Criteria Assessment Framework

Chapter 4. Dissolved Oxygen Criteria Attainment: Implications and Options for Restructuring the Open Water Designated Use.

BACKGROUND

The Issues

The Chesapeake Bay science and management community has expressed concerns with the Chesapeake Bay dissolved oxygen criteria attainment assessment of the open water designated use (MRAT 2009, CBP-STAC 2012). The method for assessing attainment in the open water designated use combines offshore waters and shallow, nearshore waters into a single volume-based evaluation unless a state has specifically delineated a sub-segment within a Chesapeake Bay management segment (U.S. EPA 2007a). According to U.S. EPA (2007a), "Neither the need nor the requirement exists for a separate assessment of dissolved oxygen criteria attainment strictly within shallow waters (0-2 meters in depth)". However, U.S EPA (2007a) goes on to state that conditions in these shallow water areas are considered to vary greatly from the open water of the mid-channels habitats. Given significant differences in local drivers of dissolved oxygen dynamics for the two habitats, and with substantial new data and analyses available, scientists and managers have continued to raise concern that dissolved oxygen patterns in the two habitats may be characteristically different warranting sub-segment separation and therefore separate dissolved oxygen criteria attainment assessments.

A supplemental concern was expressed that the sheer volume of offshore water regions may overwhelm signals of distress in shallow waters for dissolved oxygen criteria attainment assessments. In general, shallow water habitat is referenced to be a water depth of $\leq$2m (e.g. p38, U.S. EPA 2007a). The 2m depth contour was selected as the maximum depth for the lower vertical boundary of the shallow water designated use (U.S. EPA 2003b). Approximating the area and volume of all the shallow water habitat, meaning habitat inclusive of all waters less than 2m within Chesapeake Bay and its tributaries, Bay facts suggest there are at least 700,000 acres (2832.9 km^2) $\leq$ 6 feet deep (http://www.chesapeakebay.net/discover/bay101/facts). The surface area of the tidal waters of the Bay and its tributaries is estimated to be 11,601 km^2. Therefore, the shallow water habitat of the Bay is approximately 24.4% of its surface area. If we assume an average shallow water depth to be half the maximum depth of those acres, i.e. 3 feet, then the volume of shallow water habitat is estimated to 4.56% of the total Bay volume or 2.59 km^3. For comparison, 2.59km^3 is approximately the same size as the peak volume observed for deep water anoxia in Chesapeake Bay between 1985 and 2006. Considering a worst case scenario for a day where all shallow water experiences anoxia, a total Bay anoxic volume including shallow water could be 2x or 100% more than what is reported in the deep channel waters of the mainstem Bay alone. Segment specific shallow water volumes can be as much as X% of a segment and range from X to Y.

The relationship of dissolved oxygen behavior in nearshore, shallow water habitat and offshore open water has been poorly understood. Resolving the combination of science and management concerns about this relationship for the two habitats requires a basic characterization of shallow water dissolved oxygen dynamics. Comparisons between shallow water and offshore dissolved oxygen behavior in the two habitats are especially needed to support decisions on maintaining the present open water designated use dissolved oxygen criteria attainment assessment framework or provide options and implications to delineating the two habitats into sub-segments. Separate sub-segments could affect monitoring designs and programming, criteria assessment frameworks and protocols, and management considerations for the Chesapeake Bay.

The following sections of this chapter discuss 1) Chesapeake Bay open water and shallow water designated use definitions supporting the Chesapeake Bay Program partners Clean Water Act water quality standards attainment assessments, 2) recent water quality monitoring history for tidal waters of the Chesapeake Bay mainstem and tributaries relevant to understanding shallow water dissolved oxygen dynamics and high frequency patterns in dissolved oxygen behavior, 3) reviews key findings from the most recent Chesapeake Bay Program partner analyses regarding high frequency water quality dynamics and shallow water dissolved oxygen behavior conducted by the Umbrella Criteria Assessment Team (CBP STAC 2012), 4) reviews the issue of dissolved oxygen behavior comparisons between habitats contained within the open water designated use and 5) presents implications and options regarding considerations for partitioning shallow water habitat from offshore waters for monitoring, assessment and management in Chesapeake Bay **(This is the expected workshop product in April)**.

The Relationship of the Open Water and Shallow Water Designated Use Definitions.

The Chesapeake Bay ambient water quality criteria attainment assessments are habitat and season specific. Designated use definitions have been published (U.S. EPA 2003a, 2003b, 2004a) and adopted in Chesapeake Bay watershed partner water quality standards for five habitats:

- Migratory and spawning
- Open water
- Shallow water Bay grass
- Deep water
- Deep channel.

The Chesapeake Bay dissolved oxygen criteria for the open water fish and shellfish designated use were developed to fully protect the survival, growth and propagation of balanced, indigenous populations of ecologically, recreationally and commercially important fish and shellfish inhabiting open water habitats (U.S. EPA 2003a). These open water criteria were based on established dissolved oxygen concentrations to protect against losses in larval recruitment, growth effects on larvae and juveniles and the survival of juveniles and adults in tidal fresh to high salinity habitats (U.S. EPA 2003a).

The U.S. EPA designated use boundary definition for open water adopted in the Chesapeake Bay Program partner water quality standards is:

From June 1 through September 30 the open water designated use included tidally influenced waters extending horizontally from the shoreline to the adjacent shoreline. If a pycnocline is present and, in combination with bottom bathymetry and water-column circulation patterns, presents a barrier to oxygen replenishment of deeper waters, the open water fish and shellfish designated use extends down into the water column only as far as the measured upper boundary of the pycnocline. If a pycnocline is present both other physical circulation patterns (such as influx of rich oceanic bottom waters, the open-water fish and shellfish designated use extends down into the water column to the bottom water-sediment interface.

From October 1 through May 31, the open-water designated us includes all tidally influenced waters extending horizontally from the shoreline to the adjacent shoreline, extending down through the water column to the bottom water-sediment interface. (U.S. EPA 2003b).

The shoreline to shoreline definition of open water is based on the assumption that "the dissolved oxygen requirements for the species and communities inhabiting open- and shallow-water habitats are similar enough to ensure protection of both the open-water and shallow-water designated use with a single set of criteria". Shallow water habitats are, therefore, a subset of the open water designated use in Chesapeake Bay. Unless a state has specifically delineated a sub-segment within a Chesapeake Bay management segment, neither the need nor the requirement exists for a separate assessment of dissolved oxygen criteria attainment strictly within shallow waters (0-2 meters in depth) according to U.S. EPA (2007a)

In the practice of water quality standards attainment assessments, the shallow water designated use is a further subset of Chesapeake Bay shallow water habitat. The shallow-water bay grass designated use protects underwater bay grasses and the many fish and crab species that depend on the vegetated shallow-water habitat provided by underwater grass beds. The shallow water designated use boundary in a management segment is classified by light limitation application depths for underwater grasses that range from no-grow zones (0m) to 0.5m, 1m or 2m depths in the Bay. Applying the living resource-based assumptions for dissolved oxygen needs of shallow water communities with the Chesapeake Bay criteria framework resulted in the Open Water fish and shellfish designated use criteria also applying to the shallow water Bay grass habitat throughout the year. (Table X needed here? Criteria table). However, the science supporting light limitation conditions for submerged aquatic vegetation application depths has historically been stronger than the understanding of shallow water dissolved oxygen dynamics for Chesapeake Bay.

Evolution of shallow water monitoring and science in Chesapeake Bay

Habitat needs for species and communities in offshore and nearshore habitats were considered similar enough to combine for Chesapeake Bay dissolved oxygen attainment assessments (U.S. EPA 2003a,b 2004a). Historically, however, shallow-water water quality data and the ability to make paired-comparisons were rare for Chesapeake Bay. Using a wide range of data from state programs, academic institutions and citizen monitoring, comparisons of mid-channel and nearshore water quality in Chesapeake Bay were synthesized (Karrh 1999, Batiuk et al. 2000, U.S. EPA 2007a) focusing on SAV-relevant parameters (K_d, dissolved inorganic nitrogen, dissolved inorganic phosphorus, total suspended solids and chlorophyll *a*). Results showed comparability of mid-channel and nearshore water quality conditions was very site specific.

Shallow water habitats with respect to dissolved oxygen dynamics further remained poorly characterized across Chesapeake Bay and its tidal tributaries (Batiuk et al. 2009). Part of this information gap is a function of how the Chesapeake Bay long-term water quality monitoring program sampling framework was designed. The long term water quality monitoring network is based on the established stations in mid-channel positions where vertical profiles of water quality conditions are assessed and used in evaluating status, water quality standards and tracking trends.

In the late 1990s and early 2000s shallow water sampling efforts intensified by a variety of agencies and institutions with an emphasis on understanding nearshore water quality conditions in Chesapeake Bay. These focused efforts arose in response to a wide range of factors coalescing in time including 1) improvement of in-situ water quality sensor packages for long-term deployments, 2) affordability of the in-situ technology, 3) 1999-2002 pilot studies demonstrating sampling of water quality using in-situ samplers in fixed station, continuous monitoring mode and in DATAFLOW or underway sampling mode mapping conditions of estuarine surface waters (U.S. EPA 2007a), 4) computer data storage abilities to handle high density data streams from in-situ data loggers, 5) fish kill distributions in Chesapeake Bay (Maryland Department of the Environment, Chris Luckett, pers. Comm.), 6) increasing science that linked harmful algal bloom plankton species (HABs) found in estuaries with the potential to produce toxins coincidently capable of negatively affecting fish and human health (Burkholder and Glasgow 1997, Grattan et al. 1998, Stow 1999), 7) identification of such toxigenic HAB species of concern in Chesapeake Bay (Marshall 1996, Deeds et al. 2002, Marshall 2003), 8) toxigenic HAB species found at Chesapeake Bay fish kill sites in shallow water environments (Tango et al. 2006), 9) human health incidences from water contact at locations with *Pfiesteria, Pfiesteria*-like organisms or potential *Pfiesteria*-related fish kills in shallow estuarine tributaries or embayments (Oldach 1999, Glasgow et al. 2001, Schmecl and Koltai 2001, Shoemaker 2001, Shoemaker and Hudnell 2001), and 10) lack of understanding about the water quality conditions associated with coincidence of HABs, fish kills, fish health and human health events in Chesapeake Bay and other estuaries (Burkholder et al studies, Tango et al. 2005, Tango and Butler 2008, Tango et al. 2008). While short term (3-day to 1 week), fixed site deployments of continuous monitoring data in offshore waters were conducted for programs such as U.S. EPA EMAP efforts in the 1980s and early 1990s (e.g. U.S. EPA 2004), long-term, seasonal scale deployments of water quality sensors in shallow water habitats took hold in Chesapeake in the late 1990s. In 1998, MD DNR used fixed site continuous water quality monitoring to track lower Pocomoke River conditions in a region of repeated fish kills (http://www.dnr.state.md.us/bay/cblife/algae/dino/pfiesteria/97v98.html).

From 1998 into the early 2000s MD DNR activated additional synoptic and shallow-water in-situ continuous water quality monitoring sites in other Chesapeake Bay river systems beyond the mid-channel long term water quality monitoring program stations. Data were collected to support greater scientific understanding and development of management options based on HAB-fish health-human health-water quality relationships (Tango et. al 2006).

In July 2001, the Chesapeake Bay Program Monitoring and Analysis Subcommittee’s Tidal Monitoring and Analysis Workgroup formed a Tidal Monitoring Design Team. Over 2 years, the team developed recommendations for implementing a monitoring network that addressed the primary objective of supplying the water quality information needed to assess the suite of new water quality criteria for dissolved oxygen, water clarity and chlorophyll a – ultimately with the goal of removing the Chesapeake Bay and its tidal rivers from the list of impaired waters (U.S. EPA 2007a). The design of the new Tidal Monitoring Network emphasized monitoring of the shallow water designated use areas. To capture temporal variability in dissolved oxygen, the new Tidal Monitoring Network incorporated high-frequency monitoring stations in surface and nearshore locations.

In 2003 the U.S. EPA Chesapeake Bay Program initiated a Shallow-Water Monitoring Program to complement the mid-channel assessments. The resulting shallow-water monitoring program uses 1) a network of fixed site, high temporal frequency sensors located throughout shallow water habitats of the Bay and its tidal tributaries to collect local water quality measurements on scales of 15 minutes to 1 hour, 2) monthly or biweekly Dataflow, high-density spatial mapping of surface water quality conditions where measurements are taken in-situ while underway and recorded at 3-4 second intervals.

More recently the Tidal Water Monitoring Networks have grown to include extended, season long deployments of offshore, high frequency vertical water quality monitoring profilers at two sites in Virginia (York mesohaline and Rappahannock mesohaline) and at short term deployment sites in other Maryland waters, high density measurements for continuous water quality monitoring with to complement the mid-channel long term biweekly to monthly water quality profiles of Bay conditions. A NOAA surface water high frequency water quality buoy data network has also evolved on the Bay. These high frequency data streams provide new opportunities for paired comparisons of nearshore and offshore habitat conditions.

Historical Analyses Regarding Comparability of Nearshore and Offshore water quality dynamics.

The question concerning comparability of nearshore and offshore, midchannel water quality is a Chesapeake Bay issue with precedent. Batiuk et al. (2000) noted several such studies between 1991 and 1996. The studies suggested mid-channel data can be used to describe nearshore conditions, however, not all studies were in agreement. This issue was further reviewed with long term water quality monitoring program data by Karrh 1999 and Batiuk et al. 2000. In a 1999 study, the Maryland Department of Natural Resources investigated the validity of using mid-channel data to assess nearshore areas. The 13-tributary study examined water quality at 127 nearshore stations compared to 54 adjacent mid-channel stations. The study found wide variations between nearshore and mid-channel data both within and between tributaries (U.S. EPA 2007a). However, these studies focused on parameters important to underwater grass habitat (Secchi depth, dissolved organic nitrogen, dissolved inorganic phosphorus, chlorophyll a, total suspended solids and salinity) and did not evaluate dissolved oxygen behavior.

At the time of publishing the U.S. EPA (2003a) Chesapeake Bay ambient water quality criteria there remained insufficient information about characteristic dissolved oxygen behaviors of offshore versus shallow, nearshore habitat to support separating the two habitats into their own designated use assessments. More recently an evaluation targeting advances in assessing short duration dissolved oxygen criteria using offshore low frequency and nearshore high frequency monitoring data was initiated during the Chesapeake Bay Monitoring Realignment (MRAT) process (MRAT 2009). With multiple years of shallow water data collection at a wide range of site conditions across the tidal Bay and in neighboring estuaries (e.g. the Maryland and Virginia Coastal Bays), and a foundation of analyses from the MRAT effort, an Umbrella Criteria Assessment Team (UCAT) was formed to advance short duration dissolved oxygen criteria attainment assessments. Byproducts of the UCAT effort help inform the issue of comparability between offshore and shallow water dissolved oxygen behavior.

Characteristics of Chesapeake Bay high frequency dissolved oxygen dynamics with an emphasis on shallow water habitat.

The Umbrella Criteria Assessment Team used more than a decade of high frequency dissolved oxygen data from Chesapeake Bay in characterizing dissolved oxygen behavior across multiple time scales and habitats (CBP STAC 2012). The combined data sets contained more than 1 million data points. Shallow water monitoring has shown intersite, interannual variability. High frequency water quality data in nearshore habitats show sometime frequent diel to sub-seasonal scale hypoxic and anoxic events different from the deep water patterns of Chesapeake Bay and its lower tidal tributaries. High frequency data collected with vertical profilers provided comparative data with shallow water continuous monitoring data for offshore, deep water sites over multiple depths at seasonal scales of time. The following section is a brief summary of the UCAT effort that expands the scientific understanding of dissolved oxygen behavior across gradients of time (i.e. instantaneous measures to interannual comparisons), space (nearshore and offshore) and habitat (tidal fresh to polyhaline salinities). The results underpin the later discussions on nearshore and offshore water quality comparisons with respect to dissolved oxygen dynamics.

SHALLOW WATER INTRASITE VARIABILITY.

Continuous monitoring data are collected to assess variability of water quality parameters throughout the day. Previous convention suggested shallow water did not experience significant low dissolved oxygen levels (U.S. EPA 2007a). Continuous monitoring data, however, often indicate a diel scale of hypoxia, some hypoxia is severe at locations (WB). Dissolved oxygen concentrations drop to low levels during the hours of darkness and sometimes reach dangerously low concentrations at or just after sunrise. (WB App 4, U.S. EPA 2007a).

Continuous monitoring stations have detected hypoxic and anoxic events beyond the diel scale. One example illustrated a Potomac River site capturing the intrusion of anoxic deep waters into shallow water (CBP STAC 2012). Degraded dissolved oxygen conditions persisted 48-72 hours while temperature and salinities were slower to recover. A second example from the Corsica River illustrated the impact of a nearly week-long event involving and algal die off, it's degradation effects reducing dissolved oxygen to anoxic conditions and the multiday recovery to normoxic conditions (CBP-STAC 2012). The event data informed scientists and managers about proximate causal links to a large fish kill.

A 12 site assessment from the Potomac River continuous monitoring stations shows daytime sampling results have a positive bias and nighttime sampling would create a negative bias compared with the measured distribution for dissolved oxygen concentrations based on high frequency sampling throughout the diel cycle. (EP Appendix 11 I think)

Daily mean dissolved oxygen concentration experienced at individual shallow water sites ranged broadly over the course of a season. (CB A1B). Across all time scales assessed (instantaneous, daily, weekly, monthly, seasonally and annually), we see interannual differences in dissolved oxygen measures within a monitoring site (WB, CB A1B).

SHALLOW WATER INTERSITE VARIABILITY

Dissolved oxygen measurements were evaluated with respect to criteria thresholds. Dissolved oxygen violations of Chesapeake Bay dissolved oxygen criteria can occur even at sites of good water quality. (WB). Boynton noted his analysis of DO conditions in the Patuxent River estuary with data from during the 1960s, a period before this system underwent severe eutrophication, showed there were still times (though not very frequent) when surface DO criteria would have been violated. Buchanan (Appendix 1) noted that daily mean dissolved oxygen concentrations experienced at individual shallow water sites differed between neighboring sites on the Potomac River. By contrast, in a related analysis using multiple years for 9 continuous monitoring stations on the Potomac River (Appendix 11) Perry found little evidence of changing variability in dissolved oxygen concentration distributions across the Potomac River locations from tidal fresh to mesohaline sites (n=9 sites, X yrs, EP).

SHALLOW WATER SEASONAL VARIATION

Seasonal shifts in DO concentrations frequency distributions were shown to have lower concentrations and broader ranges in mid-summer, higher concentrations and less variation for spring/early summer and autumn (Buchanan Appendix 1, Perry Appendix 11). Perry (Appendix 11) combined data from 9 Potomac River sites and suggested spring may be more variable than summer and autumn.

Buchanan (Appendix 1) computed daily means at the 20 tidal Potomac embayment and river flank stations from 2004-2008 showing a spring season range from 1.0 and 16.8 mg O_2/L, a summer range from 0.36-14.9 mg O_2/L and an autumn range of 3.1-14.0 mg O_2/L in autumn. The Continuous monitoring data on the Potomac River further showed that the range of diel DO variability experienced in shallow waters reached 11.0 mg O_2/L in spring, 17.52 mg O_2/L in summer and 10.8 mg O_2/L in autumn.

EXTERNAL FACTORS ASSOCIATED WITH DISSOLVED OXYGEN VARIABILITY IN CONTINUOUS MONITORING DATA

River flow:

Boyton et al (Appendix 4) evaluated dissolved oxygen dynamics for the mesohaline Potomac River St. George Island Continuous Monitoring site (2006-08). All time scales of dissolved oxygen criteria-related indicators showed an increase in violation rate related to increasing winter-spring (January-May) river flow. For the range of flow conditions, instantaneous and daily July-only % criterion failures reflected changes across the range of flows observed. 7-day means and 30-day means had threshold responses to flow while instantaneous and daily mean responses were more linear. (WB)

Eutrophication:

River flow tends to relate to nutrient loading of a system so the increasing violation rate with increasing river flow is a glimpse into possible shallow water dissolved oxygen patterns associated with trophic status of the water way. Boyton indicated qualitative inspection of shallow water continuous monitoring data showed that most severe diel-scale hypoxia is observed at sites experiencing severe eutrophication. (WB). By comparison, as mentioned earlier, Boynton's analysis of DO conditions in the Patuxent estuary during the 1960s, a period before this system underwent severe eutrophication, showed there were still times (not very frequent) when surface DO criteria would have been violated. (WB).

Comparisons among sites were made for duration of DO criterion violation (i.e. continuous time measured below a criterion value) were made between a site exposed to severe eutrophication in the Maryland Coastal Bays – Bishopville Prong, and St. George Island in the Potomac River with reasonably good water quality, a tidal freshwater site in an enriched estuary – Jug Bay, Patuxent River, and a mesohaline site exposed to open water (Pin Oak, Patuxent River). At the Bishopville Prong site with severe eutrophication there were many criteria failures and long durations of failure (12-24 hours or longer). At less impacted sites DO criteria failures were of shorter duration, especially for the instantaneous criteria. (WB). The duration of dissolved oxygen concentrations below criterion issue is further characterized in a graphic showing a positive relationship between the violation rate of the 30-day mean and maximum continuous time observed below the 30-day mean criterion threshold for multiple Chesapeake Bay continuous monitoring sites. The results demonstrate again that as one indicator of dissolved oxygen status in the Bay responds, so do other scales of dissolved oxygen indicators. The slope of such relationships can suggest which indicator is more or less sensitive to change and a good direction for further analyses.

Dissolved oxygen means and variability increased with increasing temperature and solar angle (Buchanan Appendix 1).

Multi-scale patterns of DO behavior – nearshore and offshore.

Chesapeake Bay dissolved oxygen criteria violation rates were used as indicators of water quality status. Bivariate comparisons across gradients of violation rates demonstrate similar patterns in nearshore and offshore waters. The sensitivity of the responses in this 2 dimensional space may differ depending on the basis of the criteria being compared (e.g. 30 day mean violation compared to a 7 day mean violation, 1 day mean compared with a 10th percentile of instantaneous measurements). Three common patterns observed across data from the two habitats are:

1) As 30 day mean DO concentrations increase, variability in DO concentrations increase. (…but tails off at high DO? Observed in both CB and EP results. Not critical but worth getting correct.)
2) As the summer 30 day mean increases, the probability of violating shorter duration criteria declines (e.g. 7 day mean, instantaneous minimum).
    a. A corollary of this appears to be that as any of the criteria means increase, there is a coincident improvement in other time scale dissolved oxygen water quality indicators.
    b. Caveat: However, one author finds violation rates for the instantaneous minimum remain higher than 10% even at high 30 day mean DO concentrations (CB).

3) As violation rates increase, duration of hypoxic/anoxic events increase. Coincidently then as measures of violation rates decline we observed event durations declines.

Bounding synergy between Offshore and Nearshore Dissolved Oxygen Dynamics: A Further Derivation From The 2012 Umbrella Criteria Assessment Team Analyses.

By virtue of exploring the statistical properties of high frequency dissolved oxygen data and testing analytical methods for integrating nearshore and offshore monitoring information into short duration dissolved oxygen assessments (see Appendix E in U.S. EPA 2007) the Umbrella Criteria Assessment Team gained powerful insights into comparability of dissolved oxygen behavior in nearshore and offshore habitats. The Chesapeake Bay dissolved oxygen water quality criteria attainment assessment procedure creates a 3-dimensional representation of dissolved oxygen concentration patterns for a designated use based on monitoring data. Therefore we have estimates of water quality conditions for nearshore and offshore areas. Estimate uncertainty in dissolved oxygen concentration will be least near the data collection sites, typically mid-channel, and greatest away from those sites away from the mid-channel region. Data collected on the biweekly to monthly time scale in shallow water regions has been added to the 30-day mean assessment improving the 30 day mean assessments. Through the implementation of the Chesapeake Bay Program shallow water monitoring program, and the addition of vertical profilers in offshore habitats, have we developed high frequency data sets allowing the integration of spatial variability more thoroughly into short duration dissolved oxygen assessments. Testing data integration methods, however, provided greater comparative understanding of nearshore vs. offshore dissolved oxygen behavior. The information forms support for decisions on maintaining shallow water as part of the open water designated use or separating it into a sub segment for analysis.

The foundation for using spectral analysis in the context of Chesapeake Bay dissolved oxygen criteria attainment assessment is published by Neerchal et al. (1992) and was then modified for use by Bay analysts (U.S. EPA 2003a, 2007a, MRAT 2009, CBP-STAC 2012). Spectral analysis was demonstrated as viable using high frequency dissolved oxygen Bay data (U.S. EPA 2003a, 2007a). The most recent example included estimating 3 years of DO patterns at the TF5.6 James River long term water quality monitoring site at depths from the surface to 9m (Robertson and Lane in MRAT 2009, also available in CBP-STAC 2012). A further test demonstrated intersite comparaibilty between nearshore continuous monitoring stations (Perry in CBP-STAC 2012). This technique was used to impart the information contained about dissolved oxygen behavior from a shallow water continuous monitoring station in one location into an estimate of dissolved oxygen behavior at another location. The most valuable tests of the UCAT effort relating to our purpose here involved nearshore to offshore comparisons. More specifically, analysts conducted several tests using nearshore continuous montoring data to inform estimates of offshore conditions. The results were compared with known dissolved oxygen conditions from high frequency vertical profiles of water quality at that site.

Robertson and Lane in CBP STAC (2012) conducted comparisons of spectrally derived dissolved oxygen patterns at an offshore location from continuous monitoring data information. They compared the output with York River polyhaline vertical profiler data and found, for daily averages, spectrally derived dissolved oxygen patterns were statistically different in variability and trend from actual measurements. However, weekly averages show statistically similar variability to the real data.

Hall (CBP-STAC 2012 Appendix 8) used two separate nearshore continuous monitoring sites on the lower, mesohaline Potomac River to generate estimates of offshore dissolved oxygen behavior. Results were compared in terms of dissolved oxygen criteria violation rates using predicted and actual measurements from the Sandy Point offshore vertical water quality profiler sitting in 15 m of water. Criterion violation rates were highly comparable in the surface waters (1,3,5m) and fairly comparable with slightly more variation for the deeper water. The pycnocline appeared most problematic. Both synthetic data sets showed identical failures for the 30-day mean criterion and nearly identical 7-day mean and instantaneous minimum criterion failure rates. 30-day and 7 day means assessments translate well between nearshore and offshore in summer at this lower Potomac River MH area. There were greater discrepencies when dealing with instantaneous minimum results at high frequencies.

In the range of dissolved oxygen values near water quality criteria the analysis of the spectral analysis derived dissolved oxygen behavior produced results identical to real data. If we modify the analysis to look for differences i.e. if we test theoretically higher threshold dissolved oxygen criteria, we saw

- At 1m the analysis overestimates violations (data tendency to show lower DO but above present criteria)
- At 3m – small window of overestimating violation rate.
- At 5m – short interval of underestimation
- The patterns are similar for the 7 day mean.

This suggests slight changes in a criterion would not change violation rates in this form of assessment.

DISCUSSION

Any differences that exist between dissolved oxygen dynamics in the open water habitats of nearshore and offshore waters may be significant for the health of Bay life. The mesohaline mainstem Chesapeake Bay and lower reaches of the major tidal tributaries have a stratified water column which essentially prevents waters near the bottom from mixing with oxygenated surface waters (U.S. EPA 2003a). The Chesapeake Bay Partnership's long term water quality monitoring program has conducted significant monitoring and modeling-based characterizations of dissolved oxygen dynamics of the Bay and its tributaries based on mid-channel, fixed station assessments.

However, understanding horizontal patterns including shallow water versus offshore dissolved oxygen variability in the above pycnocline open waters of the Bay has received less attention.

The assumption underlying the combination of habitats for assessment and management is based on fish and shellfish physiology. These two habitats are shared by the open water fish and shellfish communities and the physiological oxygen requirements of these species are met the same way across habitats. That assumption addresses habitat needs from the perspective of physiology with an additional implicit assumption that the two habitats offer the same dissolved oxygen stresses across space and time. Fish and shellfish have a variety of homeostatic mechanisms to accommodate temporal shifts in habitat conditions (e.g. fish can swim away from poor habitat, some fish may gulp air under adverse dissolved oxygen stress, blue crabs may leave the water for a short period of time, selective fecundity and mortality, changes in death rates etc. Wooton 1990). If the two habitats provided the same habitat conditions on average, fish and shellfish individuals and populations could adjust their behavior while exercising the flexibility of their physiological tolerances and buffer against many environmental changes.

The Chesapeake Bay's nearshore shallow water can be exposed to rapid fluctuations in dissolved oxygen (Sanford et al. 1990). The presence of abundant, diverse primary producer communities (i.e., phytoplankton, SAV, benthic algae, macroalgae, epiphytic algal communities) in shallow water can rapidly generate large amounts of oxygen during daylight hours. Conversely, close proximity of oxygen-consuming benthic organisms and sediment processes in shallow water habitats can rapidly drive down oxygen levels at night (D'Avanzo and Kremer 1994). Shallow waters periodically experience episodes of low to no dissolved oxygen (U.S. EPA 2003a). In nearshore waters of the mesohaline Chesapeake Bay, near bottom dissolved oxygen concentrations are characterized by large diel fluctuations and daily minima during the late night and early morning hours of July and August (Breitburg 1990). Depths as shallow as 4 meters, dissolved oxygen concentrations may decline to 0.5 mg/L for up to 10 hours (Breitburg 1990, U.S. EPA 2003a, Batiuk et al. 2009). Given the underwater grasses definition of shallow water goes out to the 2m contour based on SAV light requirements, and Breitburg, U.S. EPA and Batiuk et al reference "shallow" as 4 m, this points to an important issue that we presently lack consensus on a dissolved oxygen-based definition of shallow water. Such a definition would be important to a recommendation for separating shallow water from offshore water in monitoring, assessing and managing habitats in Chesapeake Bay.

By comparison, the influence of abiotic signals from large hydrodynamic forcing factors such as the passing of weather fronts can be evident in shallow waters (e.g. R.C. Jones 7-day front patterns reference) but may be stronger in regulating dissolved oxygen variability for offshore habitats (e.g. de Jonge 1994/seiching?). Therefore, proximate differences in prominent forcing functions on dissolved oxygen behavior between offshore and shallow, nearshore waters may translate to characteristic differences in space-time patterns of dissolved oxygen stress. If there are differences in the frequency distributions of hypoxic events and their duration, then we have the foundation for the community question of whether we need to separate the habitats for monitoring, assessment and management.

Patterns of Response important to Management.

Part of the insight gained in the UCAT effort is that dissolved oxygen variability could be related back to multiple factors such as temperature, solar angle, river flow, and at least qualitatively eutrophication. Caffrey (2004) used high frequency dissolved oxygen data from 42 sites in 22 National Estuarine Research Reserves to assess net ecosystem metabolism. The sites were located nearshore in shallow water averaging 1.9m. Diel dissolved oxygen dynamics are used to underpin an understanding of higher level ecosystem processes. Factors affecting NEM therefore inform us about influences on dissolved oxygen variability. Temperature was the most important environmental factor explaining within-site variation of metabolic rates; nutrient concentrations were the second most important factor. Habitat adjacent to the monitoring site explained general trends in NEM across sites. Estuarine area was also significant in explaining patterns in NEM. Adjacent habitat, estuarine area and salinity explained 58% of the variation in NEM. Nutrient loading explained 68% of the variation in NEM among some of the sites. Metabolic rates of the NERR sites were much larger than rates from other studies. One explanation was that small, generally shallow sites located near shore may have greater allochthonous organic inputs as well as significant benthic primary production than large deep systems represented by the literature. From the perspective of understanding shallow water versus offshore water habitats, this is additional evidence that the nearshore zone has different high frequency dissolved oxygen dynamics than the offshore zone. However, important to Bay management will be patterns of responsiveness to management actions. Are shallow waters responsive to management actions?

Caffrey (2004) suggests careful selection of monitoring stations that are representative of the estuary as a whole could be used to estimate estuary-wide metabolic rates. Estimates from small tidal creeks may not be representative of the estuary as a whole, however, detection of changes in the watershed such as changes in nutrient loading, may be more apparent in shallow water than out in the channel. Caffrey shows dissolved oxygen patterns expressed as NEM for shallow water sites are responsive to changes in nutrient loading. Boyton's analyses on river flow and qualitatively for eutrophication suggest the same. The Chesapeake Bay TMDL is built upon the model that reducing nutrients will improve dissolved oxygen resources. Therefore, we have large scale and local scale measures of dissolved oxygen behavior that converge on a response behavior of improved dissolved oxygen conditions with reductions in eutrophication pressures.

The patterns are repeated across many scales of indicators nearshore and offshore. Jordan et al. 1992 expected changing frequencies in event duration to reflect improving and degrading conditions based on high frequency offshore measures (Figure X). For nearshore habitat of the Corsica River, Boyton et al. (2010?) suggest reduction of hypoxic hours with improvements in Nitrogen loading. Caffrey (2004) found greater autotrophic production with increasing DIN supporting again the idea of improving dissolved oxygen conditions via reductions in NEW when nutrients are reduced. Further, Hall and others in this report illustrated as the 30-day mean violation rate at continuous monitoring stations declines, shorter-duration criteria violation rates also decline; the rate of decline between violations for different temporal criteria scales may not be 1:1.

This may be a significant point of consideration. Most every indicator of dissolved oxygen behavior is being shown to respond by improving as water quality status improves. However, do all criteria reach the criteria thresholds even with the given buffer of 10% allowable exceedances? Buchanan (Appendix 1) suggests the instantaneous minimum values of shallow water may not come into compliance coincident with a 30-day mean that is near double the criteria threshold value. This analysis was Potomac-only data. By comparison, the regression relationship of the 30-day mean and instantaneous minimum violations for continuous monitoring data including multiple tributaries has a Y intercept that is also above zero, but the real data near the zero intercept may be nonlinear and suggest qualitatively that the instantaneous minimum data arrive near zero before the 30-day violation rate reaches zero. Elgin's work + Claire's work. Highlight here. Elgin did the inst min analysis. I believe Elgin found that, like Claire, the inst min was tough to meet unless the criteria threshold was increased. This may suggest not a wholesale separation for all assessments but special consideration needed regarding the impact of shallow water volume in a segment when shallow water can't meet the existing threshold goal.

OPTIONS AND IMPLICATIONS OF REFINING THE OPEN WATER DESIGNATED USE; A preworkshop outline of future recommendations.

I. The case for keeping shallow water embodied within the open water designated use

II. The case for separating shallow water as a dissolved oxygen based sub-segment in the open water designated use.

   a. Setting the offshore/nearshore boundary – what separates shallow water from offshore water for dissolved oxygen assessments
   b. Same criteria, separate criteria – what are they and why?
   c. Applicable seasons and definitions
   d. Monitoring
   e. Assessment

III. Management implications of the two cases.

Workshop tasks:

(Supporting future Chapter 4 recommendations)

I. Review relevant analyses regarding dissolved oxygen behavior in nearshore and offshore habitats.

II. Using the best available information on dissolved oxygen dynamics:
   1) Make a case for keeping the shallow water embodied in the open water designated use for Clean Water Act dissolved oxygen criteria attainment assessments.
   2) Make a case for separating the shallow water from the open water designated use.

In each case, define the implications, pro and con, of maintaining the existing framework and assessment compared with separating shallow water from offshore waters of the open water designated use.

Issues to consider include but are not limited to:

- What is a dissolved oxygen-based definition of the offshore/shallow water boundary?
  - Provide a recommendation
- For the option of separating the habitats,
  - do the existing open water criteria continue to apply? Is there fish and shellfish science to support a separate set of criteria for the nearshore versus offshore waters?
    - Provide a recommendation
  - is the suite of criteria applied the the same in Offshore open water and shallow water habitat? (Instantaneous minimum, 1-day mean, 7 day mean, 30 day mean?)
    - Provide a recommendation
- Do the seasons change for the assessments? If so, provide the recommendation and the science that supports the change.
- What do we recommend as the monitoring plan for assessment of shallow water dissolved oxygen criteria attainment?
- What do we recommend is the monitoring plan to support assessment of offshore open water habitat
- What changes in the analytical protocol are necessary for assessing the suite of criteria in offshore and shallow water habitats if they are separated?
- What are the potential management implications of separating shallow water and offshore waters for dissolved oxygen assessment?

Literature Cited needing editing and adjusting…

Burkholder JM, Glasgow HB. 2002.
The life cycle and toxicity of Pfiesteria piscicida revisited
Journal of Phycology 38 (6): 1261-1267.

Burkholder JM, Glasgow HB, Deamer-Melia NJ, et al. 2001.
Species of the toxic Pfiesteria complex, and the importance of functional type in data interpretation
Environmental Health Perspectives 109: 667-679 Suppl. 5.

Burkholder JM, Glasgow HB, Deamer-Melia N. 2001.
Overview and present status of the toxic Pfiesteria complex (Dinophyceae)
Phycologia 40 (3): 186-214.

Burkholder JM, Glasgow HB. 1997.
Pfiesteria piscicida and other Pfiesteria-like dinoflagellates: behavior, impacts, and environmental controls
Limnology and Oceanography 42 (5): 1052-1075 Part 2.

Glibert PM, Burkholder JM. 2006.
Toward an emerging consensus on the ecology of Pfiesteria
Harmful Algae 5 (4): 339-341.

Moeller PDR, Beauchesne KR, Huncik KM, et al. 2007.
Metal complexes and free radical toxins produced by Pfiesteria piscicida
Environmental Science & Technology 41 (4): 1166-1172.

Moeller PDR, Morton SL, Mitchell BA, et al. 2001.
Current progress in isolation and characterization of toxins isolated from Pfiesteria piscicida
Environmental Health Perspectives 109: 739-743 Suppl. 5.

Brownie C, Glasgow HB, Burkholder JM, et al. 2003.
Re-evaluation of the relationship between Pfiesteria and estuarine fish kills
Ecosystems 6 (1): 1-10.

Burkholder JM, Gordon AS, Moeller PD, et al. 2005.
Demonstration of toxicity to fish and to mammalian cells by Pfiesteria species: Comparison of assay methods and strains
Proceedings of the National Academy of Sciences of the United States of America 102 (9): 3471-3476.

Drgon T, Saito K, Gillevet PM, et al. 2005.
Characterization of Pfiesteria ichthyocidal activity - Authors' reply
Applied and Environmental Microbiology 71 (10): 6464-6464.

Drgon T, Saito K, Gillevet PM, et al. 2005.
Characterization of ichthyocidal activity of Pfiesteria piscicida: Dependence on the dinospore cell density
Applied and Environmental Microbiology 71 (1): 519-529.

Dykstra MJ, Kane AS. 2000.
Pfiesteria piscicida and ulcerative mycosis of Atlantic menhaden - Current status of understanding
Journal of Aquatic Animal Health 12 (1): 18-25.

Glasgow HB, Burkholder JM, Mallin MA, et al. 2001.
Field ecology of toxic Pfiesteria complex species and a conservative analysis of their role in estuarine fish kills
Environmental Health Perspectives 109: 715-730 Suppl. 5.

Gordon AS, Marshall HG, Shumway SE, et al. 2005.
Characterization of Pfiesteria ichthyocidal activity
Applied and Environmental Microbiology 71 (10): 6463-6463.

Lovko, VJ; Vogelbein, WK; Shields, JD; et al. 2003.
A new larval fish bioassay for testing the pathogenicity of Pfiesteria spp. (Dinophyceae)
Journal of Phycology 39 (3): 600-609.

Stow CA. 1999.
Assessing the relationship between Pfiesteria and estuarine fishkills
Ecosystems 2 (3): 237-241.

Tango P, Magnien R, Goshorn D, et al. 2006.
Associations between fish health and Pfiesteria spp. in Chesapeake Bay and mid-Atlantic estuaries

Harmful Algae 5 (4): 352-362.

Vogelbein WK, Lovko VJ, Shields JD, et al. 2002.
Pfiesteria shumwayae kills fish by micropredation not exotoxin secretion
Nature 418 (6901): 967-970.

Wang Q; Deeds JR; Place AR; et al. 2005.
Dinoflagellate community analysis of a fish kill using denaturing gradient gel electrophoresis
Harmful Algae 4 (1): 151-162.

Collier DN, Burke WA. 2002.
Pfiesteria complex organisms and human illness
Southern Medical Journal 95 (7): 720-726.

El-Nabawi A, Quesenberry M, Saito K, Silbergeld E, et al. 2000.
The N-methyl-D-aspartate neurotransmitter receptor is a mammalian brain target for thee dinoflagellate Pfiesteria piscicida toxin.
Toxicological Applications in Pharmacology 169: 84-93.

Grattan LM
Human health risks of exposure to estuary waters. 2001.
Human and Ecological Risk Assessment 7 (5): 1385-1391.

Grattan LM, Oldach D, Perl TM, et al. 1998.
Learning and memory difficulties after environmental exposure to waterways containing toxin-producing Pfiesteria or Pfiesteria-like dinoflagellates
Lancet 352 (9127): 532-539.

Griffith D. 1999.
Exaggerating environmental health risk: the case of the toxic dinoflagellate Pfiesteria
Human Organization 58 (2): 119-127.

Haselow DT, Brown E, Tracy K, et al. 2001.
Gastrointestinal and respiratory tract symptoms following brief environmental exposure to aerosols during a Pfiesteria-related fish kill
Journal of Toxicological and Environmental Health A 63: 553-564.

Kleindinst JL, Anderson DM. 2001.
Pfiesteria-related educational products and information resources available to the public, health officials, and researchers
Environmental Health Perspectives 109: 695-698 Suppl. 5.

Levin ED, Blackwelder WP, Glasgow HB Jr et al. 2003.
Learning impairment caused by a toxin produced by Pfiesteria piscicida infused into the hippocampus of rats.
Neurotoxicology & Teratology 25: 419-426.

Levin ED. 2001.
A rat model of the cognitive impairment from Pfiesteria piscicida exposure
Environmental Health Perspectives 109: 757-763 Suppl. 5.

Moe CL, Turf E, Oldach D, et al. 2001.
Cohort studies of health effects among people exposed to estuarine waters: North Carolina, Virginia, and Maryland
Environmental Health Perspectives 109: 781-786 Suppl. 5.

Morris JG, Grattan LM, Wilson LA, et al. 2006.
Occupational exposure to Pfiesteria species in estuarine waters is not a risk factor for illness
Environmental Health Perspectives 114 (7): 1038-1043.

Morris JG. 2001.
Human health effects and Pfiesteria exposure: A synthesis of available clinical data
Environmental Health Perspectives 109: 787-790 Suppl. 5.

Oldach D. 1999.
Regarding Pfiesteria
Human Organization 58: 459-460.

Schmechel DE, Koltai DC. 2001.
Potential human health effects associated with laboratory exposures to Pfiesteria piscicida
Environmental Health Perspectives 109: 775-779 Suppl. 5.

Shoemaker RC, Lawson W. 2007.
Pfiesteria in estuarine waters: The question of health risks
Environmental Health Perspectives 115 (3): A126-A127.

Shoemaker RC. 2001.
Residential and recreational acquisition of possible estuary-associated syndrome: a new approach to successful diagnosis and treatment
Environmental Health Perspectives 109 (5): 791-796.

Shoemaker RC, Hudnell HK. 2001.
Possible estuary-associated syndrome: Symptoms, vision, and treatment
Environmental Health Perspectives 109 (5): 539-545.

Swinker M, Tester P, Attix DK, et al. 2002.
Human health effects of exposure to Pfiesteria piscicida: a review
Microbes and Infection 4 (7): 751-762.

Deeds, J.R., Terlizzi, D.E., Adolf, J.E., Stoecker, D.K., Place, A.R. 2002 Toxic activity from cultures of *Karlodinium micrum* **(***= Gyrodinium galatheanum)* **(Dinophycaea) – a dinoflagellate associated with fish mortalities in an estuarine aquaculture facility.** *Harmful Algae* **1 169-189.**

Marshall, H. 2003. Toxic algae: their presence and threat to Chesapeake Bay, USA. In: Algae and their Biological State in Water. Acta Botanica Warmiae et Masuriae, Olsyztyn. 3:51-60.

Marshall, H.G. 1996. Toxin producing phytoplankton in Chesapeake Bay. Virginia J. Sci. 47:29-37.

Caffery, 2004.

Tango, P. and W. Butler. 2008. Cyanotoxins in tidal waters of Chesapeake Bay. Northeast Naturalist. 15(3):403-416.

Tango, P., W. Butler, and B. Michael. 2008. Cyanotoxins in the tidewaters of Maryland's Chesapeake Bay: The Maryland experience. Pp 179-180 in Hudnell, K. (Ed.) Proceedings of the International Symposium on Cyanobacterial Harmful AlgalBlooms, Research Triangle Park, Raleigh, North Carolina: State of the Science and Research Needs. Springer.

Tango, P., R. Magnien, W. Butler, R. Lacouture, M. Luckenbach, C. Poukish and C. Luckett. 2005. Impacts and potential effects due to *Prorocentrum minimum* blooms in Chesapeake Bay. Harmful Algae. 4:525-531.

Wooton, R. J. 1990. Ecology of Teleost Fishes. Chapman and Hall, New York, NY. 400pp.