Heterogeneity in Nantucket Harbor: A Case Study in Spatial Variation, Temporal Exposure, and Coastal Habitat Assessment

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Abstract

Nantucket Harbor has experienced substantial changes in water quality, together with losses of important biological resources that have long sustained the island. Both local and global effects have been linked to changes in nutrient concentration, water clarity, temperature and dissolved oxygen, which in turn have been associated with a decrease in eelgrass coverage and variation in bay scallop abundance. In this study I analyze data from two complementary water quality monitoring programs: the Nantucket Natural Resources Department record from 2010-2025 at four different sites across the harbor, and the continuous record from the Maria Mitchell Association monitoring buoy in 2025 at one lower harbor location. The NRD discrete sampling record revealed spatial variation and long-term temporal patterns in total nitrogen (TN), water clarity and chlorophyll-*a* concentration, with more enclosed sites characterized by more stressful values. There has been no significant change in water clarity and chlorophyll-*a* over the 15-year study period, while TN values appear to have been decreasing at some of the sites. Nevertheless, even in sites with apparent improvement, TN values frequently exceeded the 25 μM threshold established by the Massachusetts Estuary Project. The continuous data from the MMA buoy revealed that organisms in the harbor were often exposed to temperatures above the 23 °C sublethal stress threshold for eelgrass. Moreover, dissolved oxygen levels below 6 mg/L sometimes co-occurred with elevated temperatures, potentially exacerbating the impact of each stressor. There is urgency in understanding the relationship between harbor water conditions and the changes in eelgrass coverage and bay scallop abundance reported in separate Town studies. The two analyses described in this study provide complementary information needed to understand those relationships: the NRD record reveals spatial variation and long-term trends, while the continuous buoy record reveals short-term variation and co-occurring stressors that discrete sampling cannot resolve.

Keywords: Nantucket Harbor; eutrophication; eelgrass; dissolved oxygen; thermal exposure; spatial heterogeneity; coastal monitoring

1. Introduction

The human history of Nantucket, an island approximately 48 km (30 miles) south of Cape Cod, Massachusetts, has been shaped by Nantucket Harbor, a large, shallow embayment on the northern side of the island (Massachusetts Historical Commission, 1987). The harbor encompasses roughly 21.3 km^2 (5,254 acres), making it one of the largest enclosed bays in southeastern Massachusetts (Starbuck et al., 2025). The biodiversity of the harbor is anchored by eelgrass (*Zostera marina*) beds that stabilize sediments, improve water quality, and provide feeding, refuge, and nursery habitat for many species, including the iconic Nantucket bay scallop (*Argopecten irradians*) (Starbuck et al., 2025; Heck, 2025). In fact, Nantucket Harbor bay scallop populations are among the few such populations that still support both commercial and recreational fisheries (Heck, 2025).

However, long-term benthic monitoring of the eelgrass on which bay scallops and other animals depend shows substantial decline: mean eelgrass cover at consistently surveyed Nantucket Harbor sites fell from 37% in 2006 to 14% in 2024, while mean cover at sites that had more than 40% eelgrass cover in 2006 fell from 56% to 18% over the same period (Heck, 2025). Efforts to understand the causes of this decline have centered on analyses of harbor water quality. In this study, I examine two distinct datasets that describe harbor water quality at different scales of time and space.

One source of the data I used was generated by the Town of Nantucket Natural Resources Department (NRD) and its collaborators. This valuable monitoring record includes water-quality data from discrete samples across the harbor, as well as surveys of eelgrass, bay scallops, and other benthic organisms (Heck, 2025; Town of Nantucket Natural Resources Department, 2024; Starbuck et al., 2025). I used the water-quality data to examine broad spatial patterns and long-term change, and to see whether these patterns provided insight into the changes in eelgrass coverage and bay scallop abundance reported in the biological surveys.

In 2024, the Maria Mitchell Association (MMA) deployed a high-frequency water-quality monitoring buoy in Nantucket Harbor, generating the second dataset I used in this study. The buoy records temperature, dissolved oxygen (DO), pH, salinity, and chlorophyll-*a* at short time intervals. In this study, I focused on temperature and DO as exposure variables. The continuous monitoring enabled me to examine potentially stressful conditions experienced by organisms over minutes, hours, and months.

Continuous monitoring is well suited to examining stressors that vary temporally. In Long Island Sound, for example, continuous DO records resolved variation between weekly surveys and produced different estimates of the duration of hypoxic conditions (Duvall et al., 2024). Similarly, work in a shallow hypereutrophic estuary showed that low-oxygen exposure depends not only on whether hypoxia occurs, but also on its timing, duration, frequency, and vertical distribution (Jarvis et al., 2023). Temperature can further modify the biological significance of low oxygen because warming increases metabolic demand and can intensify the effects of hypoxia on coastal organisms (Gurr et al., 2018; Tomasetti et al., 2023).

Bay scallops and eelgrass are vulnerable not only to lethal extremes, but also to repeated sublethal exposure. Bay scallops show physiological responses to diel-cycling hypoxia, including elevated cardiac activity near DO thresholds and reduced metabolic performance under low oxygen (Gurr et al., 2018). Combined warming and hypoxia can reduce northern bay scallop feeding, performance, and survival (Tomasetti et al., 2023). Eelgrass is likewise affected by interacting environmental conditions, including temperature and salinity, and Massachusetts estuaries such as Waquoit Bay show how warming, deoxygenation, acidification, eutrophication, and eelgrass loss can occur together in shallow systems (Salo and Pedersen, 2014; Long and Mora, 2023). Thus the importance of continuous high-frequency monitoring is that it permits assessment of overlapping sublethal stressors as well brief extreme conditions.

In this study, I use the NRD monitoring data to examine differences among sites and long-term changes in water quality across the harbor. I use the high-frequency MMA buoy data to examine daily variation at one lower-harbor location, including the recurrence and co-occurrence of potentially stressful temperature and DO conditions. Together, the two datasets allow me to examine harbor water quality at spatial and temporal scales that neither record can resolve alone.

2. Materials and methods

2.1 Study system and monitoring framework

I analyzed two water-quality datasets from Nantucket Harbor, Massachusetts: long-term site-based monitoring by the Town of Nantucket Natural Resources Department (NRD), and continuous measurements collected at short intervals from the Maria Mitchell Association (MMA) buoy. The NRD dataset includes repeated measurements from multiple harbor sites, whereas the MMA buoy dataset consists of high-frequency measurements from a fixed location; both are shown in Fig. 1.

Together, the datasets provide complementary information: the NRD data support spatial comparisons and evaluation of long-term change across Nantucket Harbor, while the MMA buoy data support analyses of short-term temporal variation and exposure at one location.

2.2 NRD water-quality dataset

I obtained data for NRD sites NAN1, NAN2, NAN3, and NAN5 from the NRD Water Quality Analysis and Visualization (WQAV) portal. I analyzed total nitrogen, Secchi depth, and chlorophyll-*a* for the period 2010–2025. These variables were selected because they represent related but distinct aspects of water quality: total nitrogen as a measure of nutrient enrichment, chlorophyll-*a* as an index of phytoplankton biomass, and Secchi depth as a measure of water clarity and light penetration.

I removed six TN values greater than 60 μM because they were isolated extremes and likely anomalous. Because Secchi depth was repeated across depth-coded rows, I retained one observation per site and sampling date. I excluded two negative chlorophyll-*a* values. Four chlorophyll-*a* values greater than 30 μg/L were retained in the statistical analyses but omitted from Figs. 7 and 8 to improve resolution of the remaining observations.

2.3 Spatial analyses of NRD data

For total nitrogen, I plotted individual concentrations by site and year and compared them with a common threshold of 25 μM (0.35 mg N/L), based on the Massachusetts Estuaries Project target concentration for Nantucket Harbor and its subsequent use in Town monitoring reports (MassDEP, 2009; Samimy et al., 2025). I also displayed the distribution of observations at each site with boxplots. To evaluate temporal change, I calculated annual site medians, tested trends with Mann-Kendall tests, and estimated rates of change with Sen's slope. To compare sites while accounting for temporal variation, I log-transformed TN and fitted a linear model with site, year, and month as predictors. Site and month were categorical, and year was continuous. I tested the overall effect of site with an F-test and used unadjusted pairwise contrasts to compare sites.

I plotted individual Secchi depth observations by site and summarized the distributions with boxplots. I compared sites using one-way ANOVA on all site-date observations and repeated the analysis using annual site means to reduce the influence of unequal sampling among years. I used Tukey HSD to compare individual sites. To evaluate temporal change, I modeled annual site means as a function of year and site. As a sensitivity analysis, I modeled all observations as a function of year, site, and month and also examined trends separately at each site.

I first tested whether chlorophyll-*a* differed with sampling depth by modeling log(chlorophyll-*a* + 1) as a function of site, year, month, depth, and the site × depth interaction. Because no significant depth-related effect was detected, I pooled surface, middle, and bottom samples for the remaining analyses. To evaluate temporal change, I calculated annual medians for each site on the original scale, transformed them as log(median + 1), and modeled them as a function of year and site. I compared sites using a model of log(chlorophyll-*a* + 1) with site, year, and month as predictors. I used Tukey HSD on the log-transformed observations to compare individual sites.

2.4 MMA buoy dataset, instrumentation, and analyses

During the 2025 deployment, the MMA buoy's YSI multiparameter sonde was positioned approximately 0.8–0.9 m below the water surface and recorded measurements at 15-minute intervals. The sonde measured water temperature and DO and reported DO as both concentration (mg/L) and percent saturation.

Timestamps were converted from UTC to local time. I removed 153 all-zero temperature and DO records that represented instrument output rather than environmental measurements. Gaps were retained as missing values and were not interpolated.

I used operational thresholds of temperature >23 °C and DO <6 mg/L to characterize potentially sublethal exposure. The temperature threshold represents a level above which respiration may increasingly compromise eelgrass carbon balance and physiological performance (Lee et al., 2007; Jabbari et al., 2024). The DO threshold identifies conditions that may affect eelgrass physiology and elicit physiological responses in bay scallops (Vaquer-Sunyer and Duarte, 2008; Raun and Borum, 2013; Gurr et al., 2018).

I calculated daily degree-hours above 23 °C by summing the temperature exceedance above 23 °C across 15-minute intervals, with each interval counted as 0.25 hour. Daily totals were calculated only for days with at least 87 of the 96 expected measurements. I also identified observations in which temperature exceeded 23 °C while DO was below 6 mg/L and compared buoy DO saturation with surface, middle, and bottom measurements from nearby NRD site NAN1.

2.5 Statistical analyses and software

Statistical analyses and figure preparation were conducted in Python. I used pandas for data management and screening, NumPy and SciPy for numerical analyses, statsmodels for statistical modeling, and Matplotlib for figure preparation. Statistical significance was evaluated at $\alpha = 0.05$, with exact p-values reported where appropriate.

3. Results

3.1 Long-term NRD monitoring reveals temporal trends and spatial heterogeneity across Nantucket Harbor

Nantucket Harbor water-quality conditions varied across four NRD monitoring sites representing different hydrologic settings: NAN1 in the lower harbor, likely the best-flushed site; NAN2 in the mid-harbor; NAN3 near the head of the harbor; and NAN5 in Polpis Harbor, a more enclosed and likely less well-flushed embayment. This site sequence provides a basis for evaluating water-quality variation along a lower-harbor to upper/enclosed-harbor gradient.

Individual total nitrogen (TN) concentrations from 2010–2025 frequently exceeded the 25 μM (0.35 mg N/L) threshold, particularly at the upper and more enclosed harbor sites (Fig. 2). Annual median TN declined at all four sites during 2010–2025 (Fig. 3). The declines were significant at NAN2 ($p = 0.048$) and NAN3 ($p = 0.029$), but fell just short of the $\alpha = 0.05$ significance criterion at NAN1 ($p = 0.059$) and NAN5 ($p = 0.060$).

TN concentrations differed among the four monitoring sites, showing clear spatial heterogeneity across Nantucket Harbor (Fig. 4). Site was a significant predictor of TN concentration after accounting for year and month ($F = 51.06$, $p < 0.0001$). TN was lowest at NAN1 and highest at NAN5, the more enclosed site; NAN2 and NAN3 were intermediate. Pairwise comparisons indicated that all four sites differed significantly from one another.

Secchi depth, a proxy for water clarity, was greatest at NAN1, lower at NAN2 and NAN3, and lowest at NAN5, indicating lower water clarity at the upper and more enclosed harbor sites (Fig. 5). There was no evidence of temporal improvement in Secchi depth: year was not a significant predictor when annual site means were analyzed (slope = -0.011 m yr^{-1}, $p = 0.608$) or when all observations were analyzed while accounting for site and month (slope = -0.0140 m yr^{-1}, $p = 0.142$).

Secchi depth showed a spatial pattern consistent with the TN results (Fig. 6): water clarity was greatest at NAN1, where TN was lowest, and lowest at NAN5, where TN was highest; NAN2 and NAN3 were intermediate. Site differences were significant using all Secchi observations ($F = 54.36$, $p < 0.0001$) and remained significant when the analysis was repeated using annual site means to reduce the influence of unequal sampling ($F = 40.03$, $p < 0.0001$). Pairwise comparisons indicated that all four sites differed significantly from one another.

Chlorophyll-*a* concentrations were generally below 5 μg/L across the harbor sites, although elevated values occurred more often at NAN3 and NAN5 than at NAN1 and NAN2 (Fig. 7). Episodic higher values occurred at all sites, but the largest and most frequent elevated observations were concentrated at the upper and more enclosed harbor sites. Chlorophyll-*a* showed no significant temporal trend during the 2010–2025 study period; annual site medians showed no significant change over time (slope = -0.0018 log units yr^{-1}, $p = 0.819$).

The site-level distributions reinforced this spatial pattern (Fig. 8). Chlorophyll-*a* concentrations were lower at NAN1 and NAN2 and higher at NAN3 and NAN5, consistent with the TN and Secchi results showing poorer water-quality conditions at the upper and more enclosed sites. After accounting for year and month, chlorophyll-*a* differed significantly among sites ($F = 23.49$, $p < 0.0001$). Tukey HSD pairwise comparisons indicated that NAN1 and NAN2 did not differ significantly, and neither did NAN3 and NAN5; however, chlorophyll-*a* was significantly higher at both NAN3 and NAN5 than at NAN1 and NAN2.

Across TN, Secchi depth, and chlorophyll-*a*, the NRD data show persistent spatial heterogeneity in Nantucket Harbor. The lower-harbor site generally had lower nitrogen, greater water clarity, and lower chlorophyll-*a*, whereas the upper and more enclosed sites had higher TN, lower clarity, and more frequent elevated chlorophyll-*a* values.

3.2 High-frequency buoy data reveal short-term exposure conditions

The high-frequency MMA buoy record captured temperature variation across the 2025 deployment (Fig. 9). Temperatures increased from spring into midsummer, remained highest during July and early August, and declined through late summer and fall. During midsummer, temperatures repeatedly exceeded the 23 °C sublethal exposure threshold. Daily degree-hours above 23 °C, which integrate both the magnitude and duration of threshold exceedance, varied between June 23 and September 5, 2025, and were greatest in late July (Fig. 10).

Figure 11 shows the co-occurrence of warm-water and low-oxygen conditions. Temperature exceeded 23 °C while DO was below 6 mg/L in 191 of 14,743 valid paired observations (1.3%). DO was ≥6 mg/L in 97.2% of observations, between 5 and <6 mg/L in 2.4%, and <5 mg/L in 0.4%.

In Figure 12, I compare the continuous buoy record with eight distinct DO measurements from nearby NAN1. All eight measurements fell within the range recorded by the buoy on the corresponding sampling day. However, the buoy record shows substantial short-term variation in DO that was not captured by the discrete NAN1 measurements.

4. Discussion

Like many coastal New England waters, Nantucket Harbor is exposed to both global and local stressors that may affect the organisms that inhabit it. Among local stressors, nutrient loading can stimulate growth of phytoplankton, epiphytes, and macroalgae, limiting the light that reaches the eelgrass. This process of eutrophication has been implicated in declines in eelgrass coverage in the northeast United States (Hauxwell et al., 2003; Novak et al., 2023). These losses have, in turn, had negative consequences for the organisms that rely on eelgrass, including bay scallops (Tettelbach et al., 2025). Nitrogen enrichment in coastal waters is commonly assessed using total nitrogen concentrations. Several studies prepared for the Town of Nantucket have reported decreases in average TN in Nantucket Harbor, which were interpreted as evidence of improving water quality (Samimy et al., 2025; Town of Nantucket Natural Resources Department, 2024, 2025). These studies, however, reported average TN levels, which can obscure the variation occurring in different sites within the harbor.

Biological indicators are more difficult to characterize. Average adult bay scallop density was 27% higher during 2011–2014 and 2019–2024 than during 2006–2010, although the later intervals followed the initiation of NRD-directed bay scallop restoration efforts (Heck, 2025). Average eelgrass cover, by contrast, declined from 37% in 2006 to 14% in 2024 (Heck, 2025). The years 2020–2024 were the five lowest years of eelgrass cover in the monitoring record, although the rate of decline may have slowed (Heck, 2025). Together, these records show no consistent pattern among changes in average TN, eelgrass cover, and bay scallop abundance.

Annual median TN declined at all four sites in my analysis, although the decline was statistically significant only at NAN2 and NAN3 (Fig. 3). However, the extensive NRD record revealed substantial variation within sites that was obscured in reports based on average TN. Individual TN measurements frequently exceeded the 25 μM threshold at all four sites, including the lower harbor site NAN1, likely to experience the greatest exchange with Nantucket Sound (Fig. 2). Thus, even at the site with the lowest overall TN, comparatively low averages did not mean that concentrations were consistently below the threshold.

Moreover, my analysis of variation in TN revealed a spatial gradient across the harbor. Above-threshold observations were more frequent at the upper and more enclosed sites, with TN lowest at NAN1, intermediate at NAN2 and NAN3, and highest at NAN5 in Polpis Harbor (Fig. 4). This gradient is consistent with greater exchange near the harbor entrance and greater water retention farther into the harbor, although I did not measure circulation or residence time directly. Water clarity (Secchi depth) and chlorophyll-*a* concentration exhibited a similar spatial gradient, with lower water clarity and higher chlorophyll-*a* in the more enclosed parts of the harbor (Figs. 6, 8).

Heck (2025, Figs. 13–15) reported spatial differences in eelgrass coverage in the lower, central and upper harbor as well. Although the lower harbor generally had the highest eelgrass coverage, there was no consistent difference in eelgrass coverage between the central and upper harbor. Thus, while both eelgrass coverage and water quality measures vary spatially across the harbor, a causal relationship cannot be inferred.

This analysis of the NRD data established that water quality varied both spatially within the harbor and over time during 2010–2025. The buoy data added a high-frequency dataset that permitted analyses of the immediate conditions that harbor organisms experienced, including short-term variability and the duration and recurrence of potentially stressful conditions at one harbor site.

Water temperature repeatedly exceeded the 23 °C sublethal exposure threshold between June 23 and September 5, 2025 (Figs. 9, 10). Temperatures above this threshold may impair eelgrass carbon balance and physiological performance (Lee et al., 2007; Jabbari et al., 2024). In fact, warming has been associated with reduced eelgrass persistence in northeastern U.S. estuaries (Plaisted et al., 2022; Long and Mora, 2023). Thus, even the lower-harbor buoy site, near the area where TN was lowest, experienced recurrent temperatures potentially stressful to eelgrass.

The continuous record also revealed exposure to co-occurring stressors. Elevated temperature above 23 °C and DO below 6 mg/L occurred concurrently at the buoy site (Fig. 11). This co-occurrence is biologically important because warming increases metabolic oxygen demand while low DO limits oxygen availability. Experimental studies have shown that warming and hypoxia together can reduce bay scallop feeding, physiological performance, and survival (Tomasetti et al., 2023). Consequently, relatively favorable TN conditions at a site do not preclude biologically significant exposure to other stressors.

The buoy record and NRD measurements from nearby NAN1 provide complementary information on water quality in the harbor (Fig. 12). The NAN1 measurements included surface, middle, and bottom samples, and revealed no consistent vertical gradient in DO. Moreover, the fact that the NAN1 observations fell within the range recorded by the buoy provides some support for the consistency of the two datasets and for this comparison between nearby, but non-overlapping, sites. However, the discrete sampling at NAN1 did not capture the substantial variation in DO revealed by the continuous measurements from the buoy. Organisms experience the full sequence of changing conditions, including intensity, duration and recurrence of low-oxygen exposure.

Exposure to short-term stressful conditions in marine environments is likely to increase (Burger et al., 2022; Wong et al., 2024). Episodes of elevated sea temperatures, in particular, have been implicated in losses of marine biodiversity and the ecosystem services it supports (Smith et al., 2023). In fact, elevated temperatures can exacerbate the effects of other stressors such as hypoxia and acidification on the physiological performance and survival of marine organisms (Vaquer-Sunyer and Duarte, 2011; Alter et al., 2024). Although warming cannot be mitigated through local efforts alone, nutrient enrichment is locally tractable. By reducing water clarity and promoting algal production and oxygen depletion, nutrient enrichment can create additional stresses that interact with elevated temperature (Burkholder et al., 2007; Lefcheck et al., 2017). Reducing nutrient inputs will not necessarily reverse eelgrass decline, but it may alleviate one component of cumulative environmental stress and improve the likelihood of eelgrass persistence under continued warming.

Because discrete sampling is not constrained to a permanently installed instrument, the NRD program has measured water quality across a large area of Nantucket Harbor. The analysis of those data in this study revealed spatial variation in water quality that is consistent with differences in enclosure and distance from the harbor entrance. The more enclosed sites farther from the harbor entrance exhibited lower water clarity, higher chlorophyll-*a* concentrations, and higher TN. Moreover, analysis of individual measurements revealed frequent exceedances of the 25 μM threshold, even at the site closest to the harbor entrance.

In contrast, the MMA buoy is moored at one location, but its continuous recording revealed repeated periods of elevated temperature and low DO, sometimes occurring together. Nutrient enrichment, warming, and low oxygen are established eelgrass stressors, and Heck (2025) reported mean eelgrass cover at repeatedly surveyed sites substantially below its 2006 level. Thus, the combination of these two monitoring programs identified different but potentially interacting pressures on eelgrass habitat. Effective assessment and management are therefore likely to benefit from combining long-term, spatially distributed monitoring with high-frequency observations to characterize both spatial heterogeneity and the timing, duration, and co-occurrence of potentially stressful conditions.

Declaration of generative AI and AI-assisted technologies in the manuscript preparation process

The author used generative AI (ChatGPT by OpenAI) for language editing to improve clarity and readability. The author reviewed and revised all AI-assisted content and takes full responsibility for the content of the publication.

Declaration of competing interest

The author declares no conflicts of interest.

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Data availability

The analysis-ready datasets underlying this study are available in Zenodo at https://doi.org/10.5281/zenodo.22004226.

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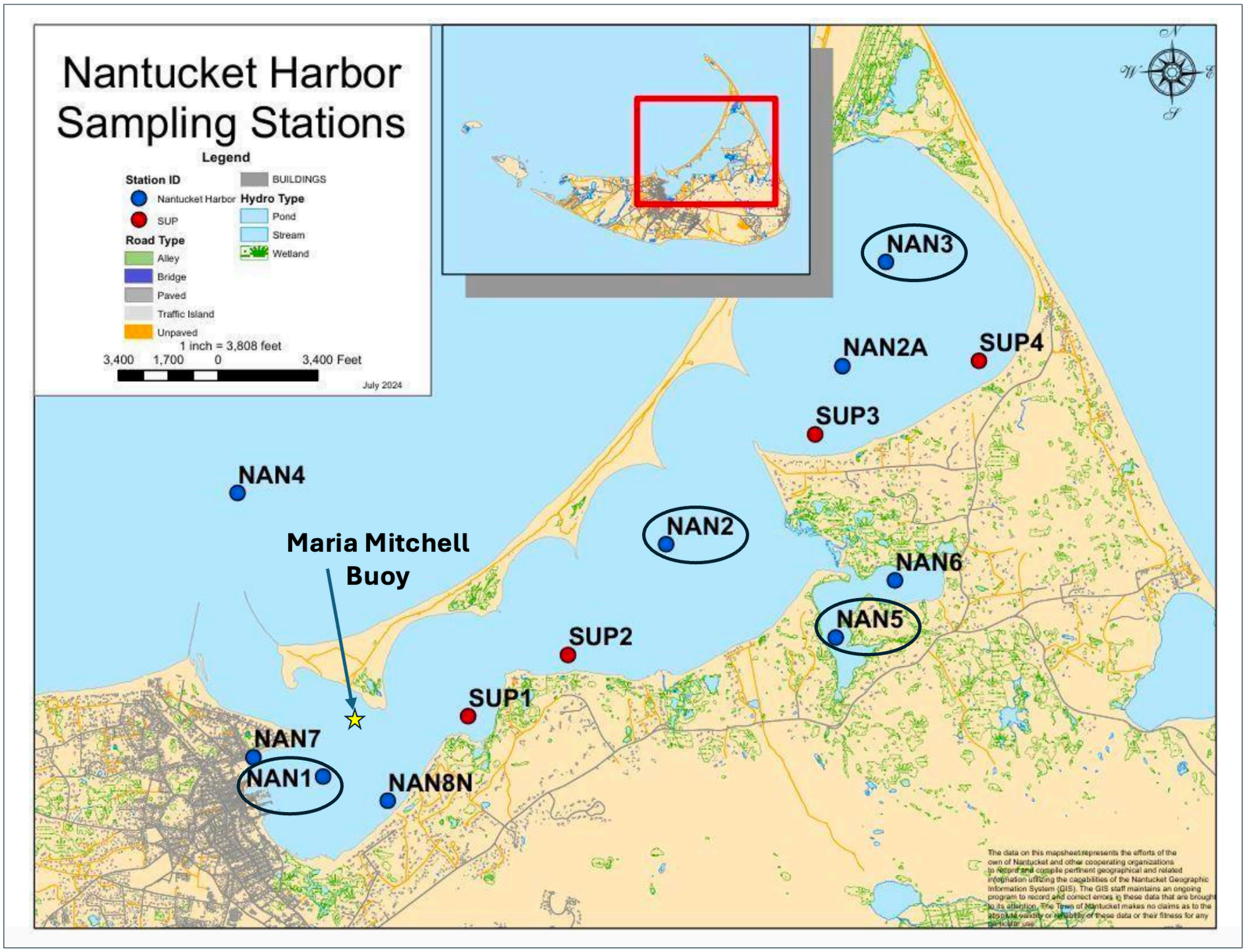

Figure 1. Nantucket Harbor monitoring locations. Base map from the 2024 Nantucket Island-Wide Estuaries and Salt Ponds monitoring report (Samimy et al., 2025), annotated by the author. Circled stations indicate the sites used in the spatial analyses: NAN1 in the lower harbor, NAN2 in the mid-harbor, NAN3 in the upper harbor, and NAN5 in Polpis Harbor. The star indicates the MMA buoy location.

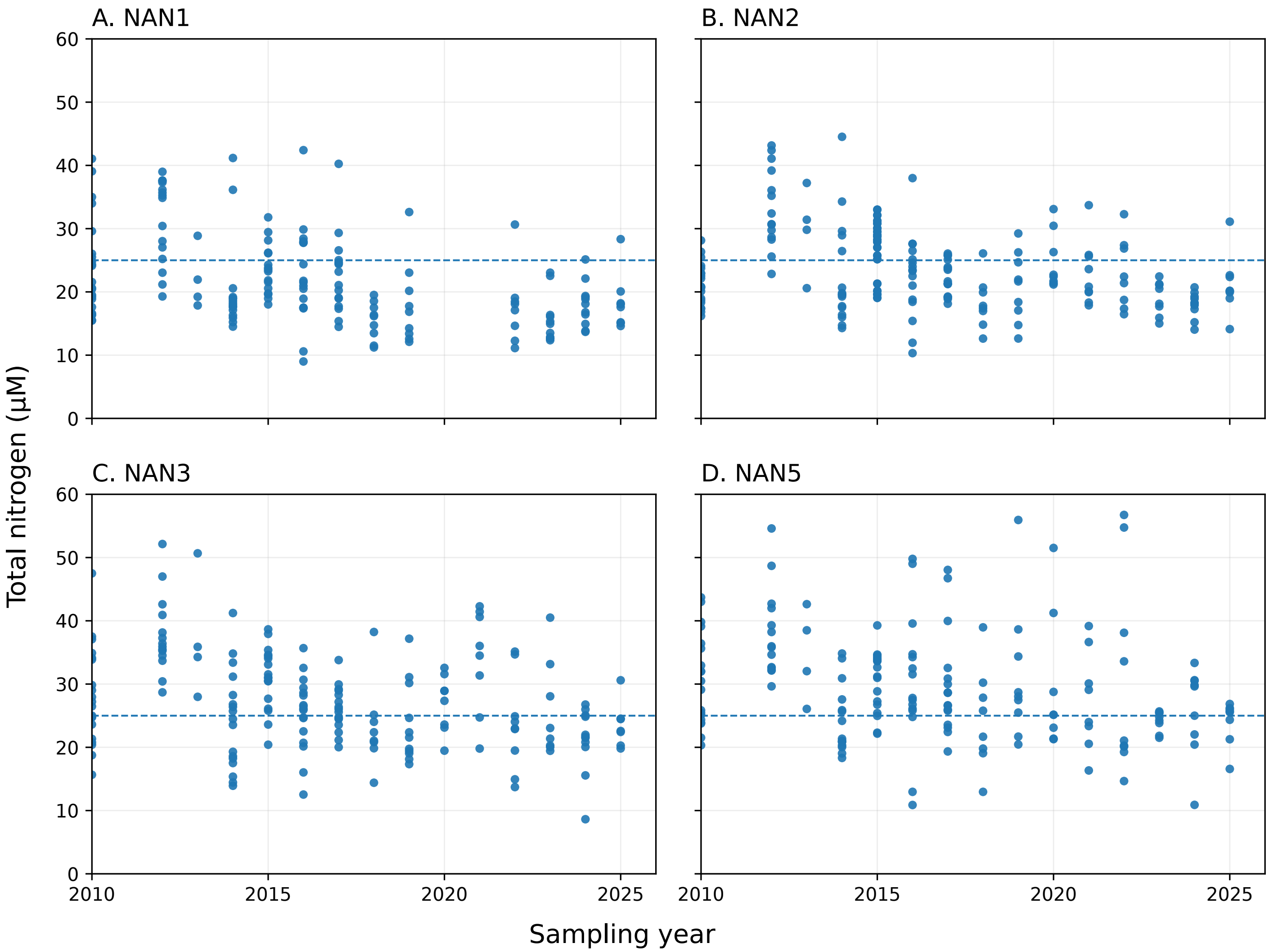

Figure 2. Total nitrogen (TN) concentrations at four Nantucket Harbor monitoring sites, 2010–2025. Panels show individual TN observations for A, NAN1; B, NAN2; C, NAN3; and D, NAN5. Values >60 µM were excluded as likely anomalous. The dashed horizontal line indicates the common 25 µM (0.35 mg N/L) threshold used here for comparison across sites, based on the Massachusetts Estuaries Project target concentration for Nantucket Harbor and its subsequent use in Town monitoring reports.

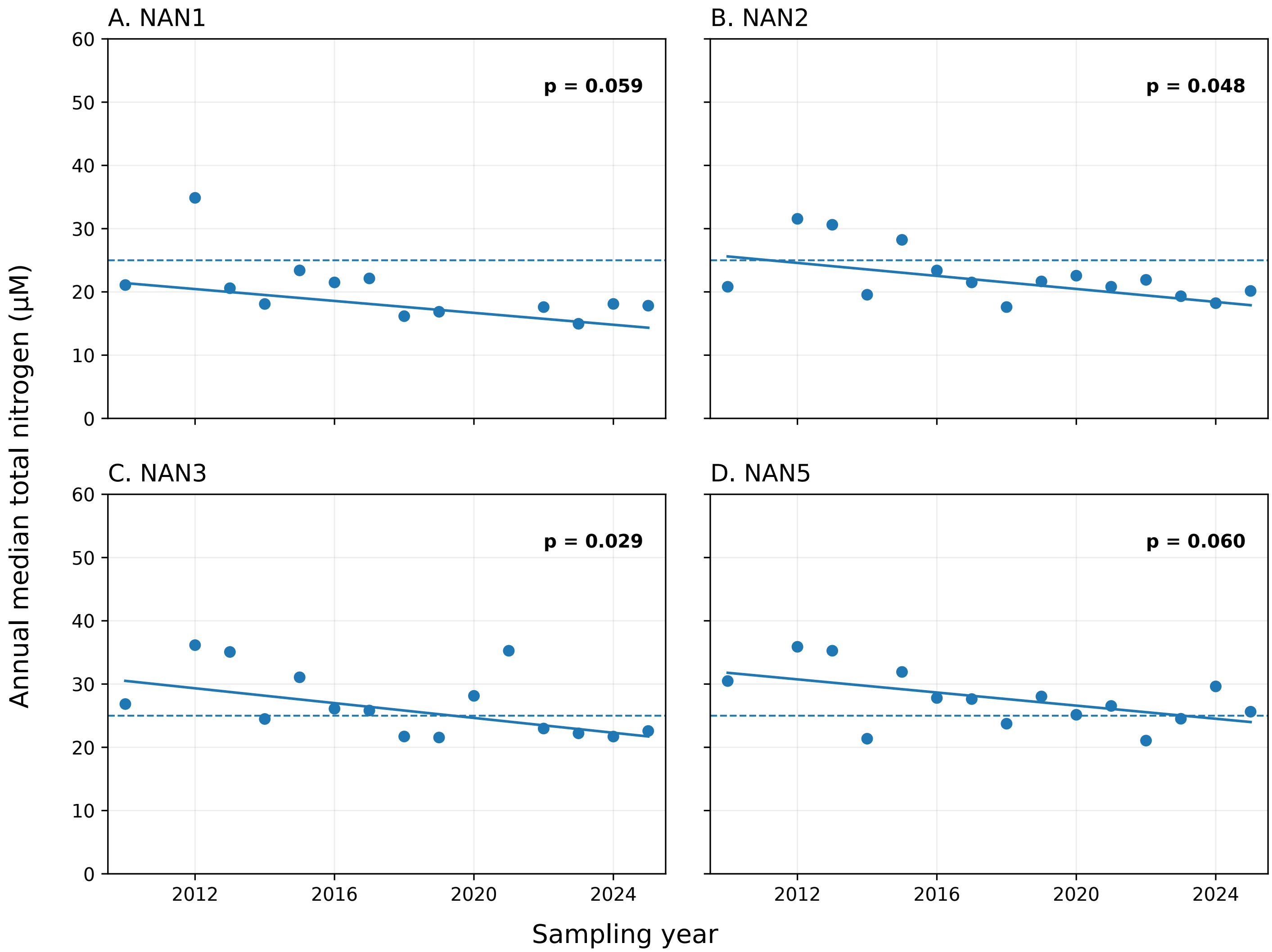

Figure 3. Annual median total nitrogen (TN) concentrations at four Nantucket Harbor monitoring sites, 2010–2025. Panels show annual median TN values for A, NAN1; B, NAN2; C, NAN3; and D, NAN5. Solid lines show Sen's slope trend estimates; p-values are from Mann-Kendall tests. The dashed horizontal line indicates the common 25 µM (0.35 mg N/L) threshold used here for comparison across sites, based on the Massachusetts Estuaries Project target concentration for Nantucket Harbor and its subsequent use in Town monitoring reports.

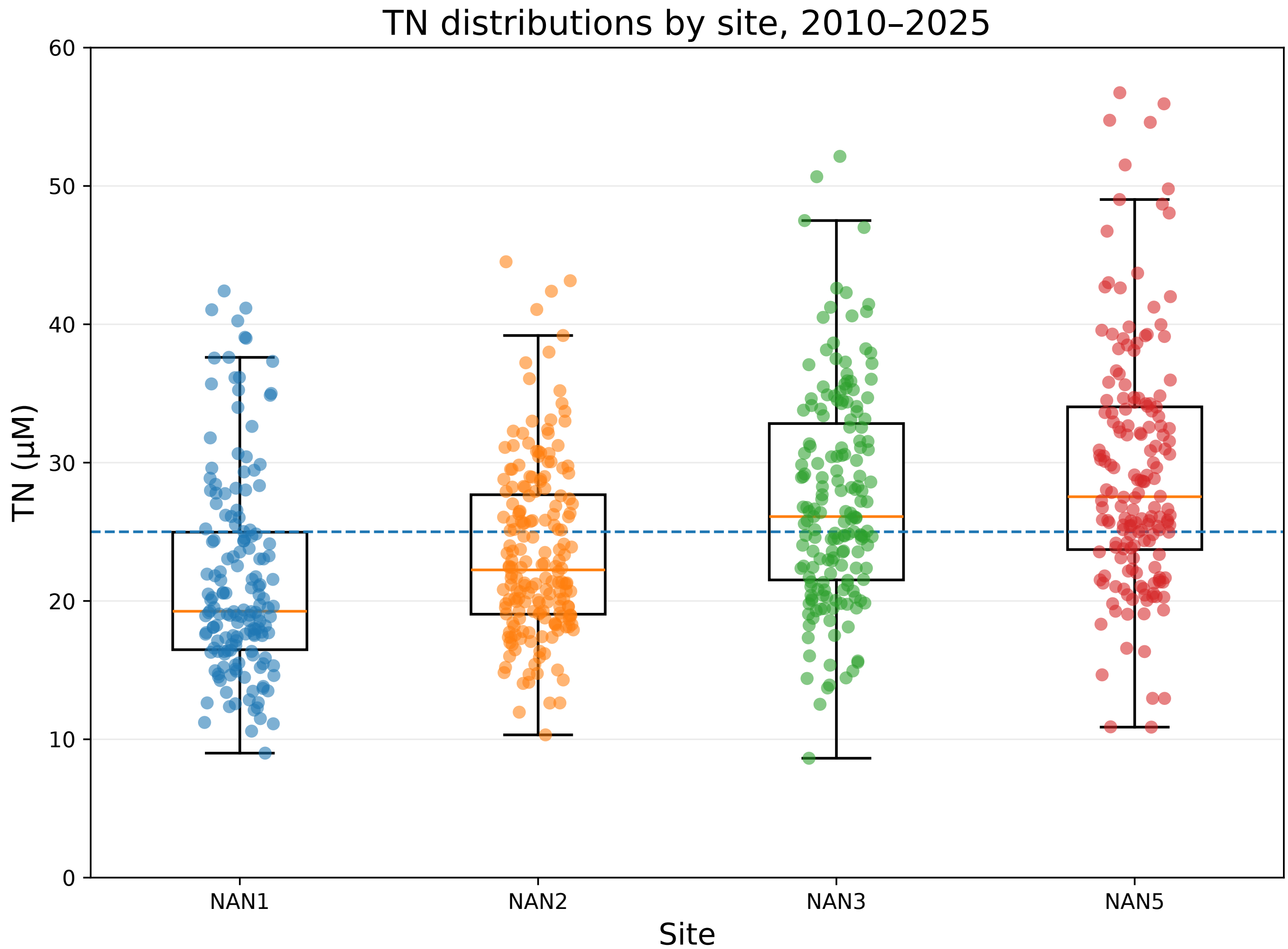

Figure 4. Total nitrogen (TN) distributions among Nantucket Harbor monitoring sites, 2010–2025. Points show individual TN observations, and boxplots summarize the distribution at each site. The horizontal line within each box indicates the median, boxes show the interquartile range, and whiskers extend to the most extreme observations within 1.5 times the interquartile range. The dashed horizontal line indicates the common 25 µM (0.35 mg N/L) threshold used here for comparison across sites, based on the Massachusetts Estuaries Project target concentration for Nantucket Harbor and its subsequent use in Town monitoring reports.

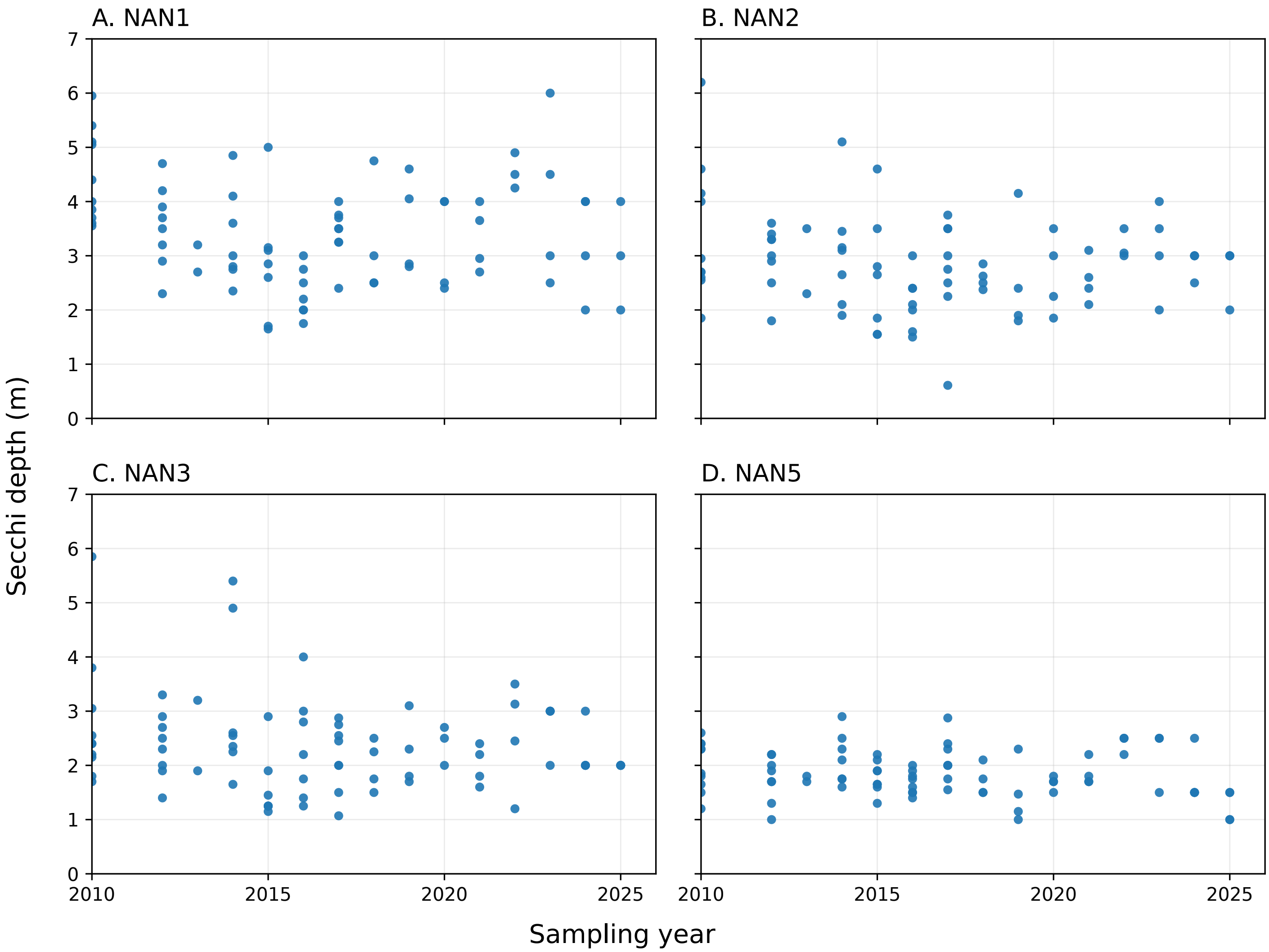

Figure 5. Secchi depth at four Nantucket Harbor monitoring sites, 2010–2025. Panels show one Secchi depth observation per site and sampling date for A, NAN1; B, NAN2; C, NAN3; and D, NAN5. Secchi depth is used as a proxy for water clarity, with greater values indicating clearer water.

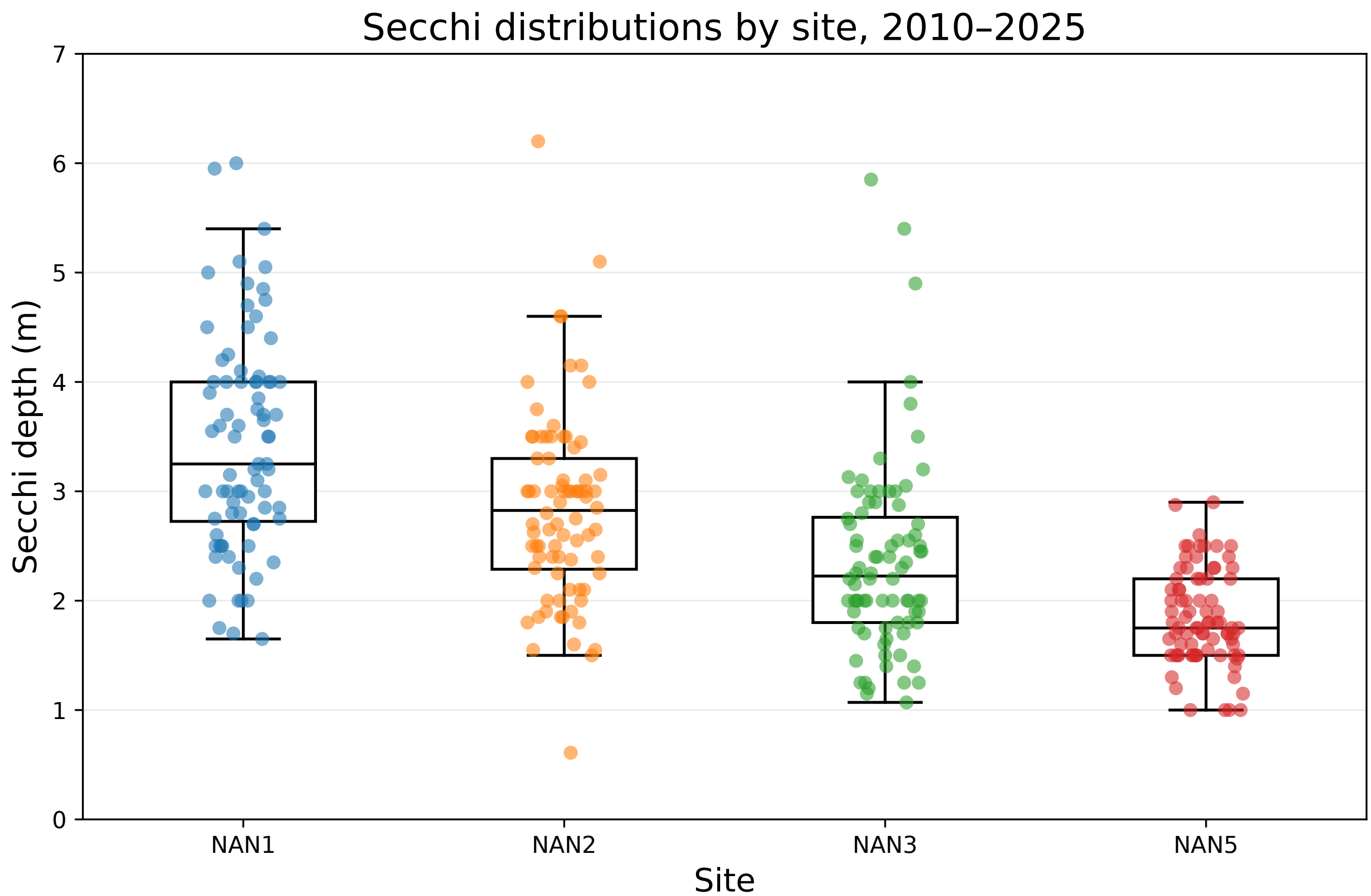

Figure 6. Secchi depth distributions among Nantucket Harbor monitoring sites, 2010–2025. Points show one Secchi depth observation per site and sampling date, and boxplots summarize the distribution at each site. The horizontal line within each box indicates the median, boxes show the interquartile range, and whiskers extend to the most extreme observations within 1.5 times the interquartile range.

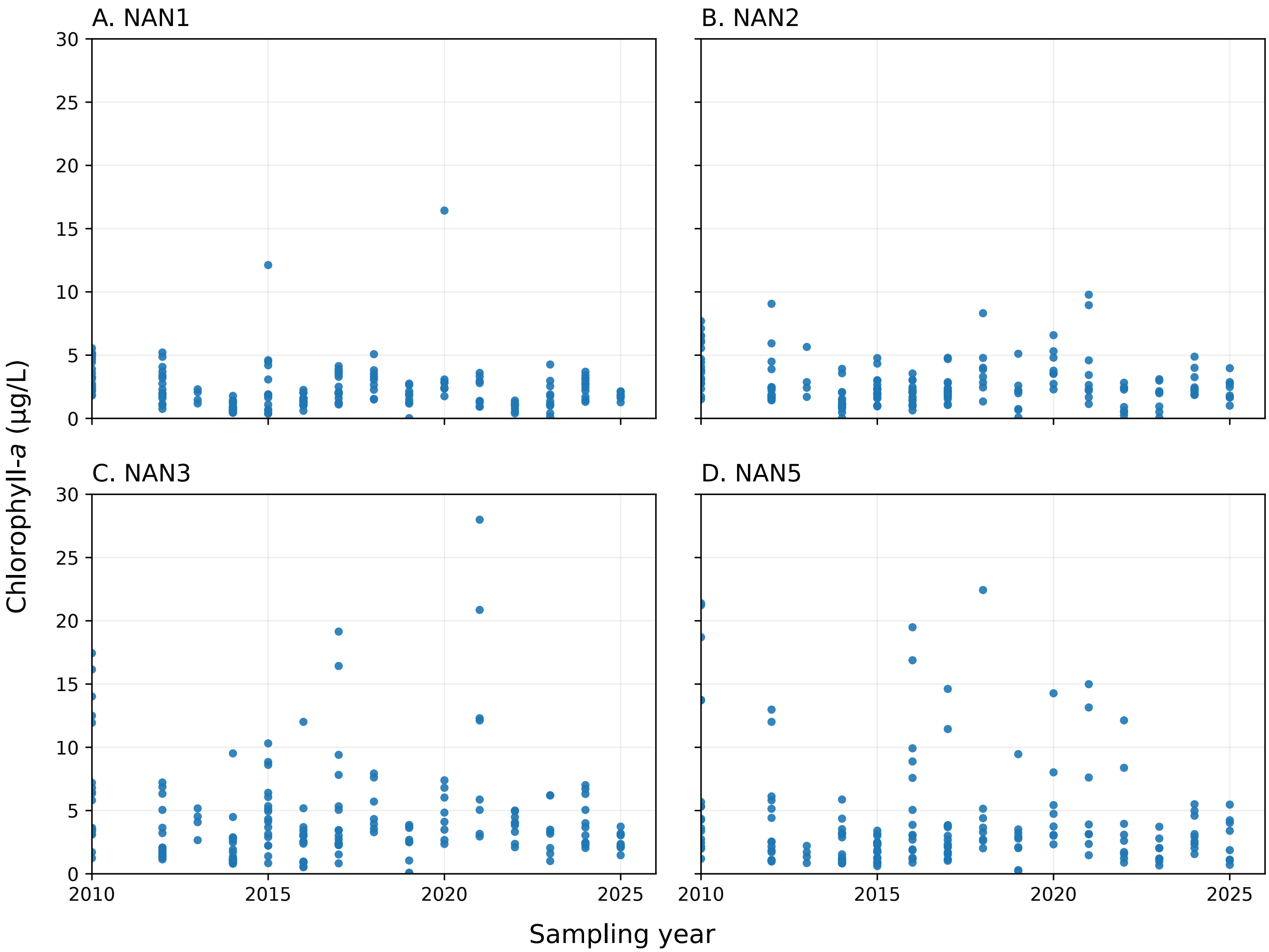

Figure 7. Chlorophyll-*a* concentrations at four Nantucket Harbor monitoring sites, 2010–2025. Panels show individual chlorophyll-*a* observations for A, NAN1; B, NAN2; C, NAN3; and D, NAN5. Surface, middle, and bottom samples were pooled because no significant depth-related effect was detected. Negative values were excluded before analysis. Four values >30 µg/L were retained in the statistical analyses but omitted from display.

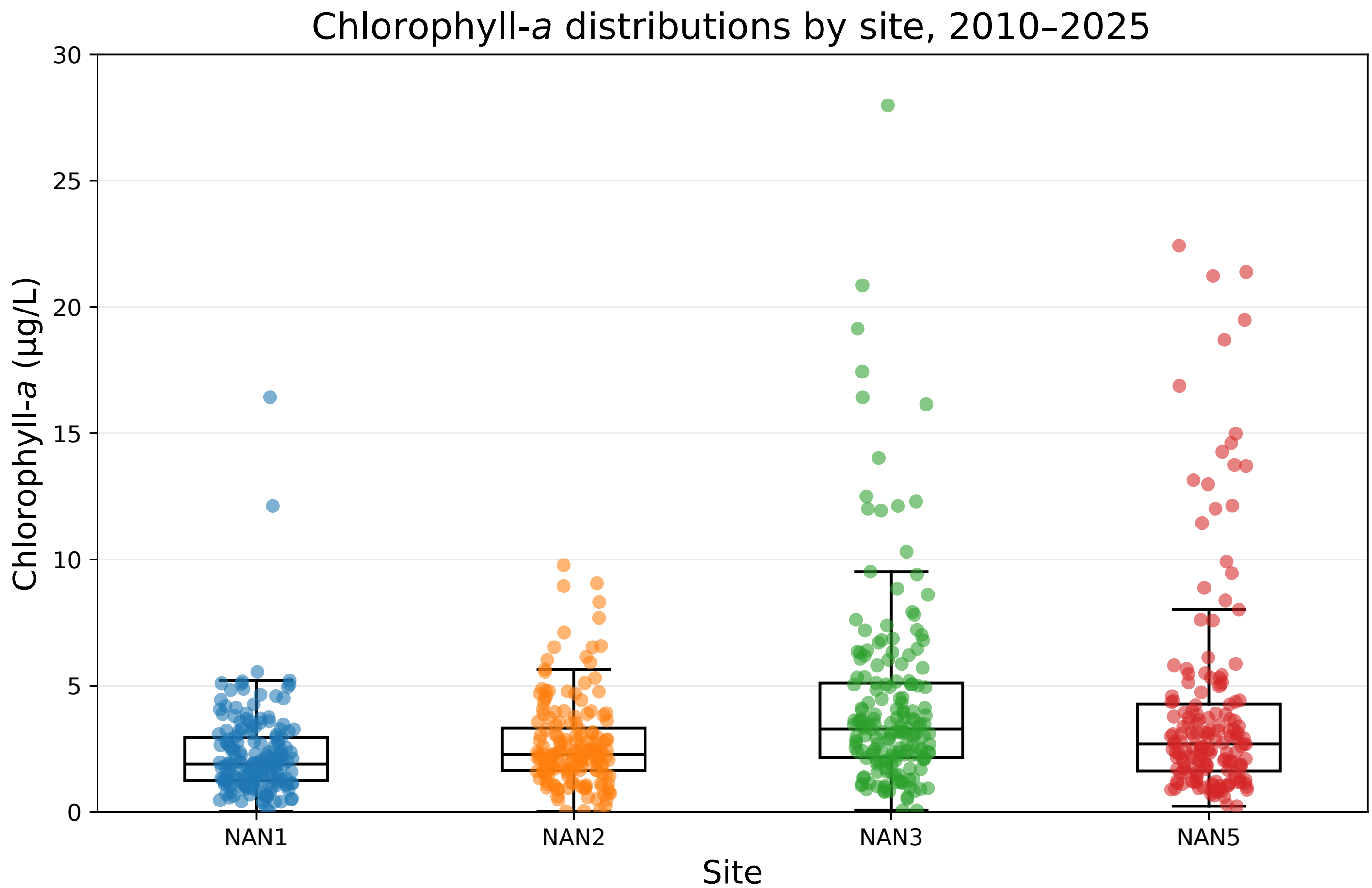

Figure 8. Chlorophyll-*a* distributions among Nantucket Harbor monitoring sites, 2010–2025. Points show individual chlorophyll-*a* observations, and boxplots summarize the distribution at each site. The horizontal line within each box indicates the median, boxes show the interquartile range, and whiskers extend to the most extreme observations within 1.5 times the interquartile range. Four values >30 µg/L were retained in the statistical analyses but omitted from display.

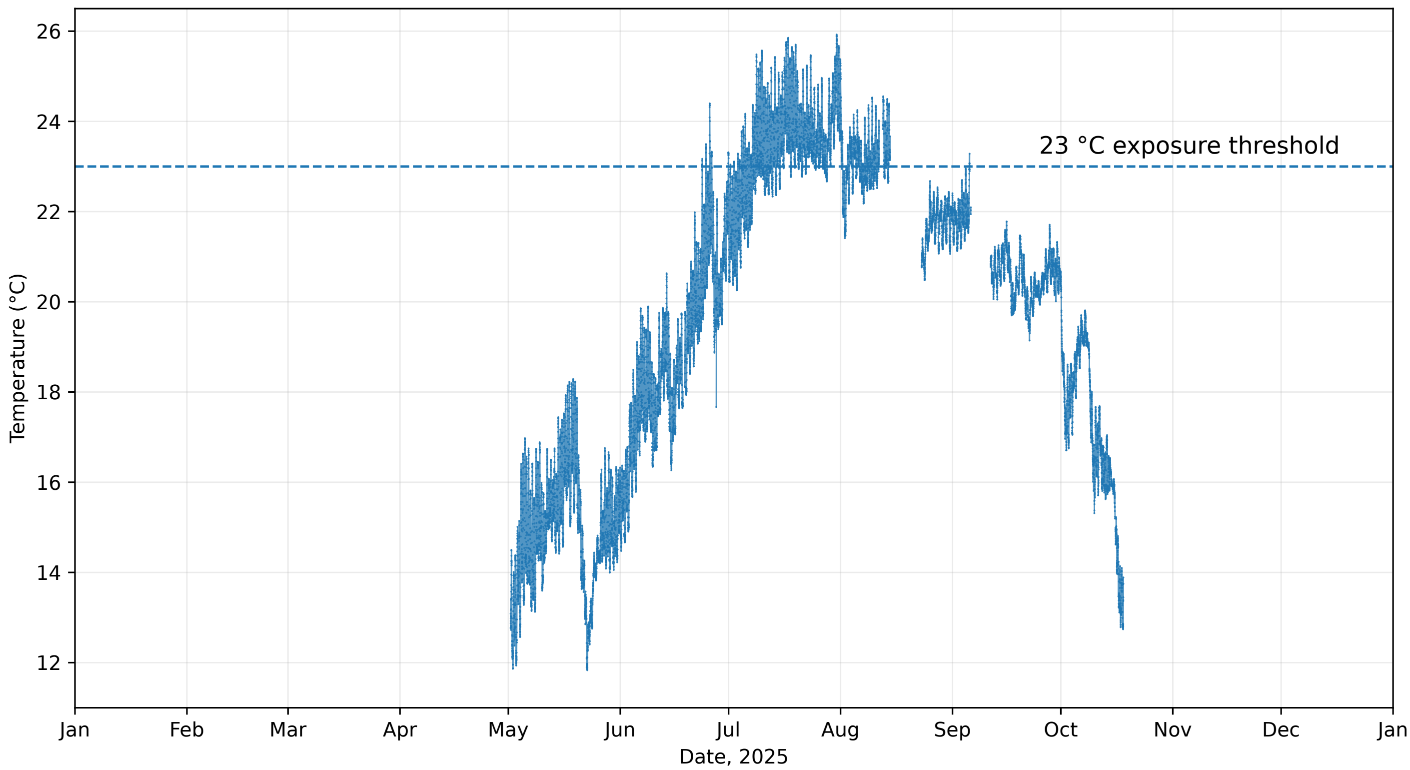

Figure 9. Continuous MMA buoy temperature record, 2025. Water temperature was recorded at 15-minute intervals during the 2025 buoy deployment. The dashed horizontal line indicates the 23 °C sublethal warm-water exposure threshold. Gaps represent intervals without usable measurements; missing values were not interpolated.

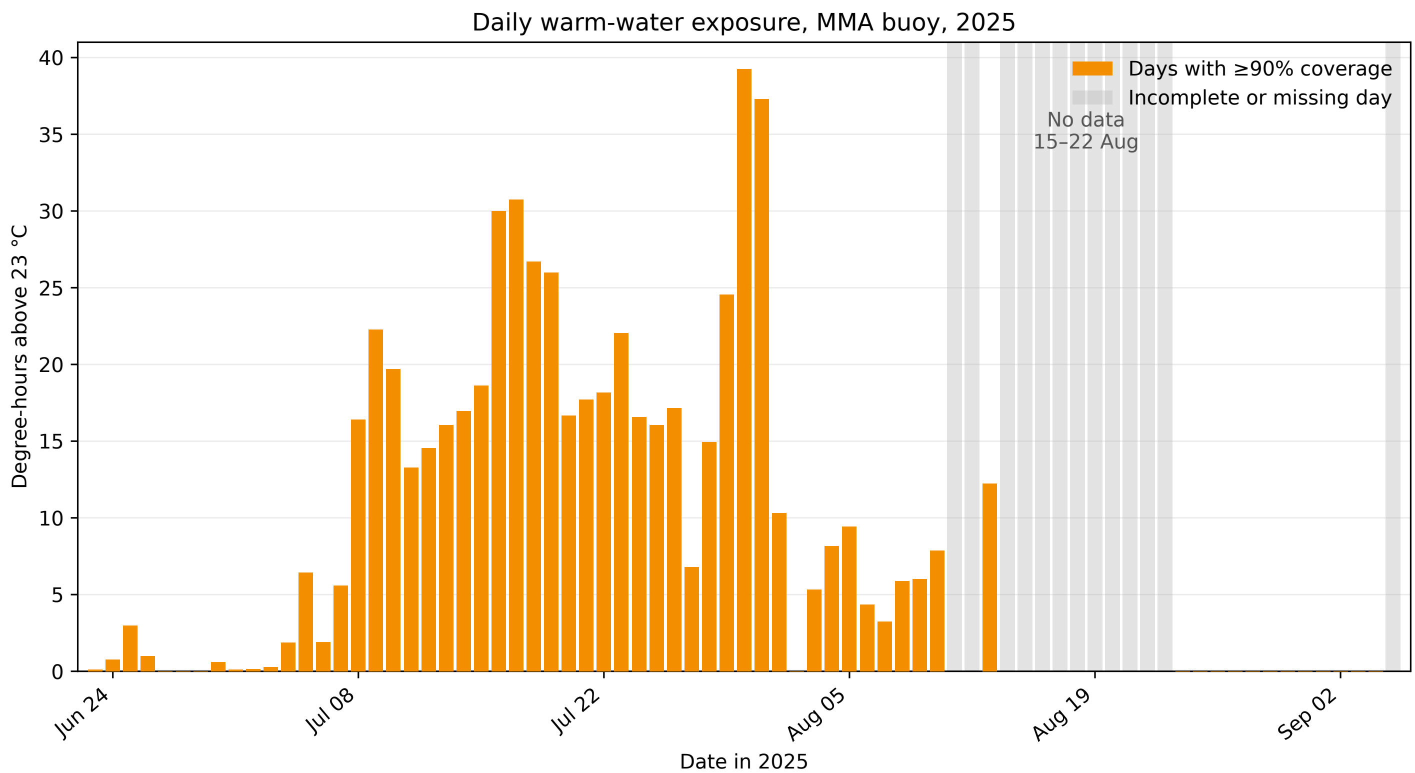

Figure 10. Daily warm-water exposure at the MMA buoy site, 2025. Bars show daily degree-hours above 23 °C, calculated by summing the temperature exceedance above 23 °C across 15-minute intervals, with each interval counted as 0.25 hour. Daily totals are shown only for days with at least 87 of the 96 expected measurements, equivalent to at least 90% coverage. Gray bands indicate incomplete or missing days.

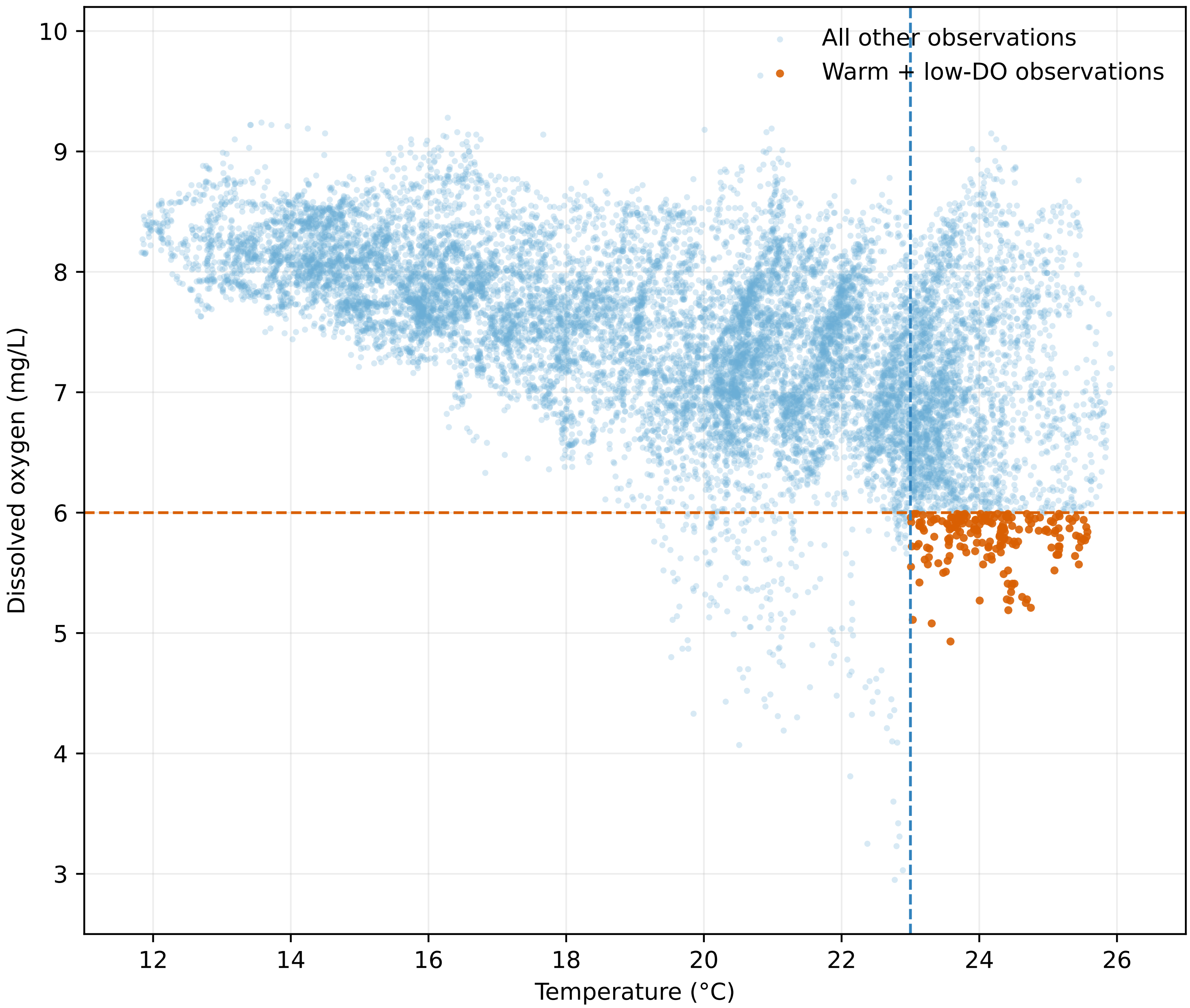

Figure 11. Co-occurrence of warm-water and low-oxygen exposure at the MMA buoy site, 2025. Each point represents a valid paired 15-minute observation of water temperature and dissolved oxygen. Dashed lines indicate the 23 °C sublethal warm-water exposure threshold and the 6 mg/L low-oxygen exposure threshold. Orange points identify observations during which temperature exceeded 23 °C and dissolved oxygen was below 6 mg/L.

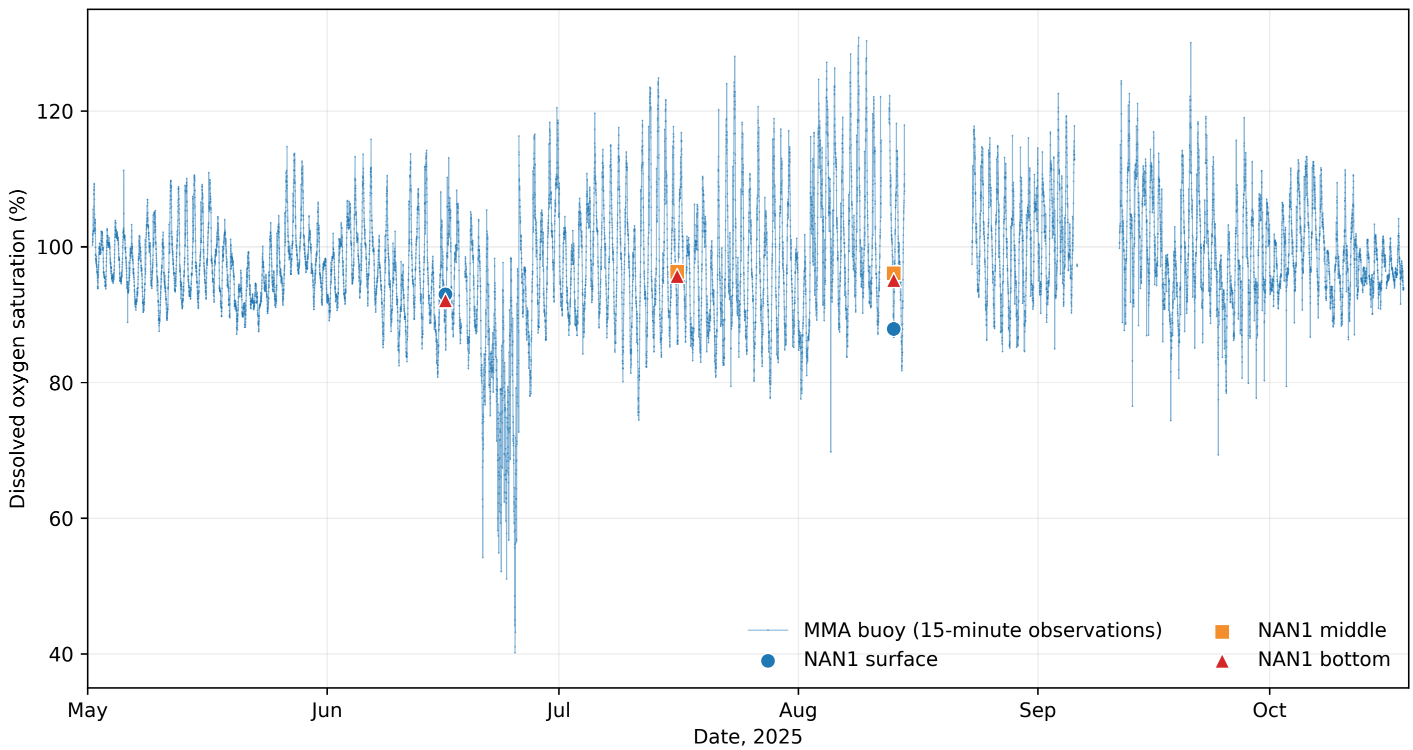

Figure 12. Continuous dissolved oxygen saturation at the MMA buoy compared with NRD grab samples from NAN1 during 2025. The blue line shows dissolved oxygen saturation measured at 15-minute intervals by the MMA buoy; lines are not drawn across missing intervals. Symbols show eight distinct surface, middle, and bottom measurements collected at NAN1 on the corresponding sampling dates. Two exact duplicate NRD export rows were removed before plotting. The MMA buoy and NAN1 are nearby lower-harbor sites in somewhat different locations (Fig. 1).