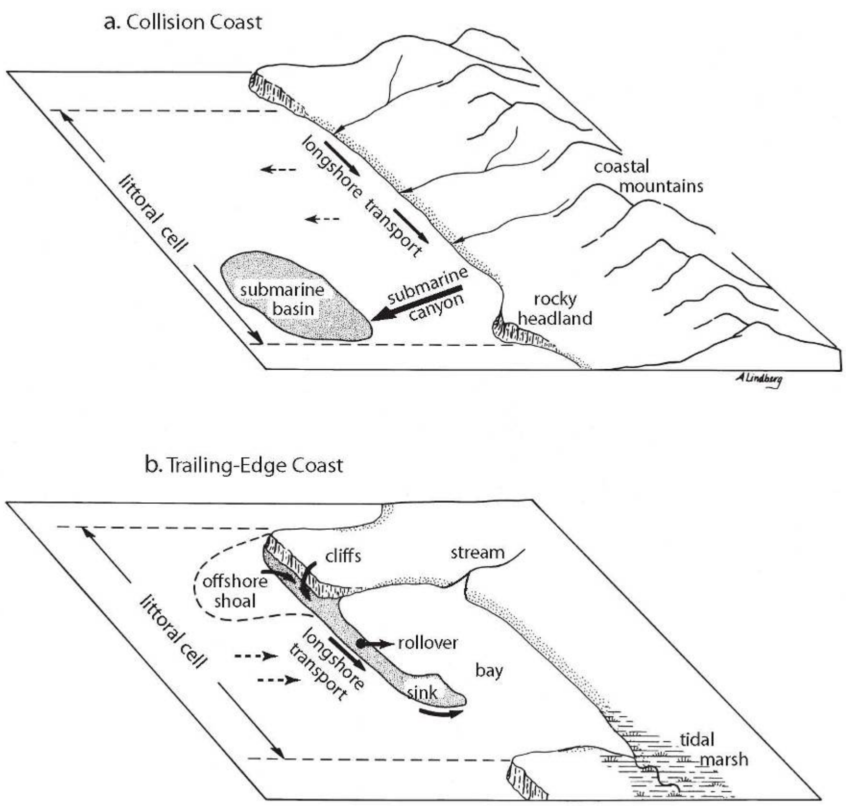


Figure E-1. Geomorphic coastal types.

E.2 Boundary Conditions related to *Far-field Bathymetry, Coastal Structures and Earth-Works*:

Bathymetry exerts a controlling influence on all of the coastal processes that affect dispersion and dilution. The bathymetry consists of two parts: 1) a stationary component in the offshore where depths are roughly invariant over time; and 2) a non-stationary component in the nearshore where depth variations do occur over time. The stationary bathymetry generally prevails at depths that exceed closure depth which is the depth at which net on/offshore sediment transport vanishes. Closure depth is typically -12 m to -15 m MSL for most California wave climate, [Inman et al. 1993]. The stationary bathymetry is typically derived from the National Ocean Survey (NOS) digital database. For the non-stationary bathymetry data inshore of closure depth (less than -15 m MSL) nearshore and beach survey data is typically used, generally provided by the US Army Corps of Engineers.

Because most of the coastline of California is a collision coast it generally has favorable bottom gradients for offshore dispersion of brine discharge, because the narrow continental shelf geomorphology provides steep shelf and nearshore bathymetry. The case of Huntington Beach is a sandy littoral cell, but if it were re-located about 5 miles to the south along the Newport Coast, then the discharge would reside in a rocky littoral cell. Because of the thin sediment cover along the Newport Coast, there are numerous rocky outcrops and reefs offshore, that would present barriers that block the offshore dispersion of brine by gravity. Therefore, discharge sites with bathymetric barriers (offshore rocky reefs and outcrops) should be avoided with negatively buoyant discharges.

Another far-field bathymetric feature to be avoided for negatively buoyant brine discharge are closed form hollow and depressions. These are not generally features found along the exposed open coast of California, (again due to the steep gradient geomorphology of a collision coast), but can be common in embayments, either from natural shoaling effects or from man-induced activities such as the dredging of navigation channels and berthing area. Figure E-2 shows a series of dredged channels and berthing areas in San Diego Bay that create closed depressions significantly deeper than the surrounding native bathymetry. Despite a resonant tidal system with 1-2 knot tidal currents in San Diego Bay, there is very little net transport after multiple tide cycles of a negatively buoyant test particle that serves as a proxy for negatively buoyant brine. In cases where there is little net transport of the brine discharge, a bathymetric depression will fill with brine and displace the lighter ambient seawater from the depression. Such accumulation of brine might lead to increased exposure of benthic organisms to elevated salinity or reduce oxygen exchange with the sediment. The potential for accumulation in local depressions should be considered in the environmental analyses and design.

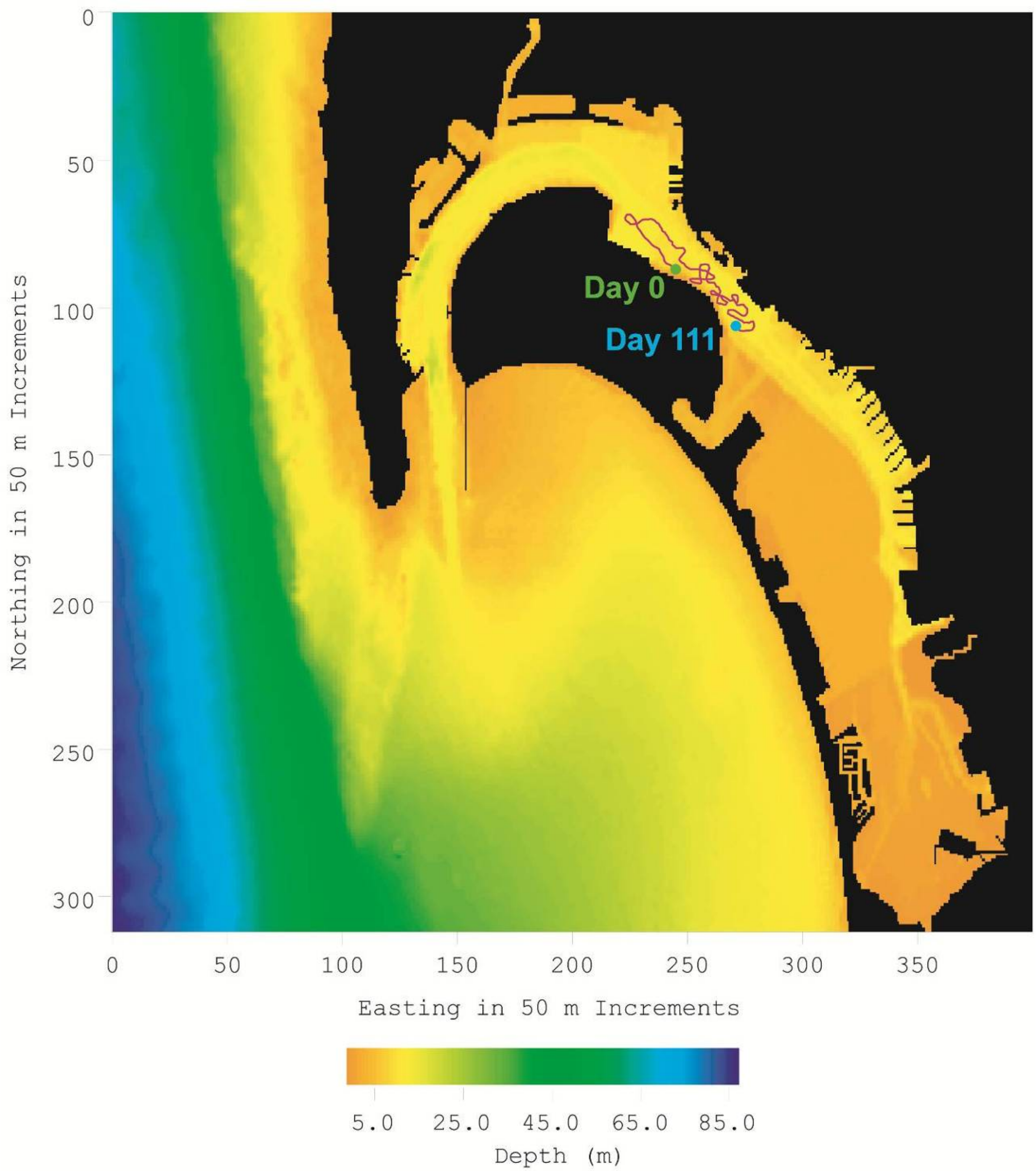


Figure E-2. Bathymetric depressions in San Diego Bay associated with dredged channels and berthing areas for deep-draft ships. Depth gradients indicated by the color bar scale. Transport trajectories of a negatively buoyant particle ($\rho_{brine} = 1.05 g/cm^3$) over 11 days of tidal exchange shown in red.

Bathymetry also exerts a strong influence on the boundaries of littoral cells and on the spatial variability of forcing functions, particularly waves. Figure E-3 shows how bathymetry has partitioned the Southern California Bight into a discrete set of littoral cells and how the bathymetry within those cells and the offshore islands (Channel Islands) has produced distinct refraction and diffraction patterns in the incident wave field throughout the Southern California Bight. (Figure E-3 uses the back refraction calculations of the CDIP data from the San Clemente array after Jenkins and Wasyl 2005). Wave heights are contoured in meters according to the color bar scale and represent 6 hour averages, not an instantaneous snapshot of the sea surface elevation. Note how the sheltering effects of Catalina and San Clemente Islands have induced variations in wave height throughout the Southern California Bight. Diffraction around these channel islands, and refraction over the inner shelf bathymetry concentrates the incident wave energy in certain regions of referred to as "bright spots.", (indicated by red colors in Figure E-3), while it dilutes wave energy in other areas referred to as "shadows" (indicated by blue colors in Figure E-3). The increased wave heights in the bright spots increase the mixing and turbulence generated over the seabed boundary layer, and induces bottom boundary currents (referred to as bottom wind). In addition, bright spots excite vigorous oscillatory wakes around intake and discharge riser structures in the nearfield (Section 6). These effects increase the mixing and dilution rates of the heavy brine that disperse rapidly along the seabed within a bright spots. Conversely, the dark areas in Figure E-3 where wave heights have been diminished (shadows), represent areas of reduced mixing and retarded dilution rates.

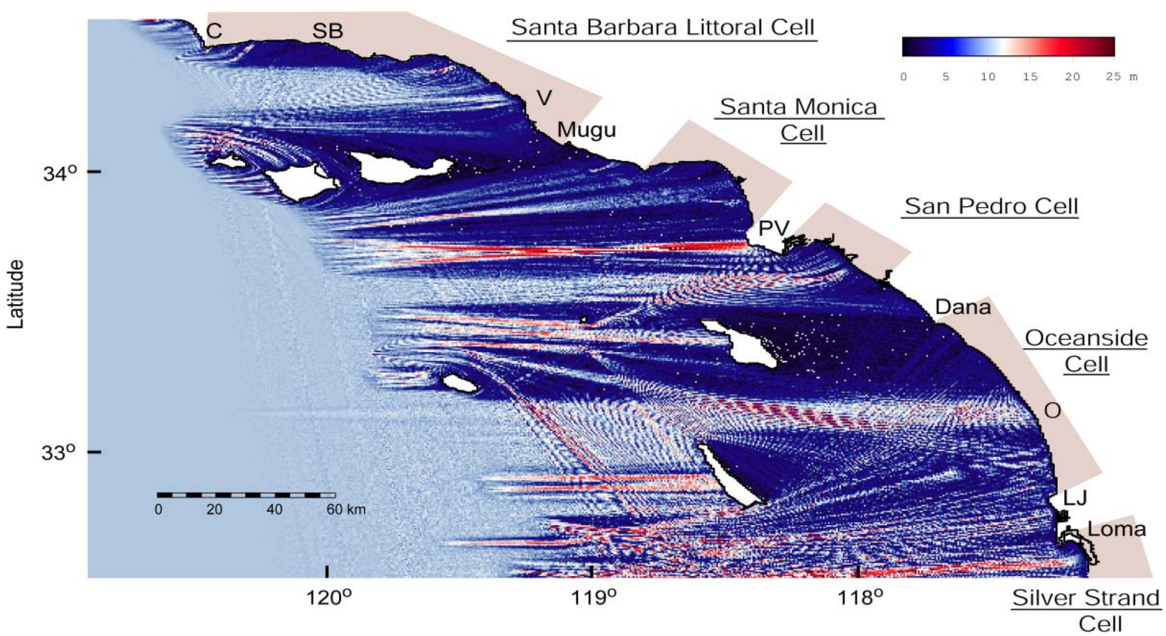


Figure E-3. Wave refraction and diffraction patterns in the Littoral Cells of the Southern California Bight. Sandy Littoral Cells include: Santa Barbara,Santa Monica Oceanside and Siverstrand Littoral Cells. Rocky Littoral Cells are Pacific Palisades Littoral Cell (between Pt Mugu and Santa Monica), and San Pedro Littoral Cell. Also shown are back-refraction pattern of waves measured by San Clemente CDIP station during the storm of 17 January 1988 with 10m high waves at 17 second period approaching the Southern California Bight from 270^0 , (from Jenkins and Wasyl, 2005).

Another aspect of bathymetric influence on wave forcing is in the generation of wave induced currents. In many of the littoral cells in Figure E-3, waves approaching from the west shoal at an angle to the coastline, giving rise to a component of the wave radiation stress directed parallel to the shoreline. In the Santa Barbara littoral, the incident wave radiation stress is directed shore parallel from west to east, giving rise to a general longshore current that flows towards the east. In the Santa Monica, San Pedro, and Oceanside littoral cells, the incident wave radiation stress is directed shore parallel from north to south, giving rise to a general longshore current that flows towards the south. These broad scale longshore currents that persist over entire littoral cells are referred to as *littoral drift*. In addition, there a locally intensified wave driven currents that flow away from bright spots and towards shadows, referred to as *divergence of drift*. When two bright spots are separated by a shadow, the opposing divergence of drift currents flowing into the shadow give rise to a seaward flowing current termed a *rip current*. These wave induced currents are often locally intensified near coastal structures as shown in the far-field hydrodynamic simulation at Oceanside Harbor in Figure E-4. Here the harbor has created a seaward bulge in the bathymetric depth contours, that focuses shoaling waves in a bright spot similar to a point break in surfing. The Oceanside Harbor breakwater also intercepts the littoral drift and deflects it seaward forming a rip current. The rip current converges with the general southward drift causing divergence of drift that locally intensifies the southward drift in the waters seaward of the harbor. As the intensified southward drift flows past the harbor, a "backwater eddy" is formed along the down-drift reach of coastline. These bathymetric and structurally induced effects on local waves and currents create an idea brine disposal site where both mixing and advection of brine discharge can be maximized. An example of this can be seen in the farfield brine dilution ratios calculated in Figure E-5 on the seabed for a potential desalination project sited at a similar harbor setting at Redondo Beach CA.

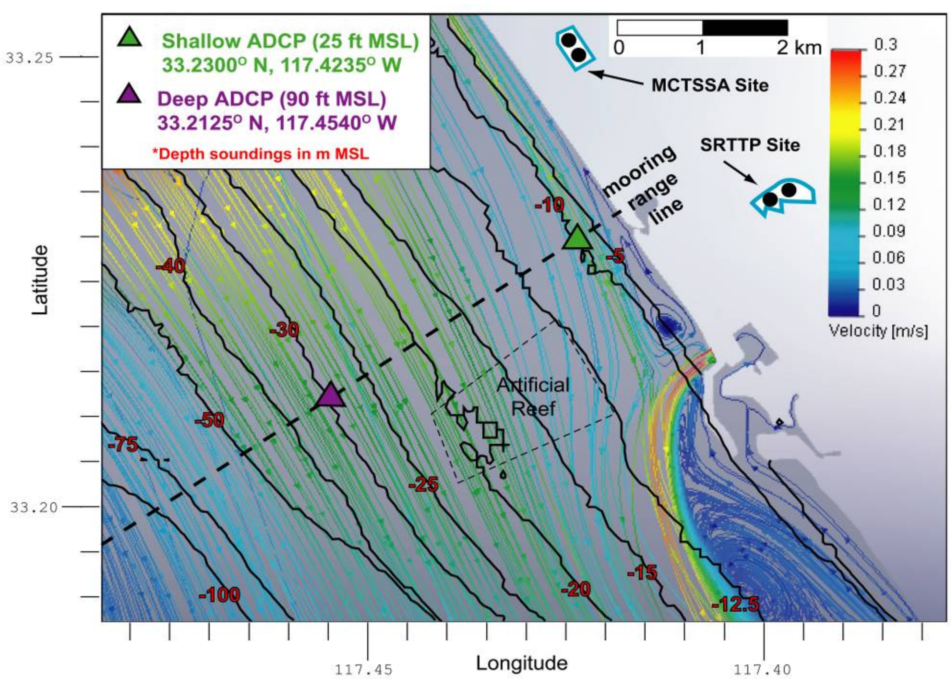


Figure E-4: Wave-induced longshore currents and rip currents, superimposed on ebb-tide at Oceanside Harbor, CA. (from Jenkins, 2011).

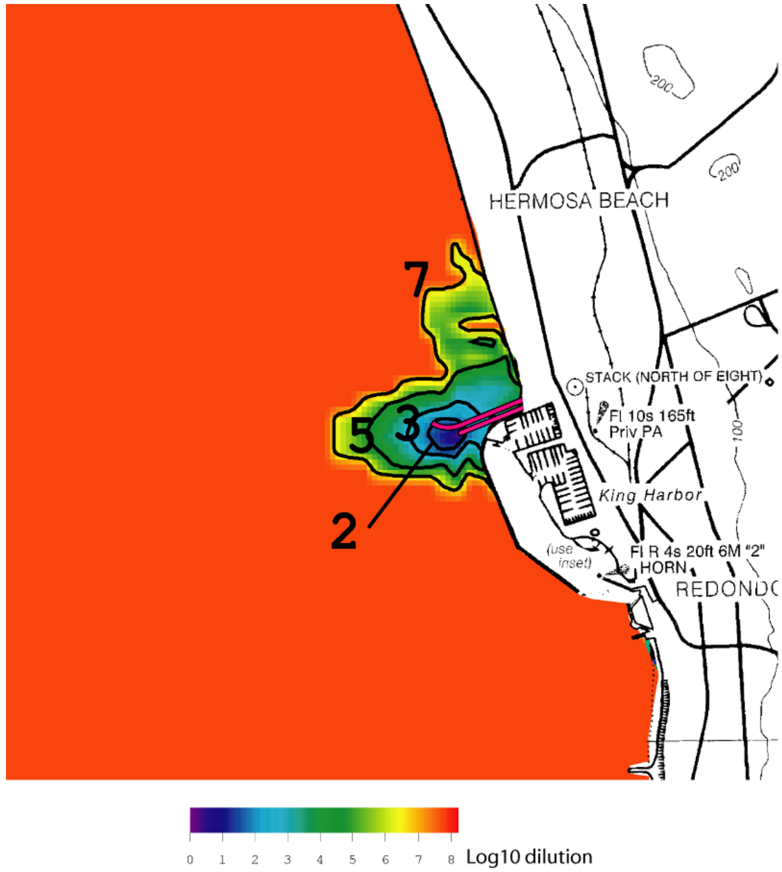


Figure E-5. Dilution of brine on the seabed as a result of mixing and advection intensification at the Redondo Beach King Harbor, after West Basin Desalination Demonstration Facility, 2010.

Here, brine discharge from a legacy power plant discharge riser located very close to shore, is deflected seaward by the Redondo Beach King Harbor jetty system intercepting the littoral drift. The result is seaward dispersion of the brine and very rapid dilution, with minimum dilution at the beach reaching at least 10,000 to 1, increasing rapidly to 10^{6}- 10^{7} to 1 as one proceeds up-coast to the northwest away from the harbor. This example illustrates that discharge sites with high ambient mixing and advection (typical of exposed, open-ocean, collision-coastlines) are preferable, particularly when siting near coastal structures will give rise to intensification of ambient mixing and advection.

E.3 Climate effects on Wind and Wave Forcing Functions

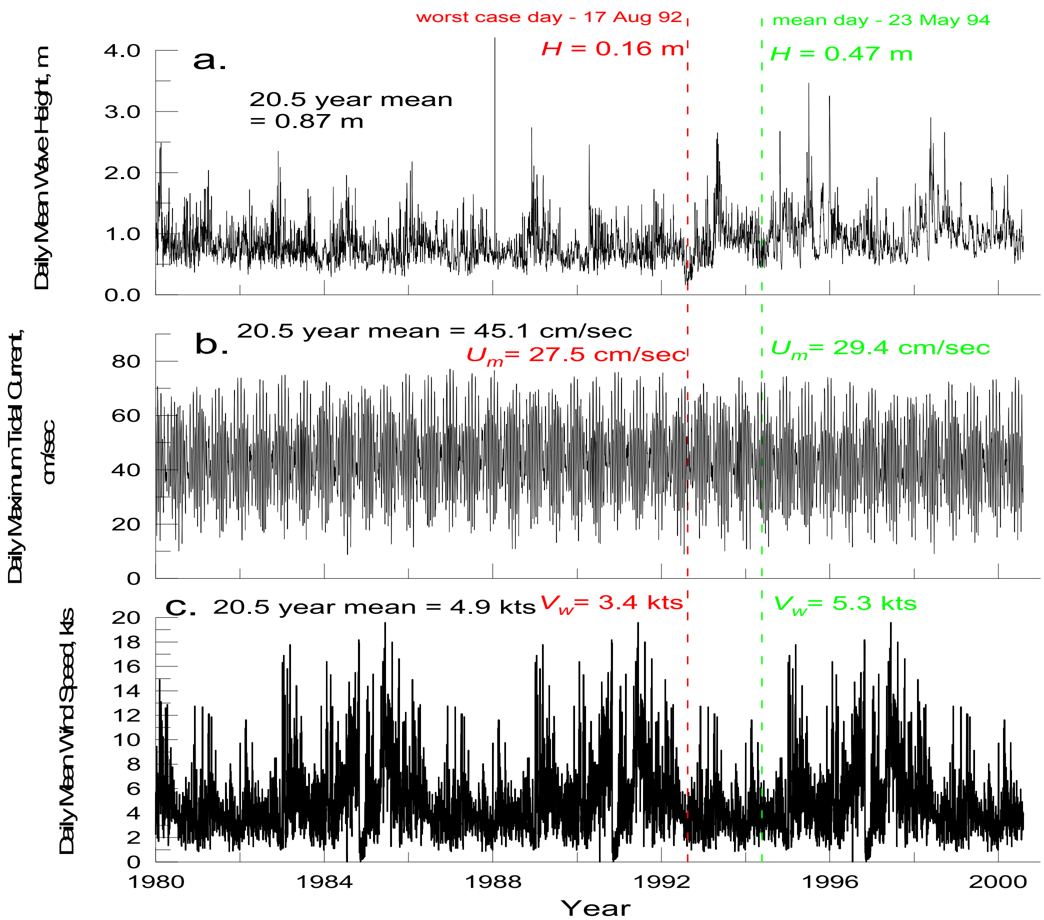
The advective and diffusive fluxes of the far-field brine dilution and dispersion processes in the nearshore are influenced by ocean temperature, salinity and the wave climate. Upon occasion, the typical seasonal weather cycles are abruptly and severely modified on a global scale. These intense global modifications are signaled by anomalies in the pressure fields between the tropical eastern Pacific Ocean and Australia/Malaysia known as the ***Southern Oscillation***. The intensity of the oscillation is often measured in terms of the ***Southern Oscillation Index (SOI)***, defined as the monthly mean sea level pressure anomaly in mb normalized by the standard deviation of the monthly means for the period 1951-1980 at Tahiti minus that at Darwin, Australia. The Southern Oscillation is in turn, modulated over multi-decadal periods by the ***Pacific Decadal Oscillation***, which results in alternating decades of strong and weak El Niño

The potential impact of variations in ocean temperature, salinity and waves can be evaluated by examining conservative or worst case scenarios. The worst case can be described by searching long-term records for historical events relevant to the discharge site that match criteria for worst-case. The criteria for worst case are based on the simultaneous occurrence of the high salinity and temperature in the receiving water during periods of low mixing and advection in the local ocean environment. The low mixing/ advection conditions arise during periods of benign weather when waves are small and winds and waves are close to stagnation. The environmental conditions are combined with worst case operating scenarios that give lowest in-the-pipe dilution of discharge constituents from a desalination facility. Table E-1 gives an example of the worst case criteria applied to each controlling variable in the computer search of the historic record for a discharge site in Huntington Beach, CA.

Table E-1. Search criteria for worst case scenario.

Variable	Search Criteria	Ecological Significance
Co-located Plant Flow Rates	Minimize	Lower flow rate results in less initial dilution in the pipe of the constituents from desalination
Ocean Salinity	Maximize	Higher salinity leads to higher initial concentrations of sea salts and backwash constituents from desalination
Ocean Temperature	Maximize	Higher ocean temperature leads to higher density contrast between receiving water and discharge
Ocean Water Levels	Minimize	Lower water levels result in less dilution volume in the nearshore and consequently lower dilution rates
Waves	Minimize	Smaller waves result in less mixing in bottom boundary layer of shoaling zone, weaker oscillatory vortices shed from discharge riser, weaker wave-induced currents, and consequently less near-bottom dilution
Currents	Minimize	Weaker currents result in less advection and less offshore dilution
Winds	Minimize	Weaker winds result in less surface mixing and less dilution in both the inshore and offshore

For the Huntington Beach example, minimum ocean mixing levels were obtained Figure E-6 from a computer search of 24 year long records of winds, waves and currents. However, the highest ocean salinity during the event day when minimal mixing conditions prevailed was 33.49 ppt, not the salinity maximum of 34.3 ppt identified in Figure 5.2. This is due to the fact that salinity maximums are mutually exclusive with mixing minimums. Salinity maximums are caused by vigorous southerly winds that create a well-mixed coastal ocean while pushing high salinity water masses along the California coast. A series of sensitivity analyses determined the salinity maximum might increase the concentration of brine discharge by 2%, but that this effect is offset by a reduction in far-field dilution caused by the effects of retarded mixing during low energy conditions. In fact the dilution rates for the mixing minimum are 99% smaller than the dilution rates during the salinity maximum. Therefore, minimal ocean mixing conditions became the dominant set of environmental variables in defining the worst case scenario. Accordingly worst case dilution modeling is based on the set of worst-case forcing parameters annotated in the example in Figure E-6.



Controlling environmental variables for brine dilution, mixing variables: a) daily mean wave height, b) daily maximum tidal current velocity, and c) daily mean wind.

Figure E-6. Minimal ocean mixing conditions for worst case discharge scenarios from forcing function climate minimums. (from Huntington Beach Desalination Project, SEIR, 2010).

E.4 Salt Wedge Sediment Dynamics Effects on *Boundary Conditions and Forcing Functions for Estuarine Embayments*

Estuarine embayments generally present very tricky site discharge scenarios. To illustrate the hydrodynamic and sediment dynamic issues related to discharging brine in these types of environments, the details of the Sacramento Delta section of the San Francisco Bay estuary are examined. The source water for this example is obtained from a channel that branches off the Suisun Bay (Figure E-7). The source water flow from Suisan Bay is due primarily to tidal exchange and Suisun Bay is also the receiving water for the brine discharge. Circulation in Suisan Bay is a complex salt wedge system driven by tidal exchange between Suisun Bay and San Pablo Bay and discharge from the Sacramento River. Therefore both the source water and receiving water would be brackish and sediment-laden and these characteristics will very daily in response to the spring-neap tidal variability, and seasonally with variation in the Sacramento River discharge.

Figure E-7 presents a composite of a Google Earth image of this site with a brine plume simulation overlaid. The plume simulation is based on jet dynamics, sedimentation, scour and burial after Jenkins et al (1992; 2007) and on algorithms for flocculation and shear stress dynamics after Aijaz and Jenkins (1993; 1994). The simulation uses salinity and flow rates of the Sacramento River based on the USGS gage station #11455420. The simulation in Figure E-7 illustrates the potential for the high salinity brine to induce flocculation of the sediment load of the Sacramento River in the neighborhood of the discharge, causing local increases in sediment deposition rates in the navigation channel of the Sacramento River and over adjoining mud flats along the river banks. Both of these alterations in the depositional features of the receiving water have potentially adverse environmental impacts, since increased sediment deposition in the navigation channel would interfere with ship traffic and increase dredging requirements along with those related impacts; while increased deposition in the mudflats would impact existing intertidal wetland habitat.

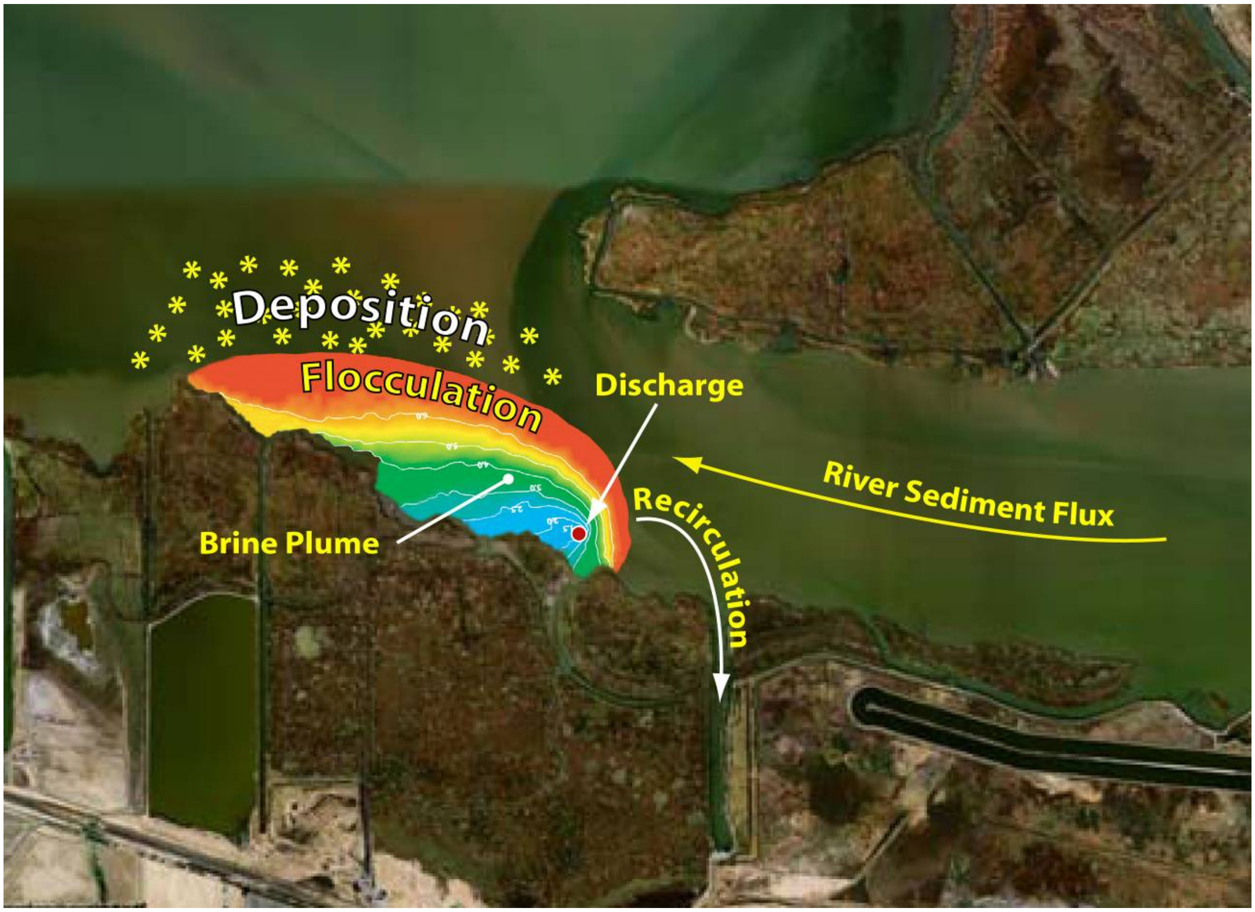


Figure E-7. Simulated brine discharge plume in the lower Sacramento River Delta. Flocculation convergence zone indicated in red, deposition zone indicated as yellow stars.

References:

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APPENDIX F: MATHEMATICAL MODELING

1. Introduction

The effluent may be negatively or positively buoyant as it enters the ocean, depending on whether the discharge is raw concentrate, or blended with power plant cooling water or domestic wastewater. Modeling positively buoyant submerged and shoreline discharges has been discussed in many publications so we do not consider them further here.

Shoreline negatively buoyant discharges (Figure 6-1a) will result in a density current that flows down the bottom slope. Because the resulting density stratification inhibits vertical mixing, dilution is relatively small and benthic organisms will be exposed to relatively high salinities. Shoreline disposal of pure concentrate by this means is therefore discouraged and is not considered further here.

The hydrodynamic mixing regions of wastewater discharges are usually considered in two phases: The near field and the far field. The distinctions are discussed in more detail in Appendix D, but briefly in the near field mixing and dilution is rapid and results from processes induced by the discharge itself, such as turbulent entrainment, whereas in the far field mixing is due to natural oceanic turbulence. Some authors include a mid-field characterized by dynamical spreading as a density current.

Near field processes operate over fairly small scales: distances of order tens of meters and times of order minutes. The far field is dominated by larger scales: distances of order hundreds of meters to kilometers and times of order hours to days. The rate of mixing in the far field is much slower than in the near field.

Because of the wide range in length and time scales, it is generally not possible to capture them all in one model, so separate near and far field models are usually employed. The far field models are probably two or three dimensional hydrodynamic models of the coastal waters. The two models must be coupled to predict the overall brine dispersion, with the output from the near field model becoming the input to the far field model.

In this Appendix we consider modeling of negatively buoyant discharges from diffusers. We first discuss some overall concepts, then near field models, then far field models. Coupling the two models together is then discussed. Finally, we discuss simple mass-balance box models which are useful tools to assess flushing and potential background build-up of contaminants. Much of the material in this Appendix is adapted from Roberts et al (2010b).

2. Characteristics of Negatively Buoyant Discharges

In order to effect high dilution of negatively buoyant effluent it will be necessary to discharge it as high velocity jets through a diffuser that effects rapid mixing by entrainment (Figure F-1). Because the jets are dense, they reach a terminal rise height and then fall back to the seabed where they spread as a density current. The highest salinity on the seabed occurs where the centerline of the jet impacts the seabed. The dilution at this point is labeled S_i (for impact point dilution) on Figure F-1. Additional dilution occurs beyond the impact point before the flow collapses under the influence of the induced density stratification. The point where this collapse occurs is the end of the near field, and the dilution at this point is the near field dilution. The length of the near field is denoted by x_n in Figure F-1 and the near field dilution is S_n. Typically, near field dilutions are of order 60% higher than impact dilutions (Roberts et al, 1997). The length of the near field is of order a few tens of meters.

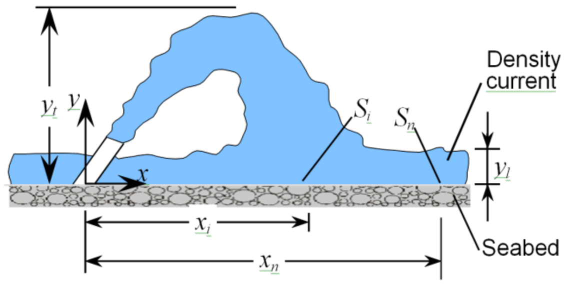

Figure F-1. Schematic depiction of brine discharge as inclined jet

For multiport diffusers, such as the one shown in Figure 6-3, merging of the individual jets and the concomitant reduction in dilution must also be considered.

Figure F-1 shows details of the different flow regions: the ascending jet phase, terminal rise height, descending jet phase, seabed impaction and transition to horizontal flow, mixing in the density current, and finally the far field.

3. Near Field Modeling

There are three main techniques for predicting the near fields of brine discharges: 1) Physical modeling using scaled laboratory models, 3) Semi-empirical equations, and 3) Numerical modeling.

3.1 Physical modeling

Physical modeling is employed primarily for predicting near field behavior. It consists of laboratory experiments using scale models that simulate the particular case being tested at a smaller scale. Tests can be carried out on any effluent, discharge configuration, and ambient conditions. For discussions of physical modeling, see Ettema et al. (2000).

The model and the prototype maintain the relative proportions (the scale factor) and are scaled in terms of both geometry and forces. In order to guarantee the correspondence between the model and the prototype behavior, the model must satisfy:

1. **Geometric similarity** where the ratio of all corresponding dimensions in the model and prototype are equal. This is commonly referred to as an undistorted model.

2. **Dynamic similarity** where the ratios of all forces in the model and prototype are the same. The main forces are inertia, gravity, and viscous forces, and their ratios are generally expressed in terms of dimensionless numbers. The ratio between inertia and viscous forces is determined by the Reynolds number. If its value is sufficiently high, as will always be the practical case, the flow is fully turbulent and viscous forces can be neglected. The brine behavior then depends

mainly on the ratio of inertial to buoyancy forces, which is expressed by the densimetric Froude number.

3. **Kinematic similarity** is equality of ratios of speeds and velocities at similar points. But if conditions 1 and 2 above are satisfied, kinematic similitude automatically follows.

Physical modeling is particularly useful where mathematical models are not verified or uncertain, such as merging multiple jets, discharges from multiport rosettes (for example, Figure 6-3), or the effects of ambient currents. Their disadvantages are that they may be relatively expensive and it is less easy to simulate a wide variety of alternatives. Examples of physical modeling of concentrate diffusers are given in Miller and Tarrade (2010), Tarrade et al. (2010), and Miller (2011).

3.2 Semi-Empirical Equations

Experimental studies of dense jets with the common design of a 60° orientation has resulted in semi-empirical equations that are widely used for diffuser design with single (or non-merging) jets. For example, in stationary environments, (Pincince and List, Roberts and Toms and others):

$$\frac{S_i}{F} = 1.6; \qquad \frac{S_n}{F} = 2.6; \qquad \frac{y_t}{dF} = 2.2; \qquad \frac{x_n}{dF} = 9.0 \tag{1}$$

Where (Figure F-1) S_i is the impact dilution, S_n the near field dilution, y_t is the terminal rise height, d the nozzle diameter, x_n the length of the near field, and F is the densimetric Froude number defined as:

$$F = \frac{u}{\sqrt{g_o' d}} \tag{2}$$

where $g_o' = g(\rho_o - \rho_a)/\rho_a$ is the initial value of the modified acceleration due to gravity, and g is the acceleration due to gravity, ρ_o is the effluent density ρ_a the receiving fluid density d the nozzle diameter and u the jet exit velocity. The values of the constants in Eq. 1 are taken from Roberts et al. (1997) and have been widely used in brine diffuser designs.

3.3 Numerical Modeling

The equations (1) will often suffice for estimating the major flow characteristics of non-merging 60° inclined jets into stationary environments. For other cases, for example other orientations, merging jets, effects of currents, or effects of bottom slopes, numerical models are now frequently employed.

Near field predictions are usually made by entrainment models or computational fluid dynamics models (CFD). However, as will be discussed, present numerical models cannot accurately simulate all flow features within a single model configuration, especially the effects of currents and jet merging.

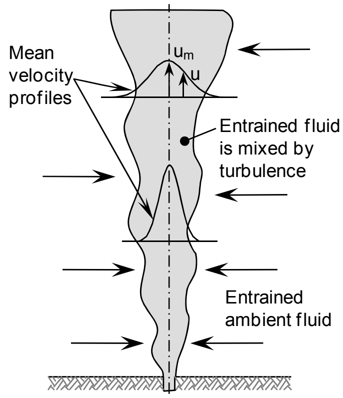

Figure F-2. Schematic view of the exit velocity along the axis of a turbulent plume and the direction of entrainment of ambient fluid into the plume.

3.3.1 Entrainment Models

Entrainment models are the most common tool for engineering analyses of jet and plume-type flows such as brine discharges.

The entrainment hypothesis was first suggested by Morton et al. (1956) and has since been applied to a variety of engineering and natural flows, as reviewed in Turner (1986). It is particularly relevant here as it has found great utility for predicting the jet and plume-type flows typical of ocean discharges. Below we summarize the essential features and limitations of these models; for details, the original references should be consulted, and for recent extensive reviews of entrainment models, see Jirka (2004, 2006), and Roberts et al (2011).

The concept of entrainment, as applied to a simple round rising plume in a stationary environment, is shown in Figure F-2.

The rising plume entrains external fluid that then mixes with and dilutes the plume fluid. The entrainment hypothesis (Fischer et al. 1979) states that fluid is entrained at the plume radius b with a velocity u_e that is proportional to the mean centerline velocity, u_m:

$$u_e = \alpha u_m \tag{3}$$

where α is the entrainment coefficient (whose value is different for jets and plumes). The rate of change of volume flux Q in the plume with distance s is then given by:

$$\frac{dQ}{ds} = 2\pi\alpha b u_m \tag{4}$$

Eqs. 3 and 4 are the essence of the entrainment hypothesis, and form the basis for most entrainment models.

Although entrainment models can be used for predicting dense jets, they are subject to a number of limitations and should be used with some caution.

Integral models assume incorporation of external fluid into the jet by entrainment and the profiles of velocity and tracer concentration to be self-similar and axially symmetric. Experimental jets often violate these assumptions, however, leading to unreliable predictions. For example, Pincince and List (1973) concluded that, although jet trajectories were reasonably predicted, dilutions were considerably underestimated. Anderson et al. (1973) concluded that the models can only predict trends, rather than exact dilutions and trajectories.

The vertical asymmetry in the tracer profiles, whereby the peak concentration is closer to the top, has been observed in many previous studies of dense jets in crossflows and inclined jets in stationary environments. Lane-Serff et al (1993) point out that the top half of the jet is gravitationally stable, with density decreasing upwards, but the bottom half is unstable, with heavier fluid above lighter fluid. This leads to the upper plume edge being sharp and well-defined, but in the lower half fluid can detrain from the jet so the lower boundary is poorly defined. Lindberg (1994) also noted in his experiments with crossflows that low momentum fluid almost immediately descended after leaving the nozzle and this continued through the jet trajectory, and Kikkert et al (2007) observed it in stationary inclined jets. This gravitational instability also leads to enhanced mixing within the jet and also between the jet and the environment.

Integral models usually do not include the additional mixing that occurs in the near field beyond the jet impact point. For inclined jets in stationary environments, Roberts, et al. (1997) find the increase in dilution between the impact point and the end of the near field to be around 60%.

At low current speeds the bottom layer forms an upstream wedge that is expelled at higher speeds. The length of the arrested wedge depends on hydrodynamic drag at the head and interfacial friction over the length of the wedge.

Merging jets from multiport diffusers result in further complications. In particular, the jets entrain, or attract, each other, sometimes called the Coanda effect. If the jets are too close together, the supply of entraining water is restricted resulting in reduced dilution. In general, entrainment models cannot predict the Coanda effect, which reduces jet rise height and dilution. For these cases, physical modeling will be more reliable.

Some common models that have been are widely used for predicting jet and plume-type flows, including dense brine discharges are Cormix, Visual Plumes (UM3), and VisJet. For a recent extensive discussion and comparison of these models for simulating dense jets in stationary environments, see Palomar et al. (2012ab).

3.3.2 CFD

Computational fluid dynamics (CFD) modeling is being increasingly applied to a wide variety of turbulent flows in nature and engineering. There are several major CFD techniques; for a review, see Sotiropoulos (2005).

One method is direct numerical simulation (DNS). The unsteady, three-dimensional Navier-Stokes equations are solved over scales small enough to resolve the entire spectrum of turbulence. In principle, DNS could model turbulent flows with virtually no modeling uncertainties but because it requires extensive computational resources it has been mainly applied to relatively simple, low Reynolds number flows. DNS is therefore not yet a practical modeling tool for simulating engineering-relevant flows.

A more realistic approach is Large Eddy Simulation (LES). The spatially filtered unsteady Navier-Stokes equations are solved to resolve motions larger than the grid size, and smaller-scale motions are modeled with a sub-grid model. For high Reynolds number flows of practical engineering interest, however, very high grid resolutions and supercomputers are still required.

The most common CFD models are Reynolds-decomposition models. Flow quantities are decomposed into time-averaged and fluctuating values and the Navier-Stokes equations are then time averaged, producing Reynolds-averaged Navier-Stokes (RANS) equations. Assumptions are made about the new terms that arise from this averaging. Probably the most common is the k-ε model that assumes an empirical relationship between turbulent kinetic energy, k, and the rate of energy dissipation, ε.

There have not been many applications of CFD to jet and plume-type flows. Hwang and Chiang (1995) and Hwang et al. (1995) simulated the initial mixing of a vertical buoyant jet in a density-stratified crossflow. They employed a RANS model with a buoyancy modified k-ε model. Blumberg et al. (1996) and Zhang and Adams (1999) used far-field CFD circulation models to calculate near field dilutions of wastewater outfalls. Law et al. (2002) used a revised buoyancy-extended k-ε turbulence closure to investigate the dilution of a merging wastewater plume from a submerged diffuser with 8-port rosette-shaped risers in an oblique current. Davis et al. (2004) used the commercial codes ANSYS and FLUENT to simulate several case studies of effluent discharges into flowing water, including a line diffuser, a deep ocean discharge, and a shallow river discharge. They concluded that CFD models are becoming a viable alternative for diffuser discharges with complex configurations.

The paucity of CFD applications to near field mixing is because of the major challenges that they face. These arise from the geometrical complexity of realistic multiport diffusers, the large difference between port sizes and the other characteristic length scales, buoyancy effects, plume merging, flowing current effects, and surface and bottom interactions. To overcome these difficulties, Tang et al. (2008) applied a three-dimensional RANS model using a domain decomposition method with embedded grids to model diffusers.

Although promising, the complexity of CFD models, the effort required to set them up, and long run times suggests that entrainment and length-scale models will continue to be used for many years.

CFD models of brine discharges have been reported by Muller et al (2011) and Seil and Zhang (2010).

4. Far Field Modeling

Hydrodynamic models of coastal circulation are being increasingly used to predict the fate and transport of coastal discharges in the far field and potential build-up of salinity in the vicinity of the discharge. For further discussion of far field hydrodynamics models see Roberts et al. (2010b).

Most models have been two-dimensional (depth-averaged) which is probably adequate for fairly shallow unstratified waters. But in deeper waters, especially if there are wind-shear effects, baroclinic processes, and density stratification, three-dimensional models are needed. In contrast to near field models, far field hydrodynamic models require extensive data input. These include currents, bathymetry, winds, density stratification, tides, and their spatial and temporal variability. The models are either finite element, finite difference, or finite volume, of which finite difference is the most common. The models should be combined with field studies to ensure reliable results.

Ocean circulation models can be combined with mass transport models to predict contaminant transport. Examples are bacteriological pollution in nearshore areas due to storm water runoff (Carnelos 2003) and marine outfalls during different flow conditions such as flood and ebb tides (Liu et al. 2007). Hydrodynamic models have also been used to predict near field plume behavior (Blumberg et al. 1996; Zhang 1995).

Some commonly used ocean circulation models are Delft3D, POM, ECOM, ROMS, Mike3, Telemac, and Elcom. These models are applicable to oceans, coastal waters, lakes, rivers, and estuaries. Some are commercial and some are open source (free).

Most models assume incompressibility and are hydrostatic and Boussinesq, so that density variations are neglected except where they are multiplied by gravity in the buoyancy force terms. The basic equations are based on continuity, momentum, and thermodynamics including temperature and salinity, and an equation of state.

Three-dimensional models are probably needed for waters deeper than about 30 m or so that are stratified. This is because the currents can be strongly sheared, not only flowing at different speeds over depth but in different directions also; two-dimensional models would not capture this variability. But for reliable results, three-dimensional models require extensive data on currents and density at the boundaries and intensive efforts to set up and verify. For these reasons they are not commonly used for smaller outfall projects, but may be part of larger ones.

Due to computational restrictions, it is usually not practical to model an area large enough that the area of interest is independent of the boundary conditions. Therefore, a common approach is to model a large area with a coarse grid and to embed a finer-scale model within it. The grid size of the smaller model is small enough to resolve scales of interest to outfall dispersion. The fine-grid model derives its boundary conditions from the larger model and is said to be nested within it.

5. Model Coupling

Coupling the near and far field models involves transfer of flow quantities, such as volume, momentum, and pollutant mass between them, possibly in both directions. This is illustrated in Figure F-3.

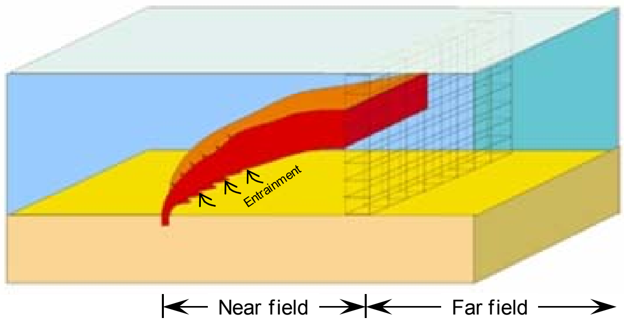

Figure F-3 Model coupling (after Bleninger 2006)

The near field dynamics are characterized by entrainment and small-scale turbulence. The jets entrain fluid that induces a current around the diffuser. This will usually be a few cm/s and its magnitude decreases with distance from the diffuser so it will generally be negligible compared to ambient currents. Therefore, typical outfalls do not significantly affect coastal circulation patterns (this may not be true for large cooling water discharges from power plants). The coupling is therefore usually considered to be one way, i.e. local currents affect the discharge, but not vice versa.

Bleninger (2006) describes an approach in which output from the near field model CORMIX is linked to a far field hydrodynamic model, Delft3D. Bleninger assumes passive, i.e. one-way, coupling. The source is introduced into the far field grid cells as a volume flux that is equal to the source volume flux multiplied by the near field dilution with a contaminant concentration equal to the source concentration divided by the near field dilution. Although this preserves the contaminant mass flux, it does not satisfy volume continuity as the entrained flow is not removed from any cells. As discussed above this is usually a good assumption for marine wastewater outfalls.

Other examples include Chin and Roberts (1985) who coupled a near field model with a far field particle tracking model. Zhang (1995) discusses different means of introducing the effluent into the far field grid. Connolly et al. (1999) used a hybrid modeling approach to predict bacterial impacts from outfalls in Mamala Bay, Hawaii. They used ECOM to simulate advective and dispersive processes in the bay. The predicted near field characteristics were directly inputted into grid cells at the predicted plume rise height following the methodology of Zhang and Adams (1999).

Dynamic, i.e. two-way, linkage between the near and intermediate fields was addressed by Choi and Lee (2007). They applied a distributed entrainment sink approach (DESA) to model the intermediate field by coupling a 3D far field model with a Lagrangian near field model (JETLAG). The action of the plume on the surrounding flow is modeled by a distribution of sinks along the jet trajectory. This establishes a two-way dynamic link at grid cell level between the near and far field models.

Suitable coupling between the near and far field models is essential for reliable prediction of impacts. If near field dilution is not accounted for, predicted far field dilutions will be much too low, leading to considerable overestimates of environmental impacts.

6. Box Models

The "background" mean concentration field near the diffuser is governed primarily by flushing due to the mean drift, horizontal diffusion (and, for non-conservative substances, chemical and biological decay). One approach to predicting the physical dilution caused by these processes is to estimate it from a solution to the two-dimensional diffusion equation (Csanady 1983a; Koh 1988). Another is a mass-balance box model (Csanady, 1983b), which is a useful and simple way to assess coastal "flushing" and the relative orders of magnitude of the various processes. The box model is shown in Figure F-4.

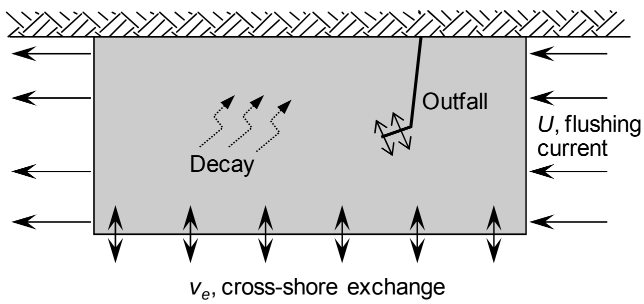

Figure F-4 Box model for estimating long-term buildup of contaminants (after Csanady 1983b)

Tidal currents distribute the effluent over an area, or "box" whose dimensions are approximately equal to the tidal amplitude. These dimensions are approximately $X = u_t T/2$ and $Y = v_t T/2$, in the alongshore and cross-shore directions, respectively, where u_t and v_t are the amplitudes of the tidal currents, and T is the tidal period. Csanady (1983b) calls this area the "extended source region."

Long-term average current speeds are usually much slower than instantaneous values. They lead to an average dilution equal to UhY/Q, where Q is the effluent flowrate, h the average depth of the plume over the extended area, and U the long-term average "flushing velocity."

This can be extended to include the other processes by applying a mass balance to the box. This yields a "long-term average dilution" S_p:

$$S_p = \frac{UhY}{Q} + \frac{v_e hX}{Q} + \frac{khXY}{Q} \qquad (8)$$

The first term on the right is the dilution due to flushing by the mean current. The second is dilution due to cross-shore mixing which is parameterized by v_e, a mass transfer "diffusion velocity," that can be assumed equal to the standard deviation of the cross-shore tidal fluctuations (probably an underestimate). The third term is "dilution" due to chemical or biological decay, where k is a first-order decay rate. The total effective dilution is the sum of these individual dilutions.

Consider a typical problem. Suppose we have a discharge Q = 4 m^3/s into a tidal current whose alongshore amplitude is u_t = 0.25 m/s, and cross-shore amplitude is v_t = 0.08 m/s, and cross-shore rms velocity is v_e = 0.04 m/s. Suppose the average current speed (the flushing velocity) is U = 0.06 m/s. For a semi-diurnal tide, the period T is about 12 hours. Suppose further that the average depth (thickness) of the wastefield is 4 m.

Then the extended source area (size of the box in Figure F-4) is:

$$X = u_T T / 2 = 0.25 \times 12 \times 3600 / 2 \approx 5{,}400 \text{ m} \approx 5.4 \text{ km} \quad \text{and}$$

$$Y = v_T T / 2 = 0.08 \times 12 \times 3600 / 2 \approx 1{,}700 \text{ m} \approx 1.7 \text{ km}$$

and the dilutions for a conservative substance are:

$$\text{Due to the mean current:} \quad \frac{UhY}{Q} = \frac{0.06 \times 4 \times 1700}{4} \approx 100$$

$$\text{Due to cross-shore exchange:} \quad \frac{v_e hX}{Q} = \frac{0.04 \times 4 \times 5400}{4} \approx 220$$

The total effective dilution, the sum of these dilutions, is about 320.

These are obviously only approximate order of magnitude calculations, but they are very useful for estimating long-term impacts. They can be applied to other substances such as toxic materials to estimate their potential accumulation.

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