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Hypolimnetic Anoxia and Sediment Oxygen Demand in California Drinking Water Reservoirs

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ABSTRACT

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Summertime hypolimnetic anoxia can occur in productive drinking water reservoirs as a result of the decay of phytoplankton. Anoxic conditions promote ecological processes that degrade water quality through the release of problem-causing compounds from anoxic sediments including phosphates, ammonia, sulfides, methylmercury, iron and manganese. Hypolimnetic aeration systems are commonly installed in reservoirs to prevent hypolimnetic anoxia, but these systems have been historically undersized due to an underestimation of the magnitude of oxygen demand in the hypolimnion. To gain insight into the sizing of hypolimnetic aeration systems, this study evaluated the effects of water current and DO concentration near the sediment-water interface on sediment oxygen demand (SOD) in nine California drinking water reservoirs of various size (5-220 million m^3) and trophic status (mean annual chlorophyll *a* of $0.5\text{-}11 \mu\text{g} \cdot \text{L}^{-1}$). SOD measured under quiescent conditions in 1.8 L experimental chambers ranged from $0.1\text{-}0.8 \text{ g} \cdot \text{m}^{-2} \cdot \text{d}^{-1}$. Currents near the sediment-water interface of $3\text{-}8 \text{ cm} \cdot \text{s}^{-1}$ induced a two to four-fold increase in SOD, and resulted in a shift from first-order to zero-order DO uptake by sediment with respect to DO concentration in overlaying water. Results support the diffusive boundary layer model for SOD, with increased DO concentration and currents resulting in a larger SOD since there is a greater diffusional driving force across a smaller diffusive boundary layer. The study also evaluated the effects of trophic status and morphometry on hypolimnetic anoxia at the nine study sites. A number of significant correlations were discovered between factor quantifying hypolimnetic anoxia (areal and mass based hypolimnetic oxygen demand, SOD) and those quantifying morphometry (mean depth of the hypolimnion, volume of the hypolimnion) and trophic status (mean annual chlorophyll *a*). These results suggest that both increased size of the hypolimnion and higher productivity lead to higher oxygen demand within the hypolimnion. In addition, shallower reservoirs had a larger fraction of their total oxygen demand exerted in the sediments versus the water column. As a result, increased mixing at the sediment-water interface after start-up of aeration systems, and the resulting stimulation of SOD, will be particularly important in productive reservoirs of moderate depth (mean depth of 10-15 m). Aeration systems should be designed to enhance SOD by maintaining high oxygen concentrations and by inducing currents at the sediment-water interface. This will increase the depth of penetration of DO into sediment and promote beneficial aerobic biogeochemical reactions in surface sediments. Aeration systems that utilize pure-oxygen with horizontal discharge of highly oxygenated water across the sediment surface, rather than the traditional air-lift aeration system, will be more successful in satisfying SOD and improving hypolimnetic water quality.

Key Words: hypolimnetic anoxia, hypolimnetic aeration, sediment oxygen demand, reservoir.

Summertime hypolimnetic anoxia is a common phenomenon in deep, productive drinking water reservoirs. Anoxia is commonly the result of phytoplankton blooms, which upon death sink into the hypolimnion and are biodegraded by respiring microorganisms. Thermal stratification isolates the hypolimnion from reaeration from the atmosphere and from phytoplankton photosynthesizing in the photic zone. Anoxic conditions in bottom water lead to a number of ecological processes that degrade the water quality in reservoirs. Sediment tends to release ammonia and orthophosphate, a phenomenon known as internal

nutrient loading, which can reinforce eutrophication (Boström et al. 1988, Ahlgren et al. 1994). Sediment can also release metals (e.g., iron and manganese) and other reduced compounds (e.g., sulfides) that degrade the aesthetic quality of source water and complicate water treatment (Sartoris and Boehmke 1987). Elevated concentrations of toxins (e.g., sulfide and ammonia) in the hypolimnion as a result of anoxia may impair aquatic biota within the reservoir and in tail-waters released from the hypolimnion (Beutel et al. 2001, Horne 1989). Anoxic conditions may also increase mercury contamination in lake biota by stimulating

the release of methylmercury from sediment (Slotton et al. 1995, Herrin et al. 1998).

Hypolimnetic aeration systems are commonly installed in reservoirs to avert hypolimnetic anoxia and its harmful consequences while preserving thermal stratification (Cooke et al. 1993). Aeration systems generally utilize a confined air-lift system where compressed air is injected at the bottom of an air-lift tube, which runs from the lake bottom to the surface (McQueen and Lean 1986). Oxygen is transferred to the hypolimnetic water as the air-water mixture travels up the tube. The air-water mixture is de-gassed in a surface chamber then redistributed back into the hypolimnion via a discharge pipe. Aeration systems that use pure oxygen gas, instead of air as an oxygen source, are growing in popularity (Beutel and Horne 1999). Oxygenation systems generally include a liquid oxygen storage tank on shore with evaporator units that transform the liquid oxygen to gas. Oxygen gas is dissolved into lake water through an on-shore contact chamber, a system of diffusers located at the bottom of the lake, or a contact chamber submerged in the lake.

Hypolimnetic oxygen demand (HOD) is an important design parameter for the sizing of hypolimnetic aeration systems. Historically, design engineers have relied on DO field data to estimate HOD (Lorenzen and Fast 1977, Ashley et al. 1987). However, judging by the failure of many aeration systems to substantially increase DO content in the hypolimnion (Smith et al. 1975, Taggart and McQueen 1981, Ashley 1983, Steinberg and Arzet 1984, Soltero et al. 1994), this design method is inadequate.

In this study, hypolimnetic anoxia was evaluated in nine drinking water reservoirs in California, some of which are candidates for the installation of hypolimnetic aeration systems. A number of hypotheses were examined in the context of developing a better method to estimate HOD for the sizing of lakes and reservoir aeration systems. Of primary interest was the effect of mixing at the sediment-water interface on sediment oxygen demand (SOD). Under quiescent conditions, diffusion of oxygen through a relatively thick boundary layer at the sediment-water interface is hypothesized to limit the rate of oxygen uptake by sediment (Jørgensen and Revsbech 1985, Nakamura and Stefan 1994). Mixing may decrease the thickness of the diffusive boundary layer, thereby increasing the flux of DO from overlying water to the sediment. This phenomenon, termed induced oxygen demand, presumably accounts for some of the observed increase in HOD after the startup of hypolimnetic aeration systems (Smith et al. 1975, Ashley 1983, Soltero et al. 1994, Moore et al. 1996, Prepas and Burke 1997, Jung et al. 1998).

This study also examined interrelationships be-

tween trophic status, morphometry and anoxia in the hypolimnion. Firstly, reservoirs with higher trophic status are expected to exhibit higher HOD and SOD since a larger amount of autochthonous organic material sinks into the hypolimnion. Secondly, reservoirs with a larger, deeper hypolimnion should have a higher percentage of their overall HOD exerted in the water column versus the sediment. Sinking organic matter in deeper reservoirs spends a longer duration in the water column, where it can undergo biodegradation prior to settling on the sediment surface. A better understanding of the effects of trophic status and morphometry on hypolimnetic anoxia will provide insight into optimizing the design of hypolimnetic aeration and oxygenation systems.

Methods

Study Sites

Hypolimnetic anoxia was evaluated in a number of reservoirs of differing size and trophic status (Table 1). The majority of reservoirs are located in the coastal region of California's San Francisco Bay Area (Upper Crystal Springs, Lower Crystal Springs, San Antonio, San Andreas, Upper San Leandro, and Lafayette), Central Coast (Lake Bard), or San Diego (San Vicente). Lake Mathews is located 30 miles inland in Southern California's Riverside County. Two of the study reservoirs, Upper Crystal Springs and San Andreas are polymictic. All other reservoirs are monomictic, thermally stratifying from roughly June through October.

Lower and Upper Crystal Springs, San Antonio and San Andreas Reservoirs are operated by the San Francisco Public Utility Commission, which serves high quality, low-nutrient water imported from the Sierra Nevada to the San Francisco Bay Area. Lafayette and Upper San Leandro are operated by the East Bay Municipal Utility District, which serves drinking water to over one million residences in the East San Francisco Bay region. Lafayette Reservoir is filled with nutrient-rich, local runoff and acts only as an emergency water supply. Upper San Leandro Reservoir is filled with a combination of local runoff and low-nutrient snowmelt imported from the Sierra Nevada. The Metropolitan Water District, water wholesaler to more than fifty agencies that serve over 16 million people in Southern California, manages Lake Mathews. The reservoir is the terminus for the Colorado Aqueduct, which transports water from the Colorado River containing low levels of phosphorus but high levels of nitrate to

Table 1.- Characteristics of study sites. NM = not measured. NA = not applicable, sites do not exhibit strong thermal stratification during the summer, thus hypolimnetic oxygen demand cannot be calculated.

Study Site	Volume (10^6 m^3)	Surface Area (km^2)	Max Depth (m)	Mean Depth (m)	Mean Depth of Hypo. (m)	Mean Annual Chl ($\mu\text{g L}^{-1}$)	Mean Annual Secchi Depth (m)	Mass HOD (kg d^{-1})	Areal HOD ($\text{g m}^{-2} \text{d}^{-1}$)	Volume HOD ($\text{mg L}^{-1} \text{d}^{-1}$)	Anoxic Factor (d)	Water Quality Data Source
Lafayette Reservoir, East San Francisco Bay	5.2	0.52	19	10.0	7.5	10.7	3.2	275	1.113	0.147	107	1998-1999 data collected by author. 1968-1967 data from Laverly and Nielsen (1970).
Lake Bard, Ventura County	14	1.0	31	14.0	3.2	NM	2.4	230	0.336	0.1045	31	1999 data from Calleguas Municipal Water District.
Upper Crystal Springs Reservoir, South San Francisco	18	2.2	16	8.2	NA	1.4	3.5	NA	NA	NA	NA	1999 data from San Francisco Public Utility Commission.
San Andreas Reservoir, East San Francisco Bay	22	2.0	19	11.0	NA	1.7	5.4	NA	NA	NA	NA	1999-2000 data from San Francisco Public Utility Commission.
Upper San Leandro Reservoir, Oakland	36	2.4	40	15.0	4.0	5.1	4.9	650	0.402	0.101	63	1997-1999 data collected by author and East Bay Municipal Utility District.
Lower Crystal Springs Reservoir, South San Francisco	50	2.5	37	20.0	9.9	1.9	4.9	400	0.378	0.038	6	1999 data from San Francisco Public Utility Commission.
San Antonio Reservoir, East San Francisco Bay	58	3.2	44	18.1	10.0	3.5	3.0	490	0.353	0.035	14	1999-2000 data from San Francisco Public Utility Commission.
San Vicente Reservoir, San Diego	111	4.3	58	25.8	22.3	NM	3.4	7090	1.83	0.082	147	1999-2000 data from San Diego Water Department. 1992-1999 data from Flow Science (2000).
Lake Mathews, Riverside	220	10	63	22.0	23.3	0.5	6.4	3180	0.792	0.034	18	1997-1999 data from Metropolitan Water District.

Southern California. Lake Bard is operated by the Calleguas Municipal Water District, which distributes potable water to over half-a-million people in southern Ventura County. The reservoir stores water purchased from the Metropolitan Water District to meet emergency and peak demands. San Vicente Reservoir is operated by the San Diego Water Department, which serves over one million inhabitants of Southern California. The reservoir is filled with relatively high-nutrient water imported from the Colorado River and the Sacramento-San Joaquin Delta.

Trophic status parameters reported in Table 1 include mean annual chlorophyll *a* (chl *a*) and mean annual Secchi disk depth. Annual rather than summer averages were used in this analysis due to the relatively long growing season in these California reservoirs. Annual averages were calculated based on field data obtained from the applicable utility or measured by the author using standard methods. Note that no chl *a* data were available for Lake Bard or San Vicente, thus these sites were excluded from some regression analyses. Unfortunately, nutrient data sets were too patchy at a number of sites to allow for adequate statistical evaluation. See Beutel (2000a) for more details on individual sites and sampling methods.

Hypolimnetic Anoxia

A number of parameters quantifying hypolimnetic anoxia were estimated for the study sites. Upper Crystal Springs and San Andreas were excluded from this analysis since they did not exhibit thermal stratification. Mass-based HOD (MHOD, $\text{kg} \cdot \text{d}^{-1}$) was calculated as the maximum slope of DO mass in the hypolimnion versus time (Lorenzen and Fast 1977). Hypolimnetic mass of DO was estimated from DO profiles by calculating the volume-weighted DO concentration, and multiplying this by the total volume of the hypolimnion. MHOD can be converted to an areal HOD (AHOD, $\text{g m}^{-2} \cdot \text{d}^{-1}$) by dividing by the surface area of the hypolimnion, or to a volume-based HOD (VHOD, $\text{mg L}^{-1} \cdot \text{d}^{-1}$) by dividing by the volume of the hypolimnion. Anoxic factor (d^{-1}) was calculated using the method outlined by Nürnberg (1995). The value is equivalent to the number of days that an area of sediment equal to that of the lake surface area is covered by anoxic water.

Sediment Core Collection

Sediment cores were collected from each reservoir between 1995 and 2000. Replication, frequency, and spatial extent of sediment collection varied between study sites (Table 2). Sediment was initially collected with a 15 cm by 15 cm Eckman dredge. Sediment was

soft enough to plug the jaws of the dredge, resulting in the collection of a minimally disturbed sediment surface with overlaying water. The dredge was brought to the surface, and a sediment-water interface sample was sub-sampled into a cylindrical Plexiglas chamber 1.8 liters in volume (Fig. 1). The sub-sample was collected by slowly pushing the chamber top into the sediment and overlaying water captured in the dredge. A cap and gasket were then mated with the bottom of the chamber by hand while the chamber was still in the sediment. The capped chamber was pulled out of the dredge and bolted onto a round Plexiglas base. Sediment-water interface samples consisted of a sediment core 4-8 cm thick with a surface area of 71 cm^2 , and 100-300 ml of overlaying water. After collection of the sediment-water interface sample, chambers were gently filled with bottom water and transported to the laboratory. At Lake Mathews, sediment was too sandy to use the Eckman dredge, so divers collected cores by hand using the chamber top and a modified bottom cap that remained in place after insertion.

Sediment Oxygen Demand Incubations

Once in the laboratory, chambers were incubated in the dark, at in situ temperatures ranging from 12-16°C. Chambers were first allowed to sit for 1-2 days to allow any resuspended matter to settle out of the water column. Chambers were then gently aerated to saturated DO conditions, and capped airtight. DO in chamber water was measured with time using a Yellow Springs Instruments DO meter (model 55) and probe (model 5739). To measure DO, chamber water was mildly stirred with a glass rod then the DO probe was inserted 2-4 cm into the access port in the top of the chamber and gently agitated until a stable DO reading was achieved. The chambers were then topped off with a few milliliters of de-ionized water and capped. The duration of SOD incubations generally ranged from 2-8 days.

In most cases, oxygen demand of chamber water was also measured separately in BOD bottles incubated in the dark, at in situ temperatures ranging from 12-16°C. The rate of oxygen demand in water samples was used to correct total oxygen demand measured in the chamber to obtain oxygen demand exerted only by the sediment. In cases where chamber water was not measured separately (Lafayette, Lake Mathews, Upper San Leandro), oxygen consumption in chamber water was assumed to equal the VHOD measured in the field.

SOD was measured under quiescent, moderately mixed and highly mixed conditions. In the mixed

Table 2.-Summary of experimental chamber incubations. NM = not measured. NA = not applicable. Standard deviations for sample with n greater than three are shown in parentheses.

Study Site	Number of sample events	Number of sample stations	Total number of replicates	Unmixed Chamber Incubations				Moderately Mixed Chamber Incubations				Highly Mixed Chamber Incubations			
				Number of replicates exhibiting 0-order DO uptake	Average 0-order SOD ($\text{g m}^{-2} \text{d}^{-1}$)	Number of replicates exhibiting 1st-order DO uptake	Average 1st-order uptake velocity (m d^{-1})	Number of replicates exhibiting 0-order DO uptake	Average 0-order SOD ($\text{g m}^{-2} \text{d}^{-1}$)	Number of replicates exhibiting 1st-order DO uptake	Average 1st-order uptake velocity (m d^{-1})	Number of replicates exhibiting 0-order DO uptake	Average 0-order SOD ($\text{g m}^{-2} \text{d}^{-1}$)	Number of replicates exhibiting 1st-order DO uptake	Average 1st-order uptake velocity (m d^{-1})
Lafayette	1	1	3	2	0.396	1	0.0851	NM	NM	NM	NM	NM	NM	NM	NM
Lake Bard	1	1	2	0	NA	2	0.0301	2	0.579	0	NA	NM	NM	NM	NM
Upper Crystal Springs	1	1	2	0	NA	2	0.0194	0	NA	2	0.0373	NM	NM	NM	NM
San Andreas	2	2-3	5	0	NA	5	0.0173 (0.0041)	NM	NM	NM	NM	4	0.418 (0.091)	1	0.0493
Upper San Leandro	3	2-4	6-9	0	NA	9	0.0433 (0.0149)	6	0.450 (0.217)	0	NA	NM	NM	NM	NM
Lower Crystal Springs	1	1	2	0	NA	2	0.0364	0	NA	2	0.0805	NM	NM	NM	NM
San Antonio	2	3	6	2	0.140	4	0.0246 (0.0082)	NM	NM	NM	NM	6	0.757 (0.232)	0	NA
San Vicente	1	4	4	0	NA	4	0.0552 (0.0019)	1	0.419	3	0.0800 (0.0144)	0	NA	4	0.1975 (0.0376)
Lake Mathews	2	3	6-12	5	0.0798 (0.0283)	7	0.0300 (0.0139)	5	0.246 (0.320)	3	0.0953 (0.0226)	5	0.505 (0.264)	1	0.0471

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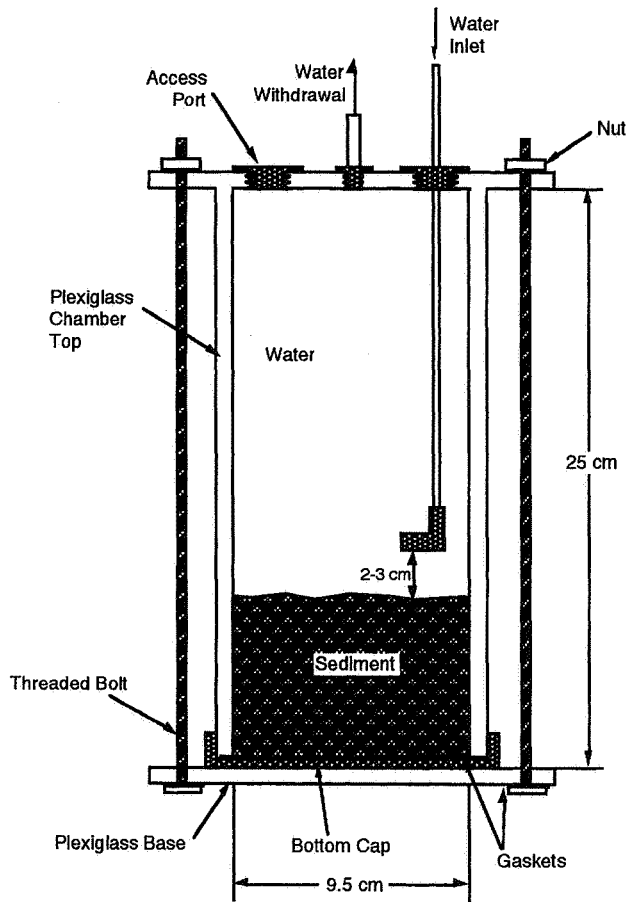


Figure 1.- Schematic of experimental sediment-water chamber.

chambers, water currents at the sediment-water interface were achieved via a recirculating peristaltic pump. The pump drew water from the top of the chamber and discharged it out a small nozzle 2-3 cm above the sediment-water interface. Velocity out of the discharge nozzle was $3\text{--}4\text{ cm}\cdot\text{s}^{-1}$ for moderately mixed conditions and $6\text{--}8\text{ cm}\cdot\text{s}^{-1}$ for highly mixed conditions. Visual inspection of chamber water indicated that velocities were low enough to avoid sediment resuspension. In some experiments mixed and unmixed SOD was determined in replicate chambers incubated in parallel. In other experiments (San Andreas, San Antonio, San Vicente, and Lake Mathews), SOD was first measured under quiescent conditions, then chamber water was re-aerated and SOD was measured under mixed conditions in the same chamber.

Note that the chambers used in this study were initially developed to examine the dynamics of sediment release of nutrients and metals under oxic and anoxic conditions (Beutel 2000a). They are not ideal for quantifying the effects of mixing on SOD since chamber geometry does not allow for an even flow of water

across the sediment-water interface. Future studies should utilize a more rectangular chamber with appropriate inlet and outlet structures so as to promote a uniform flow regime across the sediment, and to facilitate the quantification of water velocity at the sediment-water interface.

Calculation of Sediment Oxygen Demand

DO consumption for each experimental incubation was evaluated to see if SOD was zero-order or first-order with respect to DO concentration in water overlying the sediment (Chapra 1997). For a zero-order relationship, the rate at which DO in overlying water decreases is unrelated to DO concentration, and DO decreases linearly with time. For a first-order relationship, the rate at which DO in overlying water decreases is proportional to the DO concentration, thus DO decreases exponentially with time. To evaluate the fit of a zero-order model, raw data sets of DO concentration versus time were first corrected by subtracting DO uptake in chamber water as determined by the BOD experiments. The r^2 value of a least squares best fit of the corrected DO concentration versus time was then calculated. To evaluate the fit of a first-order model (exponential uptake of DO with time), the r^2 value of a least squares best fit of the natural log of DO concentration versus time was calculated using the raw data set. SOD was classified as linear unless the r^2 value of the exponential model was 0.02 units larger than the r^2 value of the linear model.

SOD for a given data set was calculated differently depending on which model fit best. If the data set fit the zero-order model, SOD ($\text{mg}\cdot\text{L}^{-1}\cdot\text{d}^{-1}$) was calculated as the slope of the best-fit line of the corrected DO concentration versus time. SOD was then converted to a flux ($\text{g}\cdot\text{m}^{-2}\cdot\text{d}^{-1}$) by multiplying by the volume of chamber water and dividing by the sediment surface area. If the data set fit the first-order model, a DO uptake velocity k_1 ($\text{m}\cdot\text{d}^{-1}$) was estimated. The use of this factor allows for the simple estimation of SOD as k_1 multiplied by DO concentration in overlaying water. A rate constant (d^{-1}) was first calculated as the slope of the best-fit line of the natural log of DO concentration with time. SOD ($\text{mg}\cdot\text{L}^{-1}\cdot\text{d}^{-1}$) was then estimated at a DO concentration of 5 and 10 $\text{mg}\cdot\text{L}^{-1}$ by multiplying the rate constant by 5 or 10, then subtracting the rate of DO uptake in the chamber water. The magnitude of k_1 was then estimated as the average of SOD at a DO of 5 $\text{mg}\cdot\text{L}^{-1}$ divided by 5 and SOD at a DO of 10 $\text{mg}\cdot\text{L}^{-1}$ divided by 10.

SOD for all quiescent data sets was also calculated from the slope of the best-fit line of the cor-

rected DO concentration versus time for DO greater than $5 \text{ mg} \cdot \text{L}^{-1}$. This is termed $\text{SOD}_{>5}$. Estimating SOD at high DO concentrations in overlaying water minimizes the impact of first-order uptake kinetics on oxygen uptake, and allows for a simple comparison of the magnitude of SOD from all sites irrespective of whether the oxygen uptake was linear or exponential in nature. Where replication was in triplicate or higher, the standard deviation of average SOD and k_1 values is noted. All SOD and k_1 values were standardized to a temperature of 14°C using the relationship, $\text{SOD}(t_1) = \text{SOD}(t_2) \times 1.065^{(t_1-t_2)}$, where t_1 is 14°C and t_2 is the incubation temperature (Truax et al. 1995).

Results

Hypolimnetic Anoxia

Table 1 includes various parameters related to hypolimnetic anoxia for all study sites, excluding those that did not thermally stratify (Upper Crystal Springs and San Andreas). MHOD ranged from $200\text{--}700 \text{ kg} \cdot \text{d}^{-1}$ for most sites, but values were an order of magnitude higher ($3,000\text{--}7000 \text{ kg} \cdot \text{d}^{-1}$) for larger study sites (San Vicente and Lake Mathews). AHOD ranged from $0.3\text{--}1.8 \text{ g} \cdot \text{m}^{-2} \cdot \text{d}^{-1}$, while VHOD ranged from $0.03\text{--}0.15 \text{ mg} \cdot \text{L}^{-1} \cdot \text{d}^{-1}$. Anoxic factor for the sites ranged from 6-147 d.

Sediment Oxygen Demand

SOD results under unmixed, moderately mixed and highly mixed conditions are summarized in Table 2. Under unmixed conditions, DO uptake was predominantly first-order in nature with 36 of 45 chamber incubations exhibiting exponential DO uptake with time. The magnitude of k_1 ranged from 0.017 to $0.085 \text{ m} \cdot \text{d}^{-1}$. Mixing at the sediment-water interface resulted in an increase in SOD and a shift from first-order to zero-order DO uptake in many chambers. The number of chamber incubations exhibiting first-order DO uptake under mixed conditions dropped to 16 of 45. In zero-order chambers, SOD under moderately mixed conditions ranged from $0.25\text{--}0.58 \text{ g} \cdot \text{m}^{-2} \cdot \text{d}^{-1}$, while k_1 values in first-order chambers were $0.037\text{--}0.095 \text{ m} \cdot \text{d}^{-1}$. Under highly mixed conditions, SOD in zero-order chambers ranged from $0.42\text{--}0.76 \text{ g} \cdot \text{m}^{-2} \cdot \text{d}^{-1}$, and k_1 values in first-order chambers were $0.047\text{--}0.198 \text{ m} \cdot \text{d}^{-1}$.

Fig. 2 shows two differing responses of SOD to moderate mixing in experimental chambers. In the

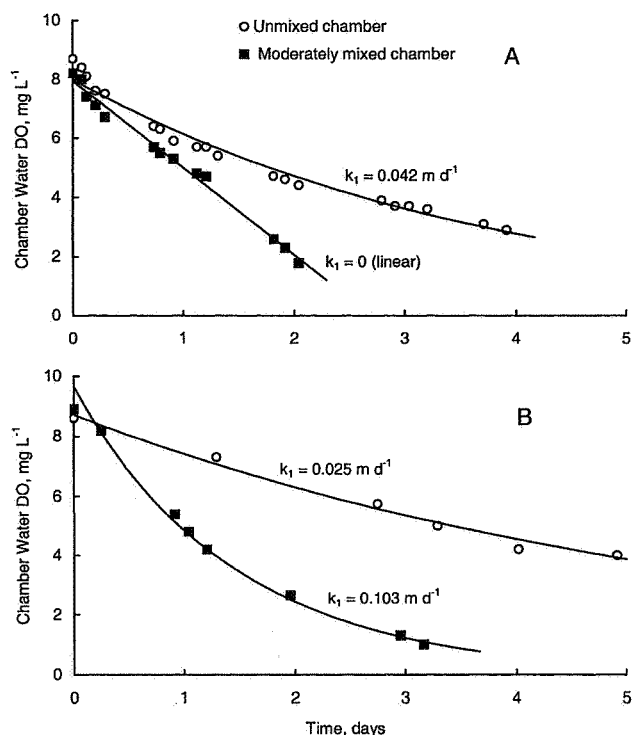


Figure 2.—SOD under unmixed and mixed conditions in sediment from Upper San Leandro Reservoir (A) and Lake Mathews (B). Curves show best-fit line or exponential curve with the value of the first-order DO uptake velocity (k_1). Note that increased velocity at the sediment-water interface in the mixed chambers increased SOD. In the Upper San Leandro chamber, DO uptake shifted from first to zero-order with respect to dissolved oxygen in overlaying water.

more common response, mixing caused both an increase in SOD and a shift from first-order to zero-order DO uptake (Fig. 2A). In some experimental chambers, mixing increased SOD, but first-order uptake of DO was still apparent (Fig. 2B). Fig. 3 summarizes $\text{SOD}_{>5}$ for all study sites. Under quiescent conditions, $\text{SOD}_{>5}$ ranged from $0.1\text{--}0.8 \text{ g} \cdot \text{m}^{-2} \cdot \text{d}^{-1}$. Under moderate mixing, $\text{SOD}_{>5}$ ranged from $0.2\text{--}0.7 \text{ g} \cdot \text{m}^{-2} \cdot \text{d}^{-1}$, and on average increased by a factor of 2.0 ± 0.57 (average \pm standard deviation, $n = 6$) compared to unmixed conditions. $\text{SOD}_{>5}$ under high mixing ranged from $0.5\text{--}0.9 \text{ g} \cdot \text{m}^{-2} \cdot \text{d}^{-1}$, and on average increased by a factor of 3.9 ± 0.97 ($n = 4$).

The magnitude of k_1 values under different mixing regimes can be used to evaluate the combined effects of DO concentration and mixing on SOD. Fig. 4 shows estimated SOD in sediments from San Vicente Reservoir as a function of DO concentration and level of mixing. Based on experimental chamber results, SOD exhibited first-order characteristics under all mixing regimes, and an average k_1 value was estimated for quiescent ($0.055 \pm 0.002 \text{ m} \cdot \text{d}^{-1}$), moderately mixed ($0.080 \pm 0.014 \text{ m} \cdot \text{d}^{-1}$), and highly mixed conditions

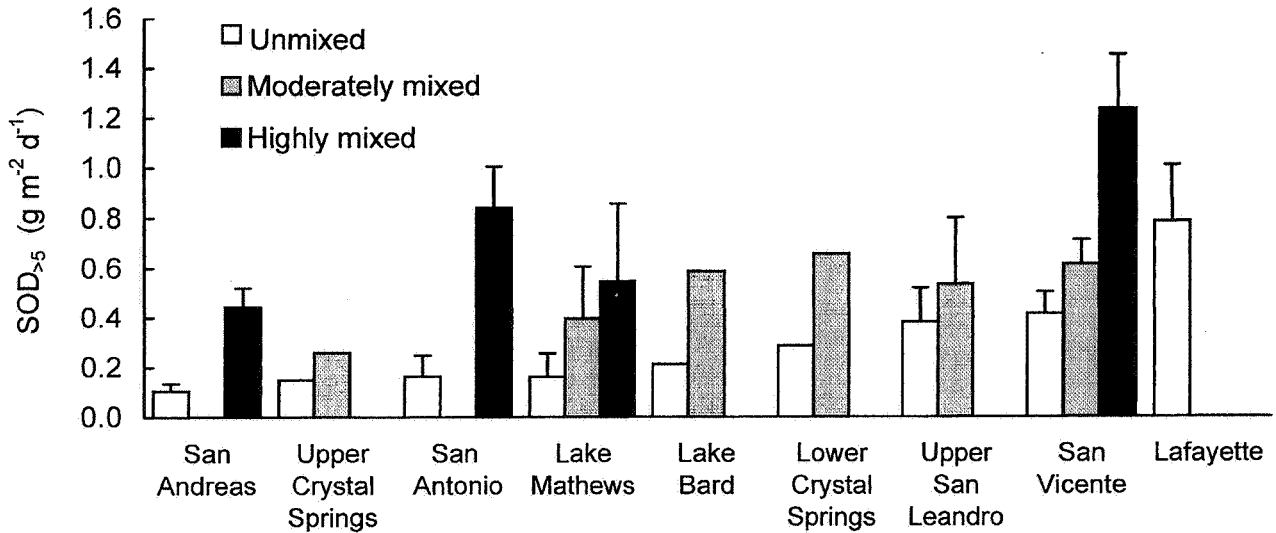


Figure 3.—Summary of SOD_{>5} results. Error bars show one standard deviation. SOD generally rose as mixing at the sediment-water interface increased. In addition, SOD increased with trophic status. Reservoirs are shown in approximate order of increasing trophic status from left to right.

(0.198 ± 0.038 m · d⁻¹). SOD was then calculated as a linear function of k₁ multiplied by DO concentration for the three different mixing conditions. Fig. 4 predicts roughly a ten-fold increase in SOD when comparing quiescent, low oxygen conditions (2-4 mg · L⁻¹ DO) to highly mixed, saturated conditions (8-10 mg · L⁻¹ DO).

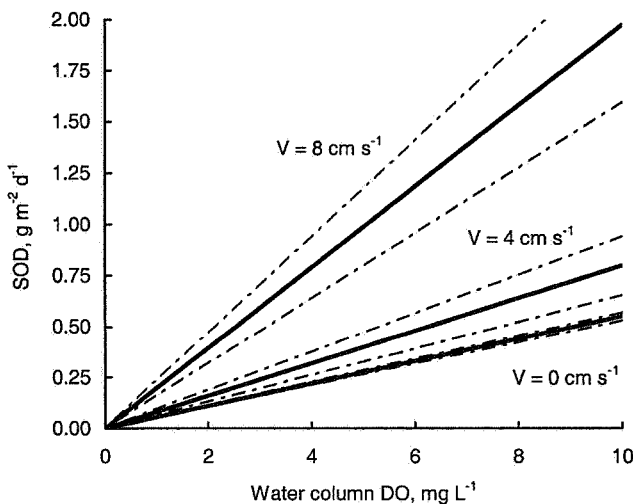


Figure 4.—SOD in San Vicente Reservoir as a function of water column DO and velocity (V) near the sediment-water interface developed from experimental chamber incubations. Dotted lines show plus/minus one standard deviation of estimated SOD. Note that SOD increases both as a function of increased mixing at the sediment-water interface and increased DO concentration in water overlaying sediment.

Correlation Between Anoxia, Trophic Status and Morphometry

Relationships between hypolimnetic anoxia, SOD, trophic status, and morphometry were evaluated using a Pearson's correlation matrix for applicable variables (Table 3). SOD_{>5}, VHOD, and anoxic factor all correlated significantly with chl a (Fig. 5). Secchi disk depth was excluded from Table 3 since it did not correlated with any of the parameters quantifying hypolimnetic anoxia.

The ratio of SOD_{>5} to AHOD, an indicator of the relative importance of oxygen demand in the water column versus the sediment, showed a significant negative correlation with morphometry including volume, surface area, and mean depth of the hypolimnion (Fig. 6). This relationship allows for a simple method to estimate SOD as fraction of AHOD (g m⁻² · d⁻¹) using mean depth of the hypolimnion (Z_{mean-hypo}, m):

$$SOD_{>5} = AHOD (-0.030 Z_{mean-hypo} + 0.905) (r^2 = 0.78) \quad (1)$$

Since AHOD can be easily estimated from field DO profiles, Equation 1 allows for the estimation of SOD without the need for intensive experimental work.

Fig. 7 shows AHOD and VHOD as a function of depth and trophic status. VHOD for sites of similar trophic status appears to drop with increasing mean

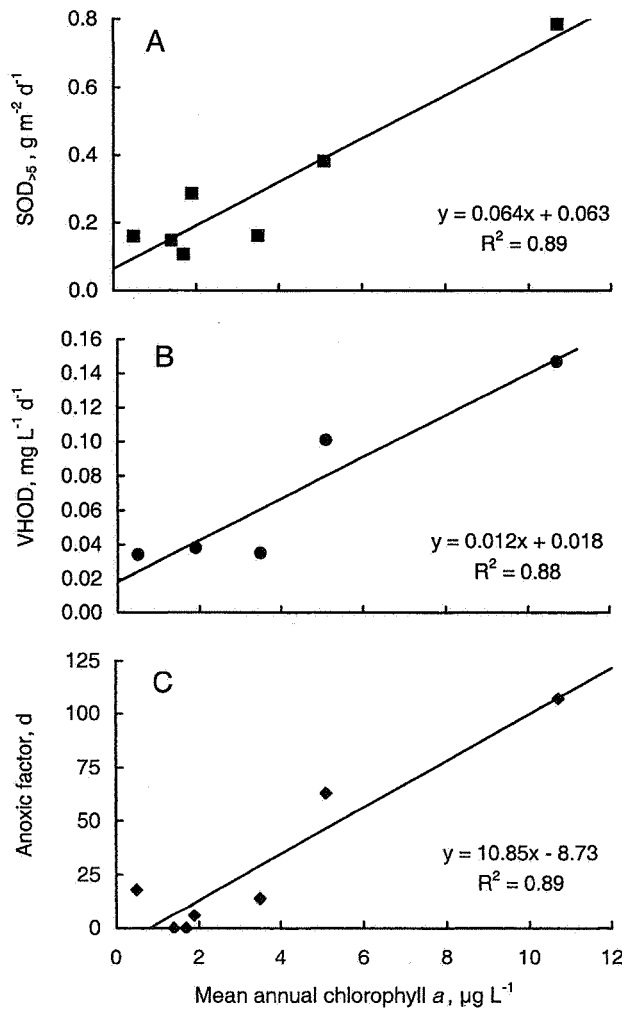


Figure 5.—SOD (A), VHOD (B) and anoxic factor (C) as a function of chl a. Best-fit linear regression lines and equations are shown. These significant relationships support the premise that hypolimnetic anoxia is related to trophic status.

depth. In contrast, AHOD increases with depth for sites of similar trophic status. Multiple linear regression analysis was performed to evaluate empirical relationships between parameters quantifying hypolimnetic anoxia, and those quantifying morphometry and trophic state. AHOD ($\text{g m}^{-2} \cdot \text{d}^{-1}$) or MHOD ($\text{kg} \cdot \text{d}^{-1}$) can be estimated as a function of mean depth of the hypolimnion ($Z_{\text{mean-hypo}}$, m) or the volume of the hypolimnion (V_{hypo} , 10^6 m^3), and chl a ($\mu\text{g} \cdot \text{L}^{-1}$), as follows:

$$\text{AHOD} = 0.0451 Z_{\text{mean-hypo}} + 0.102 \text{ chl } a - 0.326 \quad (r^2 = 0.95) \quad (2)$$

$$\text{MHOD} = 32.8 V_{\text{hypo}} + 17.6 \text{ chl } a + 94.0 \quad (r^2 = 0.98) \quad (3)$$

The above equations can be used to estimate the effects of changes in hypolimnetic size or trophic status on oxygen demand in the hypolimnion.

Discussion

This comprehensive study of nine drinking water reservoirs of varying size and trophic status expands the published SOD data on reservoirs in the Western United States. SOD values measured in this study, roughly $0.1\text{--}0.8 \text{ g m}^{-2} \cdot \text{d}^{-1}$ under quiescent conditions, are comparable to rates reported by many researchers. SOD values reported for oligotrophic and mesotrophic lakes generally range from $0.05\text{--}0.3 \text{ g m}^{-2} \cdot \text{d}^{-1}$ (Hutchinson 1957, Bowman and Delfino 1980, Mathias and Barica 1980, Adams et al. 1982), while rates reported for eutrophic lakes range from $0.2\text{--}1.2 \text{ g m}^{-2} \cdot \text{d}^{-1}$ (Hutchinson 1957, Edberg and Hofsten 1973, Brewer et al. 1977,

Table 3.—Correlation coefficients for pertinent study parameters. * = $p < 0.05$. ** = $p < 0.01$, NS = not significant, $n = 6$ to 8 . Significant positive correlations were found between trophic status (chl a) and a number of parameters quantifying hypolimnetic mean depth.

	Hypo. Volume	Hypo. Surface Area	Hypo. Mean Depth	Mean Annual Chl a	Mass HOD	Areal HOD	Volume HOD	Anoxic Factor
Hypo. Surface Area	0.97**							
Hypo. Mean Depth	0.97**	0.91**						
Mean Annual Chl a	NS	NS	NS					
Mass HOD	0.87*	0.86*	0.83*	NS				
Areal HOD	NS	NS	NS	NS	0.84*			
Volume HOD	NS	NS	NS	0.94*	NS	NS		
Anoxic Factor	NS	NS	NS	0.94**	NS	0.88*	NS	
SOD>5	NS	NS	NS	0.95**	NS	NS	0.81*	0.77*
SOD>5:AHOD	-0.87*	-0.78*	-0.88**	NS	NS	NS	NS	NS

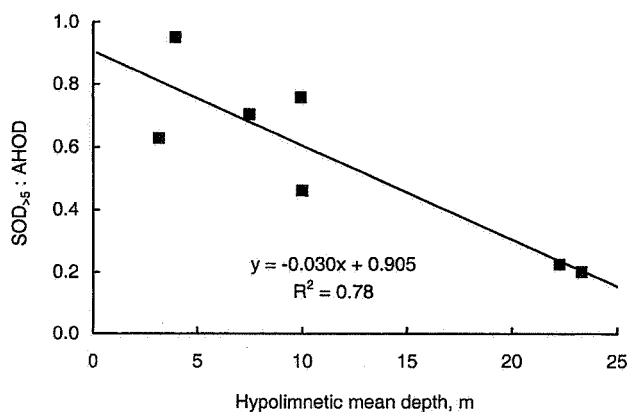


Figure 6.—Ratio of SOD to AHOD as a function of mean depth of the hypolimnion. Best-fit linear regression line and equation is shown. Results suggest that oxygen demand in deeper reservoirs is exerted more in the water column versus at the sediment-water interface.

Mathias and Barica 1980, Belanger 1981). Martin et al. (1985) report median SOD values for lakes and reservoirs ranging from 0.2-0.4 $\text{g m}^{-2} \cdot \text{d}^{-1}$. Higher rates of SOD, ranging from 1-4 $\text{g m}^{-2} \cdot \text{d}^{-1}$, have been reported in shallow, warm, eutrophic lakes (Belanger 1981, Veenstra and Nolen 1991, Beutel 2000b).

Effects of Mixing at the Sediment-Water Interface on SOD

Increasing water currents at the sediment-water interface resulted in a two to four-fold increase in SOD and, in many chambers, a shift from diffusional limitation (first-order DO uptake) to substrate limitation (zero-order DO uptake). Belanger (1981) reported a similar increase in SOD in sediments from a shallow, eutrophic lake when the flow rate through experimental apparatus was increased. SOD in sediments from a eutrophic lake in Washington State exhibited a three-fold increase when overlying water was mixed at 2-3 $\text{cm} \cdot \text{s}^{-1}$, and a four-fold increase at currents ranging from 8-10 $\text{cm} \cdot \text{s}^{-1}$ (Moore et al. 1996). In littoral sediments from two eutrophic lakes, Mackenthun et al. (1998) reported that increases in velocity from 0 to 2-3 $\text{cm} \cdot \text{s}^{-1}$ near the sediment surface resulted in a two to three-fold increase in SOD. Both Jørgensen and Revsbech (1985) and Josiam and Stefan (1999) measured an easing of diffusional limitation on SOD when turbulence was increased at the sediment-water interface of aquatic sediments.

Results of this study support the diffusive boundary layer model for SOD, in which a motionless sublayer of water at the sediment-water interface acts as a barrier to the diffusion of oxygen from overlying water into sediment (Jørgensen and Revsbech 1985, Nakamura et

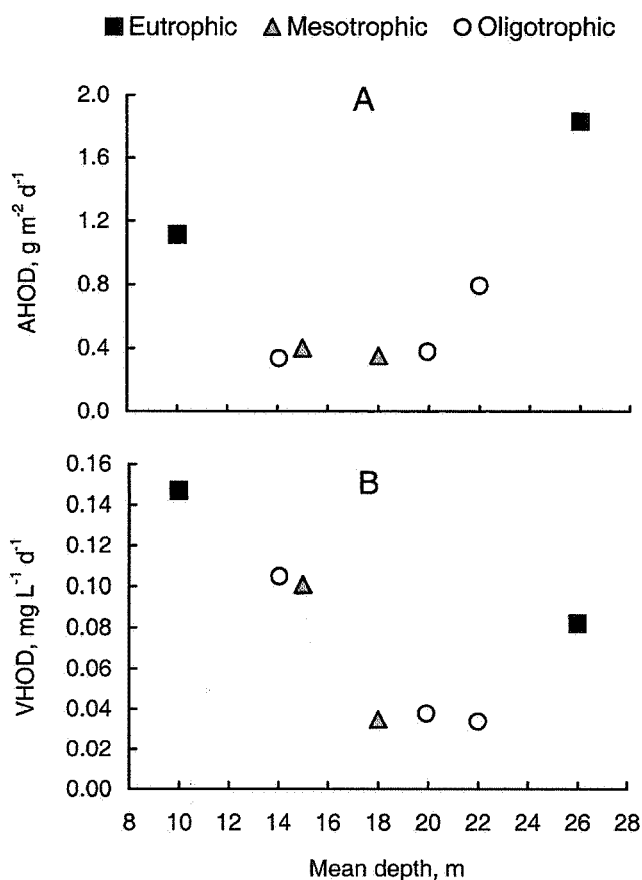


Figure 7.—AHOD (A) and VHOD (B) versus mean depth. VHOD for sites of similar trophic status appears to drop with increasing mean depth. In contrast, AHOD appears to increase with depth for sites of similar trophic status.

al. 1994). With increasing flow velocity near the sediment water-interface, the sublayer is depressed or disrupted resulting in a shift away from diffusional limitation on SOD and an increase in the magnitude of SOD. This phenomenon has implications for the proper sizing of hypolimnetic aeration systems since increased mixing at the sediment as a result of system operation may increase SOD.

Effects of Water Column DO Concentration on SOD

Under unmixed conditions, DO uptake by most sediments was first-order with respect to DO concentration in chamber water. Thus, DO concentration is a major controlling factor in the rate of sediment oxygen uptake. These results also support the diffusive boundary layer model for SOD, which predicts that SOD is linearly related to DO concentration (Nakamura et al. 1994), with DO concentration acting as the diffusional driving force for the flux of DO from water into

the sediment. The dependency of SOD on DO concentration may be an additional reason for the historical undersizing of aeration systems. As DO decreases in the hypolimnion with time, SOD decreases as well. Oxygen demand in the hypolimnion calculated from a series of seasonal DO profiles will underestimate potential oxygen demand since SOD is decreasing with time.

A number of studies have observed decreases in the rate of oxygen consumption in the hypolimnion or at the sediment-water interface as a function of decreasing DO concentration in the water column. Reports of decreasing HOD once water column DO drops below $3\text{--}4\text{ mg}\cdot\text{L}^{-1}$ are common (Hargrave 1972, Welch 1974, Mathias and Barica 1980, Ellis and Stefan 1989). Hargrave (1969) reported a linear decrease in SOD with decreasing DO in overlaying water in Marion Lake, Canada. Moore et al. (1996) made a similar observation in experimental chambers containing sediment from a eutrophic lake.

Correlation between Trophic Status, Morphometry and Anoxia

The observed correlation between parameters quantifying hypolimnetic anoxia and chl *a* in this study support the supposition that increased trophic status is linked to hypolimnetic anoxia. Unlike chl *a*, Secchi disk depth was not a useful parameter for defining the trophic status of study sites since Secchi disk depth did not follow its conventional exponential relationship with chl *a* (Carlson 1977). At some sites blue-green algae dominated, and due to their colonial morphology, transparency did not change substantially with increased biomass (Horne and Goldman 1994). Transparency at other sites appeared to be influenced by inorganic suspended solids. Researchers have observed a similar correlation between trophic state and hypolimnetic anoxia. Welch et al. (1976) reported that winter oxygen depletion rates in Ontario lakes were correlated with chl *a* and total phosphorus, and Mathias and Barica (1980) reported that average SOD in eutrophic lakes ($0.80\text{ g m}^{-2}\cdot\text{d}^{-1}$) was three times that of oligotrophic lakes ($0.23\text{ g m}^{-2}\cdot\text{d}^{-1}$). Veenstra and Nolen (1991) report a correlation between SOD and mean annual chl *a* in a study of five Southwestern lakes.

Relationships (e.g., Equations 2 and 3) were also observed that relate hypolimnetic anoxia to trophic status and morphometry. The relationships suggest that increasing either productivity or hypolimnetic size results in a larger oxygen demand exerted in the hypolimnion. The equations developed in this study can be used to evaluate the relative impacts of eutrophication on anoxia. For example, based on

equation (2), an increase in mean annual chl *a* from $4\text{ to }8\text{ }\mu\text{g}\cdot\text{L}^{-1}$ in a reservoir with a hypolimnetic mean depth of 10 m would result in roughly a 40% increase in AHOD ($0.53\text{ to }0.94\text{ g m}^{-2}\cdot\text{d}^{-1}$).

Depending on the method with which hypolimnetic anoxia was quantified, the correlation between anoxia and morphometry can be either negative or positive (Fig. 7). For example, VHOD for sites of similar trophic status appears to drop with increasing mean depth. This is a result of oxygen demand being diluted into a larger hypolimnetic volume. Welch et al. (1976) and Mathias and Barica (1980) also reported that oxygen depletion rates in Canadian lakes were negatively correlated with mean depth. In contrast, AHOD appears to increase with depth for sites of similar trophic status. While Hutchinson (1957) suggested that AHOD is independent of depth, Charlton (1980) noted a positive linear correlation between AHOD and maximum depth for mesotrophic and eutrophic lakes using data reported by Hutchinson (1957). In an evaluation of 65 North American lakes, Nürnberg (1995) found that average phosphorus and a lake shape factor, mean depth divided by the square root of lake surface area, accounted for that nearly two-thirds of the variance in anoxic factor.

Two observations from this study, the apparent increase in AHOD for lakes of a similar trophic status with increasing mean depth, and the negative correlation between SOD:AHOD and mean depth of the hypolimnion, support a conceptual model of oxygen consumption in the hypolimnion developed by Charlton (1980). In shallow lakes, organic matter settles to the sediments relatively quickly. As a result, there is little time for biodegradation in the water column, and the majority of oxygen demand is exerted in the sediment resulting in a high SOD:AHOD. In contrast, the longer duration of settling in deeper lakes allows for more complete degradation of organic matter in the water column and a low SOD:HOD. In addition, in shallower systems there is greater potential for the burial of organic matter before it exerts its full oxygen demand. Thus, for lakes of comparable trophic status, shallower systems have lower AHOD due to burial of non-biodegraded organic matter.

Implications for Aeration System Design

Based on this study, a design SOD for reservoirs can be approximated by multiplying SOD, either measured experimentally or based on DO field data and morphometry using Equation 1, by an inducement factor based on predicted velocities at the sediment-water interface. At bottom water velocities ranging

from 3-4 cm s⁻¹ the inducement factor is approximately two, while at velocities ranging from 6-8 cm s⁻¹ the inducement factor is approximately four. The design SOD value would then be added to the remaining oxygen demand exerted in the hypolimnetic water column.

Moore et al. (1996) used such a method to determine induced SOD in Lake Newman, Washington. Velocity at the sediment-water interface as a function of distance away from a horizontal diffuser was estimated using standard hydrodynamic equations. Sediment was split into sections located progressively away from the diffuser line. Design SOD was then calculated as the summation of unmixed SOD multiplied by a velocity-based inducement factor for each sediment section multiplied by sediment area. As discussed above, SOD can also be a function of DO concentration in overlaying water, and this relationship may need to be incorporated into determining the design SOD.

Designers of aeration systems should expect induced SOD to be a more crucial issue in shallow reservoirs. Correlations developed in this study between the ratio of SOD to AHOD and morphometry suggests that reservoirs of high productivity and moderate depth (mean depth of 10-15 m) are particularly vulnerable to induced oxygen demand. These systems have high SOD, and a large fraction of their total oxygen demand is exerted in the sediment. As a result, mixing in the hypolimnion could increase overall oxygen demand in the hypolimnion substantially. Based on this study, SOD in shallow reservoirs can account for 60-95% of total AHOD (see Fig. 6). Assuming SOD is 75% of AHOD, and assuming a 4-fold increase in SOD as a result of mixing and increased DO levels, overall AHOD would increase by a factor of 3.3. In contrast, a deeper reservoir with a SOD:AHOD ratio of 15% would only have a 1.5-fold increase in AHOD.

In order to reduce capital and operation costs, aeration and oxygenation system designers may be tempted to minimize induced SOD by designing systems that do not cause substantial mixing at the sediment-water interface, and that maintain minimal DO concentration in the hypolimnion. However, systems should be designed to enhance SOD so as to maximize the depth of penetration of DO into sediment. Only with adequate DO penetration, will aeration or oxygenation promote aerobic biogeochemical reactions in surface sediments that will decrease the release of problem-causing compounds from sediment.

The effects of inducing SOD on the depth of DO penetration can be estimated. Assuming that SOD is limited by diffusion, that oxygen consumption in sediment is zero-order with respect to DO in the pore water, and that diffusivity in the sediment is constant with depth, SOD can be estimated as (Berner 1980):

$$\text{SOD} = (2\text{CRD})^{1/2} \quad (4)$$

Where C (mg L⁻¹) is the oxygen concentration at the sediment-water interface, R (mg L⁻¹ · d⁻¹) is the consumption rate of oxygen in the sediment, and D (m² · d⁻¹) is the diffusivity of oxygen in sediment. The depth of oxygen penetration into the sediment, X (m), can also be estimated as:

$$X = (2\text{CD}/R)^{1/2} \quad (5)$$

Combining Equations 4 and 5 yields the simple equation:

$$X = \text{SOD}/R \quad (6)$$

Equation 6 shows that the depth of penetration of oxygen into sediment is proportional to SOD, and inducing SOD via mixing at the sediment-water interface and/or the maintenance of high levels of DO in water overlaying sediment will enhance the penetration of oxygen into the sediment.

A thicker layer of oxidized sediment resulting from induced oxygen demand should act as a better barrier against the release of reductant-soluble, problem-causing compounds from sediment. For example, Rysgaard et al. (1994) found that increasing levels of DO in water above lake sediment increased DO penetration and enhanced coupled nitrification-denitrification, thereby enhancing the loss of nitrogen from the aquatic ecosystem. A lack of oxygen penetration into sediment may account for the continued release of phosphate from sediment in Swiss lakes that have been artificially aerated since 1982 using bubble plume oxygenation systems (Imboden 1985, Gächter and Wehrli 1998). In Lake Baldegg, Switzerland, oxygen penetration into sediments appears to be marginal though hypolimnetic water remains oxygenated (DO > 3-4 mg L⁻¹). The presence of *Beggiatoa* sp., a microaerophilic, sulfide-oxidizing bacteria, on the sediment surface indicates that the sediment-water interface is only slightly oxygenated (Jørgensen & Revsbech 1983). In addition, the lack of spawning success of whitefish (*Coregonus* sp.) at Lake Baldegg has been attributed to limited oxygen penetration into the sediment (Gächter and Wehrli 1998). The fish deposit eggs on profundal sediments, and without adequate oxygen in surface sediments the eggs do not survive. In Lake Hallwil, Switzerland, water near much of the sediment-water interface is sometimes low in oxygen (<0.5 mg/L) even when the bubble plume oxygenation system is in operation (Andreas Lorke and Dan McGinnis, Swiss Federal Institute for Environmental Science and Technology, unpublished), and oxygen penetration into surface sediment is negligible.

The most appropriate method to satisfy high oxygen demand in profundal sediments, and ensure adequate oxygen penetration into the sediment and

resulting improvements in water quality, is the use of hypolimnetic aeration with pure oxygen. Unlike systems that rely on air, pure oxygen systems have the capability to overcome increases in sediment oxygen demand resulting from increased mixing and DO concentrations in the hypolimnion after system start-up. Three types of hypolimnetic oxygenation systems have been developed include bubble plume systems, diffuser systems, and submerged contact chambers (Beutel and Horne 1999). Unlike bubble plume systems that generate a vertical current as oxygen bubbles rise through the water column, and diffuser systems that dissolve oxygen into the water column at the bubble rises away from the sediment, oxygenated water from a submerged contact chamber can be discharged horizontally across the sediments (Speece 1994). With horizontal discharge, the highly oxygenated water is better able to satisfy SOD and maintain well-oxygenated conditions at the sediment-water interface. Such systems should especially be utilized in shallow reservoirs where much of the HOD is exerted in the sediment.

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