

FINAL REPORT

for

***Long-term quantification of nitrogen bioextraction and carbon capture by
seaweed and bivalve aquaculture in Long Island Sound***

Suffolk County, NY

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EXECUTIVE SUMMARY

This study investigated the potential of bivalve (Eastern oyster) and macroalgal (sugar kelp and *Ulva*) aquaculture as a bioextraction strategy to mitigate excessive nitrogen (N) and carbon (C) loads across a eutrophication gradient in Long Island Sound (LIS), from poorest water quality in the western-most site of the East River to increasingly improved water quality moving further east from Oyster Bay, Northport Harbor, and Mount Sinai Harbor. Data collected throughout the study confirmed that while N-loading has decreased from sewage treatment plants (STPs), symptoms persist due to slow-remediating non-point sources like septic systems. The primary findings established that the effectiveness of macroalgal bioextraction is highly site-specific and species-specific. Sugar kelp (*Saccharina latissima*) dominated N removal in the highly energetic and nutrient-impacted East River, reaching an estimated 250 kg N removed per hectare per year, demonstrating its superior capacity in high-flow, high-nutrient conditions. In contrast, *Ulva* spp. extracted more nitrogen than kelp in enclosed, shallow harbors like Oyster Bay and Mount Sinai Harbor, suggesting it is better adapted to the higher temperatures and fluctuating nutrient pulses of those environments, thus necessitating a species selection model based on site hydrodynamics. Oysters grew better in Mount Sinai and Oyster Bay compared to the East River, potentially due to poorer water quality, including acidification/low pH, at the latter site. It is estimated that oyster farming could yield annual nitrogen bioextraction rates of 135 kg, 185 kg, and 260 kg per hectare at the East River, Oyster Bay, and Mt. Sinai sites, respectively. Combining the bioextraction services of all three species, it is estimated that a one-hectare sugar kelp-*Ulva*-oyster farm would be capable of removing 390 kg, 242 kg, and 306 kg of nitrogen per year in the East River, Oyster Bay, and Mt. Sinai Harbor, respectively. This total bioextraction in the East River exceeded prior estimates made by the Nassau County Nine Element Plan for seaweed-oyster aquaculture (380 kg per hectare per year), while rates for Mount Sinai Harbor and Oyster Bay fell short of that mark. These rates could have removed 3% of the total nitrogen load in Oyster Bay and 6% for Northport Bay (located in close proximity to Northport Harbor within the same embayment), which are both sites in Nassau and Suffolk County that were identified by watershed management plans to require nitrogen load mitigation. Given that there are opportunities to optimize nitrogen bioextraction, especially by *Ulva*, and given that more than 1% of surface waters could be used for aquaculture in some regions, bioextraction by bivalves and seaweeds represents a promising nitrogen

mitigation approach.

Beyond simple N removal, the study found improvements in localized water quality within seaweed deployments, building off previous studies that collectively indicate seaweeds act as a crucial buffer against two major LIS impairments: acidification and acute hypoxia. Continuous monitoring showed that pH levels were consistently higher within kelp and *Ulva* deployments compared to control sites (up to 0.3 units difference in Northport), indicating a localized buffering effect against ocean acidification. Simultaneously, seaweed presence was linked to the prevention of acute hypoxia, as control sites frequently measured below the acute dissolved oxygen (DO) minimum of 3 mg/L DO, while seaweed sites did not. These environmental benefits are critical for calcifying organisms, as oyster growth was inversely correlated with seaweed productivity, growing fastest in Mount Sinai Harbor and slowest in the East River, supporting the hypothesis that acidification stress limits oyster growth in areas where kelp thrives; however, oysters co-grown with *Ulva* in Oyster Bay achieved higher final weights than controls, indicating that the localized water quality-modifying effects of co-culture can successfully offset the negative impacts of the surrounding estuary. Ultimately, calculated yields demonstrate that bioextraction must complement, not replace, watershed remediation efforts, with Oyster Bay offsetting 3% of 60% N reduction needed for Oyster Bay and Northport yields potentially offsetting 6% of the required 37% N reduction. Still, the dual capacity of aquacultured macroalgae to remove N and provide localized pH and DO buffering provides a vital, validated tool for protecting bivalve aquaculture and the coastal ecosystem against the complex threats of eutrophication and acidification, complementing land-based management strategies during the multi-decade period required to achieve full watershed restoration, as projected by watershed management plans.

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INTRODUCTION

Excessive nitrogen (N) loading has impaired Long Island Sound (LIS) for decades. Phytoplankton communities in LIS are strongly N-limited (Gobler et al., 2006), and N-enriched effluent from dozens of sewage treatment plants (STPs) promote the overgrowth of phytoplankton and subsequent hypoxia in western LIS (Parker & O'Reilly, 1991; O'Shea & Brosnan, 2000; Anderson & Taylor, 2001). A total maximum daily load (TMDL) plan for LIS called for a 58% reduction in N-loading which was achieved largely via the upgrading of STPs (LISS CCMP, 2015). Consequently, since the beginning of this century, the size, intensity, and duration of summer hypoxic zones in LIS have all decreased (CTDEEP, 2003-2020). Despite this success, however, the symptoms of eutrophication across LIS are still evident, especially in many of the enclosed, shallow bays and harbors along New York (NY) and Connecticut (CT) coastlines. During the past 15 years, multiple species of new harmful algal blooms (HABs) have emerged in LIS coastal waters (Hattenrath et al., 2010; Hattenrath-Lehmann et al., 2013; Hattenrath-Lehmann et al., 2015A), which have been shown to be promoted by excessive N loading (Hattenrath et al., 2010; Hattenrath-Lehmann et al., 2015). In addition, hypoxia and anoxia occur for extended periods annually in more than a half dozen nearshore locations not covered by regular CTDEEP/LIS Partnership monitoring (LIMMN, 2014-2020). While upgrading STPs resulted in the relatively rapid reversal of water quality impairment in the main stem of western LIS, recent studies have concluded that on-site septic systems are the largest source of N for the entire north shore of Suffolk County (SCSWP, 2020) and Nassau County (NCSWP, 2020) on Long Island, representing the entire southern coastline of LIS. In CT, along the northern coastline of LIS, non-point sources contribute to the majority of N loading via fertilizers and onsite septic systems to many subwatersheds (Latimer & Charpentier, 2010). While significant efforts are underway to upgrade the more than 500,000 septic systems across Long Island, the process is slow, and even after most systems are upgraded, the decadal travel time of groundwater along the north shore of Long Island will delay the realization of improved water quality (SCSWP, 2020). As such, interim measures to mitigate eutrophication in coastal waters are required to complement on-going watershed remediation of excessive N loads.

One approach for mitigating N loads after they have entered coastal waters is bioextraction. For decades, scientists have cited the possibility of using filter feeding bivalves to combat eutrophication (Officer et al., 1982; Newell, 2004; Galimany et al., 2017). Indeed, the process of

filter-feeding can mitigate algal blooms before they arise (Officer et al., 1982; Cerrato et al., 2004; Rose et al., 2015). Resultant improvements in water clarity can have additional ecosystem benefits to submerged aquatic vegetation, such as eelgrass beds (Wall et al., 2008). While the ability of wild stocks of bivalves to combat eutrophication has met with some skepticism (Pomeroy et al., 2006), aquaculture offers the possibility of purposely stocking dense arrays of bivalves in targeted regions and holds the promise of significant N removal and improved water quality in targeted regions (Sebastiano et al., 2015). Still, while bivalves can clear phytoplankton from the water column via filter-feeding, only a fraction of consumed N is incorporated into bivalve tissue and shell (Petersen et al., 2019). The remaining N is deposited into sediments where a small fraction may be denitrified (Newell, 2004; Petersen et al., 2019) but a large fraction can be reintroduced into the water column via diffusive sediment fluxes that can promote additional primary production (Gobler, 2020).

In contrast to bivalves, almost all the N extracted by seaweeds is incorporated into macroalgal tissues and, when grown and harvested via aquaculture, that N can be permanently removed from the watershed (Rose et al., 2015). The aquaculture of seaweed is a nascent industry in LIS. While CT has permitted commercial seaweed farming for over 10 years and currently has about ten permitted kelp farms in LIS waters, commercial seaweed aquaculture in NY is more recent, with only three permitted kelp farms in the South Shore Estuary and none in LIS. Aside from sugar kelp, no other seaweed is being commercially farmed currently in either state. More broadly, seaweed farming is a nascent industry across the entire United States, with the commercial cultivation of sugar kelp (*Saccharina latissima*) initiated a little over a decade ago in waters off the coast of Maine (Flavin et al. 2013; Kim et al. 2019). Today, seaweed aquaculture is one of the fastest growing maritime industries in the coastal waters of New England, extending from Maine to New York, and is also a growing industry on the west coast, particularly in Alaska (Kim et al., 2017; Kim et al 2019). Across these regions sugar kelp is the dominant species being farmed, but other species also include skinny kelp (*Saccharina angustissima*), bull kelp (*Nereocystis luetkeana*), ribbon kelp (*Alaria esculenta*), Pacific dulse (*Devaleraea mollis*).

The potential for seaweed bioextraction in LIS is clear. Kim et al. (2015) demonstrated that sugar kelp could be grown to yield between 5 and 9 kg fresh weight (FW) per meter of line per year. GreenWave's Thimble Island Farm in LIS has produced up to 100,000 lbs of sugar kelp annually for the past several years. Since 2018, the Gobler Laboratory has grown kelp on ten

different oyster farms across LI, including two sites in LIS. The lab established NY's first kelp hatchery at the Stony Brook-Southampton Marine Lab and has pioneered new approaches for the aquaculture-style grow-out of kelp in shallow estuaries (1 – 3 m). Traditionally, sugar kelp aquaculture had been limited to deep water columns only (> 8 m). Using modified gear, we have achieved yields in shallow estuaries such as Moriches Bay and Great South Bay (14 kg FW per meter) that have exceeded maximum yields in deeper water regions (Kim et al., 2015; K. Barbery, GreenWave, pers. comm.). This has important implications for LIS, as its shallow harbors and bays along the coastline are more in need of nutrient mitigation than deeper, open water regions.

While the grow-out of sugar kelp in NY and CT holds great promise as a bioextraction approach, it is a cold-water macroalgae, and thus its ability to remove N is confined to winter and spring months when water quality is maximal in LIS. Warm-water seaweeds (*Ulva* spp.) have a growing season that is the opposite of kelp (spring to fall) and thus may be a complementary, sustainable approach for bioextraction during seasons when ocean acidification, hypoxia, and HABs are most likely to occur in LIS (Anderson & Taylor, 2001; Wallace et al., 2014; Hattenrath-Lehmann et al., 2015). Thus, this may present an opportunity to rotate the cultivation and harvest of both seaweed “crops” throughout the year for continuous N removal, with sugar kelp thriving in the winter to spring months and *Ulva* spp. thriving in the spring to fall months.

Beyond bioextraction, seaweeds can have multiple other positive effects in coastal zones. As photosynthetic organisms, seaweeds continually oxygenate surrounding waters and remove carbon dioxide as they grow. Consequently, seaweeds hold the promise to combat two of the most significant water quality impairments in LIS: hypoxia and ocean acidification. Research has demonstrated that while acidification and hypoxia inhibit the growth and survival of bivalves (Talmage & Gobler, 2010; Gobler et al., 2014; Stevens & Gobler, 2018), co-locating bivalves with seaweeds can significantly improve the growth of multiple bivalves exposed to high levels of CO₂ (Young & Gobler, 2018). Kelp and other seaweeds have been shown to combat the intensity of ocean acidification via the assimilation of *p*CO₂ (Frieder et al., 2012; Anthony et al., 2013; Wahl et al., 2018). Finally, there is evidence demonstrating that seaweeds grown via aquaculture, including kelp and *Ulva* spp., can inhibit the proliferation of HABs (Tang & Gobler, 2011; Yang et al., 2015; Sylvers & Gobler, in preparation). Given the threats of hypoxia, acidification, and HABs to bivalves (Griffith & Gobler, 2020), all the water quality-modifying effects of seaweeds could benefit bivalves via increased growth and survival. While there have been some studies of

N bioextraction by seaweeds in LIS, extremely little is known about the ability of seaweeds to benefit water quality and bivalves, and studies exploring how different seaweeds grown in different locations via different methods affect bioextraction, bivalves, and water quality have not been performed.

The objectives of this project were to: (1) Quantify the removal of nitrogen and carbon by seaweeds and bivalves across Long Island Sound; (2) Quantify changes in water quality within large deployments of seaweeds and bivalves at nearshore and open water regions of Long Island Sound; (3) Quantify changes in water quality within control sites within the same ecosystem to assess the localized nature of the effects; (4) Quantify fractions of nitrogen loads offset by actual and modeled deployments of seaweeds and bivalves across nearshore and open water regions of Long Island Sound.

METHODS

Bivalves (Eastern oyster, *Crassostrea virginica*) and seaweeds (sugar kelp, referred to simply as kelp hereforth, *Saccharina latissima*; and *Ulva* spp.) were grown in tandem and separately at four sites across a eutrophication gradient in LIS including, from west to east, (1) the East River located within the Bronx River Estuary (40.806352, -73.797822), (2) Oyster Bay within the Oyster Bay/Cold Spring Harbor Complex (40.874375, -73.482346), (3) Northport Harbor with Northport Bay (40.892042, -73.357350), and (4) Mount Sinai Harbor (40.963303, -73.031812) (Fig 1). These four sites not only represented a strong contrast in eutrophication but were also representative of many sites in LIS. Regarding the LIS harbor sites in NY waters, nine years of weekly water quality monitoring data (2014-2022) by the Gobler Lab had revealed that Mount Sinai Harbor, the easternmost site, was well-flushed, rarely hypoxic, and did not experience HABs, attributes it shared with several other central and eastern LIS harbors including Port Jefferson Harbor, Stony Brook Harbor, and the Mattituck Inlet (LIMMN, 2014-2020). In contrast, the Oyster Bay/Cold Spring Harbor Complex and Northport Harbor experienced annual HABs caused by multiple species (Hattenrath-Lehmann et al., 2015; Hattenrath-Lehmann et al., 2015A; Hattenrath-Lehmann et al., 2018) and extended periods of hypoxia, attributes it shared with the nearby Huntington Harbor and Hempstead Harbor (LIMMN, 2014-2020). Finally, the East River is located in New York City, the nation's largest metropolis, and has received effluent discharge from

dozens of sewage treatment plants over many decades and thus has very high levels of nutrients and decades of recorded hypoxia (IEC 1991-2023).

These four sites represent a water quality/eutrophication gradient, starting with the westernmost site at the East River as the most nitrogen-impaired and poorest water quality, and improving as sites are located further east, from Oyster Bay to Northport Harbor to Mount Sinai Harbor as the least nitrogen-impaired and comparably good water quality. As such, these sites offered an excellent opportunity to understand how different seaweeds grown via different approaches performed under differing water quality conditions that were prevalent across the gradient of LIS and how they, in turn, might have benefited bivalves and water quality. While aquaculture had been traditionally performed in open water regions, watershed management plans indicated that shallow, enclosed harbor regions that directly receive the largest nutrient loads (SCSWP, 2020) are most in need of water quality improvement. This project provided a sense of how aquaculture of seaweeds and bivalves contributed toward such improvement.

Seaweed-oyster Cultivation

Seaweeds and bivalves were co-cultivated over a two-year period, with cultivation methods adapted for each site following approaches refined during the past decade in the Gobler Lab. Each year of the study (2024 and 2025) was divided into two rotating growing seasons suited for each seaweed: kelp season during the colder months (December-May), and *Ulva* season during the warmer months (May-November). Kelp was cultivated for two growing seasons (2024 and 2025) at each of the four study sites. *Ulva* was cultivated at three of the study sites (East River, Oyster Bay, Mount Sinai Harbor), but was not cultivated in Northport Harbor due to potential user conflicts during the warmer months since the site was located off a public access beach. Oysters were cultivated year-round with both seaweeds at two of the study sites (East River and Mount Sinai Harbor, representing the westernmost and easternmost locations along the water quality gradient), and for one growing season in Oyster Bay with *Ulva*. The growing season for oysters encompasses the warmer months from April to November, and thus overlaps with the entire *Ulva* growing season and only the tail-end of the kelp growing season. Cultivation techniques for each species are described below.

Kelp cultivation

Kelp was cultivated along horizontal ropes (four to six lines per site) using one of three

general methods depending on water depth, including the (1) ‘suspended’ line method in deep open waters (East River), (2) ‘staked’ line method in shallow (i.e., < 4 ft MLW) nearshore waters (Northport Harbor, Mt. Sinai Harbor), and (3) ‘boat-slip’ method used in empty marina boat slips during winter (Oyster Bay).

In the deep waters (~20-30 ft MLW) at the East River site, kelp was cultivated along ‘suspended’ horizontal longlines following standard industry methods (Flavin et al 2013). The suspended longlines were anchored by moorings on either end and suspended ~4 ft below the surface using buoys (Appendix Fig 1). Suspended kelp lines were approximately 150 ft in length from mooring buoy to mooring buoy, which included 130 ft of ½” rope (i.e. the kelp line) and a 10-ft ‘pigtail’ at either end to which the kelp line was tied (Appendix Fig 1).

In shallow near-shore waters (<4 ft MLW) in Northport and Mt. Sinai Harbors, kelp was cultivated along ‘staked’ horizontal longlines. This staked-line method was developed in 2018 by Doall & Gobler, and has been used to produce high kelp crop yields in multiple locations across coastal areas of Long Island, NY. Kelp lines (~ 100 ft long each) were staked a fixed distance above the bottom using screw-down anchors on either end of each line (Appendix Fig 2). The four-foot-long screw anchors were screwed approximately three feet into the bottom so that kelp lines were elevated about 12-18 in. off the bottom. As opposed to suspended lines, staked lines remain in a fixed position above the bottom and do not rise and fall with the tides.

Finally, in Oyster Bay, kelp was cultivated in empty boat slips at the Theodore Roosevelt marina following dockside seaweed cultivation techniques developed and refined by the Gobler Lab since 2015. Empty, inactive marinas during winter provide a potentially unique opportunity for nutrient bioextraction as marinas are often located in near-shore nitrogen-impacted waters, and provide sheltered areas for cultivation without the need for a boat. Within the empty boat slips, horizontal kelp lines were suspended 2-3 ft below the surface between a floating dock and outer pilings, rising and falling with the tides.

At each cultivation site, kelp lines were seeded in December using spools of ‘seedstring’ produced in the Gobler Lab kelp hatchery, which has been producing kelp seed spools annually for research purposes since 2019 (Appendix Fig 3). Reproductive parental tissue was collected by SCUBA divers off Montauk Point in early November of each year. Following collection, reproductive kelp blades were immediately brought back to the lab where the sorus tissue was

excised, cleaned, and desiccated overnight in a refrigerator (2-3 °C). The following day, the prepared tissue was placed in 10 °C filtered seawater to induce spore release, and the spores were set onto spools of string in aquarium tanks at known setting densities. The spools of seedstring were grown in aquarium in f/2 nutrient enriched sterilized seawater under controlled environmental conditions (10 °C; 12:12 light-dark cycle), with weekly water changes performed until the juvenile sporophytes were large enough (~2 mm) to seed kelp lines at the field sites, which occurred in December each year, approximately 5 to 6 weeks after the initial spore release.

***Ulva* cultivation**

Ulva spp. was vegetatively propagated from wild specimens collected from Shinnecock Bay (2024) and Moriches Bay (2025). At three study sites (East River, Oyster Bay, Mount Sinai Harbor), the wild-collected *Ulva* was co-cultivated with oysters in commercial floating oyster grow-out bags that are commonly used by the Long Island oyster aquaculture industry (Appendix Figure 4). Each floating bag was composed of a rectangular oyster grow-out bag (36" x 18" x 3") made of semi-rigid polyethylene mesh, with plastic, air-filled cylindrical floats (3" diameter) attached to each long side with 4 plastic ties. Two longline clips were strapped on one side of each bag to attach the bags to horizontal ropes. Bags were opened and closed on one end using stainless steel clips. To control biofouling, the bags were flipped over once every one to two weeks to expose the underside to sun and air and kill biofouling organisms. This method of grow-out for *Ulva* had been effectively used by the Gobler lab in previous experiments (Sylvers et al. 2025) and was similar to the methods of Chemodanov et al. (2019).

In 2024, *Ulva* cultivation experiments were conducted in the East River and Mount Sinai Harbor from July to October. In 2025, *Ulva* cultivation experiments were expanded to three sites (East River, Mount Sinai Harbor, Oyster Bay), with *Ulva* cultivated for the entire growing season from May to November, opposite months to when kelp was cultivated during its respective growing season. Triplicate grow-out bags were inoculated with 50 grams of *Ulva* in 2024, and 20 grams of *Ulva* in 2025. The *Ulva* was allowed to grow for up to four weeks, at which point it was harvested and bags were re-inoculated with the same starting biomass. *Ulva* was thus cultivated continuously through its growing season, but unlike kelp which was harvested once at the end of its growing season, *Ulva* was harvested approximately once every four weeks through its growing season. The 4-week harvest period was determined from previous experiments conducted by the

Gobler that showed that after four weeks, *Ulva* biomass begins to crash (Appendix Fig 5).

Oyster cultivation

Oysters were grown within the same type of commercial floating grow-out bags used to cultivate *Ulva* in two treatments: with *Ulva* (experimental) and without *Ulva* (control). In the experimental treatment, *Ulva* and oysters were co-cultivated in the same grow-out bags. In the control treatment, oysters were grown within floating bags along separate trawl lines positioned 3 to 40 m away from the *Ulva* treatment depending on site, following the research of Wallace and Gobler (2015) that demonstrated that the strong influence of dense seaweed deployments in well flushed regions could be limited to this extent. For each of the three sites (East River, Oyster Bay, Mt. Sinai Harbor), triplicate cages of seed oysters were deployed in control and experimental treatments. Each replicate cage was stocked with one hundred juvenile oysters spawned by the East Hampton Town hatchery in 2024.

In East River and Mt. Sinai, the same cohort of oysters were grown over a ~15-month period (July 2024–November 2025) that encompassed two growing seasons (2024, 2025). In Oyster Bay, oysters from the same cohort that had been maintained at Stony Brook–Southampton Marine Station were deployed in May 2025 and cultivated during the 2025 growing season (May–November).

Different over-wintering techniques were used for oysters cultivated in the East River and Mt. Sinai Harbor. At the East River site, oysters remained in floating bags throughout winter, which were secured to a dock at SUNY Maritime College away from the kelp lines which were in deep open waters. In the shallow near-shore waters of Mt. Sinai Harbor, where the threat of surface ice exists during winter, oysters were moved to wire cages that sat on the bottom below the surface. Cages with oysters from the experimental ‘with *Ulva*’ treatment were placed between the horizontal lines of kelp and were surrounded by kelp to continue co-cultivation with seaweed. Cages with oysters from the control ‘without *Ulva*’ treatment were placed away from the influence of the kelp lines to continue cultivation without seaweed.

Kelp Monitoring

Kelp biomass growth and the carbon and nitrogen (CN) content of kelp tissue were monitored on a monthly basis from December through May by collecting triplicate samples from

each open-water kelp line deployed at each site. For each sample, all kelp biomass, including blade, stipe, and holdfast, was excised from a 15 cm length of line, and transported back to the Southampton Marine Station laboratory on ice packs in coolers. Upon arrival, the samples were first cleaned of biofouling and spun in a salad spinner to remove excess water before the following measurements were made: wet weight, average kelp blade length, blade width, and stipe length. Following these measurements, the samples were dried at 60 °C in a laboratory drying oven. After complete drying, each sample was prepared and weighed for analysis of CN content, as described in the subsection below titled *Seaweed Tissue C and N*. In addition to making CN measurements on whole kelp samples (i.e. blade + stipe), blades and stipes were also separated and dried for individual CN analysis. Kelp biomass was expressed in terms of wet weight per unit length of line (i.e., kg m⁻¹), which was referred to as line yield. CN content was expressed as a percentage of tissue dry biomass.

To quantify the amount of carbon and nitrogen sequestered by kelp, line yields (kg m⁻¹) at peak biomass were first converted from wet weight to dry weight yields, using the average ratio of dry to wet kelp weight obtained for each site. The dry weight yields were then multiplied by the carbon/nitrogen percentage in the dry kelp tissue, as determined from the CN analysis of dried tissue samples described below. The result expressed carbon and nitrogen sequestration in terms of kilograms of carbon and nitrogen per meter of kelp line. Bioextraction by kelp was scaled to a per hectare farm (100 meter by 100 meter square plot) assuming 1.5 meter spacing between kelp lines, resulting in 67 100-meter kelp lines, or 6,700 linear meters of kelp line per hectare. The assumption of 1.5 meter line spacing is practical for real-world application, and in fact is effectively being used at NY's first commercial kelp farm in Moriches Bay. Additionally, a 1.5-meter line spacing has been used in previous bioextraction estimates for sugar kelp (Kim et al. 2015), allowing for comparisons between studies. The end of the growing season was characterized by a deterioration of the kelp caused by biofouling, grazing, and blade senescence (deterioration due to age) as waters warmed during spring. The optimal harvest time was considered to occur just prior to the onset of kelp deterioration and biofouling when line yields were at or near maximum. When kelp growth slowed and kelp tissues began to decay in mid-to-late spring, all kelp was harvested from lines and brought to the Stony Brook – Southampton campus for drying and/or composting. One-way ANOVAs and Tukey HSD tests were used to compare the optimal line yields among sites. After final kelp harvesting, *Ulva* spp. were deployed

as described above.

Ulva monitoring

To accommodate rapid growth and bioextraction, all but 20 g of *Ulva* was completely harvested from grow-out bags and replaced every 3 to 4 weeks from May through November, an approach the Gobbler Lab had iteratively discovered maximized yields of *Ulva* in Long Island coastal waters (Appendix Fig 5). At the end of each cultivation round, *Ulva* was removed from bags, cleaned of biofouling, spun in a salad spinner to remove excess water, and weighed. Subsamples of the cleaned seaweed were weighed and dried in a drying oven at 60 °C. After complete drying, each sample was prepared and weighed for analysis of CN content, as described in the subsection below titled *Seaweed Tissue C and N*.

The amount of *Ulva* biomass grown in a bag over the course of a cultivation period, referred to as ‘bag yield’, was computed as the difference between final and initial wet weights (kg bag⁻¹). Since daily growth is a percentage of the starting amount of biomass, the absolute amount of biomass grown over a given period of time is dependent on the initial starting quantity of *Ulva* in each bag. The daily growth rates were therefore computed as a percentage per day using the following equation:

$$GR = (10(\log(F/I)/t) - 1) \times 100$$

where: GR = growth rate (% per day)

F = final wet weight

I = initial wet weight

T = experiment duration (days)

To quantify the amount of carbon and nitrogen sequestered by *Ulva*, bag yields (kg bag⁻¹) at each harvest were first converted from wet weight to dry weight yields, using the average moisture content of *Ulva* tissue obtained at each site from *Ulva* samples dried over the course of the study. The dry weight yields were then multiplied by the carbon and nitrogen percentages in the dry *Ulva* tissue, as determined from the CN analysis of dried tissue samples described above. The result expressed carbon and nitrogen sequestration in terms of kilograms of carbon and nitrogen per bag of *Ulva*. Bioextraction by *Ulva* was scaled to a per hectare assuming 4,000 surface bags per hectare (50 100-meter rows of 80 bags each), which is a realistic farm design comparable to farm designs currently deployed by Long Island oyster farms. Harvested *Ulva* were processed as described above for kelp.

Oyster monitoring

The average weight and height (i.e. longest shell dimension) of every oyster in each replicate grow-out bag was measured every three to four weeks when *Ulva* was harvested, and weight and height-based growth rates were calculated in g d^{-1} and mm d^{-1} , respectively. Growth in ‘with seaweed’ and ‘without seaweed’ treatments was compared using Student t-tests. At three time-points (April/May 2025, July 2025, and final harvest in October/November 2025), 20 oysters were removed from each site, transported to the lab on ice packs in coolers, and shucked for measurements of dry tissue weight and dry shell weight. After complete drying, each tissue and shell sample was prepared and weighed for analysis of CN content, as described in the subsection below titled *Bivalve Tissue C and N*. The average amount of carbon and nitrogen (grams) per oyster at harvest at each site was calculated by multiplying the average dry weight of tissue and shell per oyster by the average CN content (%) of tissue and shell, respectively, and then adding the two components together. Bioextraction was scaled to a per hectare basis assuming 800,000 oysters harvested per hectare per year (200 oysters from each of 4,000 floating bags). This is a conservative harvest estimate and below the estimate of one million oysters per hectare used by Grizzle et al (2017).

Water Quality Monitoring

The extent to which deployment of seaweeds and/or bivalves modified existing water chemistry was monitored in multiple ways. Firstly, multi-parameter sondes were deployed to continuously monitor levels of dissolved oxygen (DO), temperature, and pH (NBS scale) within seaweed deployments and control locations away from the deployments. Continuous data were internally logged within sondes every 10 min and were cleaned and downloaded every two weeks. To complement these continuous measurements, discrete samples were collected at each site and region for the measurement of dissolved nutrients and chlorophyll *a*. Samples were collected every two weeks in triplicate from sites with and without the aquaculture of seaweeds.

Dissolved nutrients (nitrate/nitrite) were collected and quantified in triplicate on filtered (0.2 μm polysulfone capsule) seawater samples and frozen for later analysis. Frozen dissolved nutrients samples were thawed prior to analysis. Once frozen samples were fully thawed, analyses were performed on the samples within an hour. Extracted chlorophyll *a* was measured from samples collected on glass fiber filters and analyzed fluorometrically (see below, Parsons &

Strickland, 1963; Parsons, 2013). All sample equipment was cleaned and decontaminated via rinsing with 10% HCl followed by liberal rinsing with deionized water; 10% HCl was reused and ultimately removed by Stony Brook University Environmental Health and Safety.

Seaweed Tissue C and N

For seaweed tissue processing prior to C and N analysis, samples of *S. latissima*, and *Ulva* spp. were placed in weighing tins (~1 g) in a drying oven set to 60°C for several days until completely dry. Following this, samples were homogenized into a fine powder using a mortar and pestle before an aliquot of tissue (~4-6 mg; weighed on the Scientech ZSA 120 digital microbalance) was placed in analytical-grade tin capsules (~5 mm). The carbon and nitrogen were measured on a Thermo Elemental Analyzer at the NYS Center for Clean Water Technology, which held NYSDOH ELAP certification for this analysis. For each sample analysis run, a series of standards with known carbon and nitrogen concentrations that covered the expected range of elemental C and N contents in the sample set were included. The data from these standards were used to calculate elemental C and N in seaweed samples. Quantification of tissue C and N was based on methods by Sharp (2005).

Bivalve Tissue and Shell C and N

Sample bivalves were weighed on a Scientech ZSA 120 microbalance before being shucked. The internal tissue was separated from the shell, and both were weighed and placed in separate drying dishes. The tissue and shell of the bivalves were placed in a drying oven set to 60°C and allowed to dry to a constant mass (Carroll et al., 2008; Abeels et al., 2012). Following this, samples were homogenized into a fine powder using a mortar and pestle before an aliquot of tissue (~4-6 mg; weighed on the Scientech ZSA 120 digital microbalance) was placed in analytical-grade tin capsules (~5 mm). The elemental CN content of the tissue and shell was measured at the New York State Center for Clean Water Technology (Abeels et al., 2012). For each sample analysis run, a series of calibration standards with known carbon and nitrogen concentrations that covered the expected range of elemental C and N contents in the sample set were included. The data from these standards were used to calculate elemental C and N in oyster tissue and shell samples. Quantification of tissue C and N was based on methods by Sharp (2005).

Dissolved Nutrients

Samples of seawater taken on-site were analyzed for nitrate using USEPA standard methods 353.2. All analyses were performed by the New York Center for Clean Water Technology, which held NYSDOH ELAP certification for these analyses. Nitrate/nitrite was analyzed by diazotizing with sulfanilamide and then coupled with N-(1-naphthyl) ethylenediamine dihydrochloride. Nitrate was quantitatively reduced to nitrite by passage of the sample through a copperized cadmium column, resulting in a measurement of nitrate and nitrite in the sample. A set of standards was also prepared before analyses. Following preparation of reagents and standards, the manifold was set up, data system parameters were inputted, and deionized water was pumped through all reagent lines to check for leaks and smooth flow. Reagents were then allowed to flow through the reagent lines until the system reached a stable baseline. The instrument was then calibrated by injection of standards into the system. Samples and/or standards were then placed into the sampler of the instrument with information inputs required by the data system (i.e., concentrations, replicates, and QuikChem scheme). The data system of the instrument prepared a calibration curve by plotting reaction response versus standard concentrations. Concentration of the samples was calculated from the generated regression equation. Values that fell between the lowest and highest calibration standards were reported, while samples that exceeded the highest standard were diluted and reanalyzed. Upon completion of analyses, samples and reagents used during analyses were carefully disposed in marked containers and safely stored until containers could be properly disposed.

Suspended chlorophyll a

Prior to the day of sampling, 7 mL scintillation vials were labeled, in triplicate, with the site, date of collection, and space left for indication of the volume of water filtered. The sample bottles containing the water collected in triplicate from each site were kept cool (4°C) but not frozen, and were filtered within < 6 hours after the water samples were collected. Gloves were worn and basic aseptic techniques were followed to avoid contamination of the sample. Using sterilized forceps, a glass fiber filter (pore size = 0.7µm) was placed on the base of a filtration system before attaching the filter tower. The sample bottles were inverted several times before a suitable volume (100mL) was measured in a graduated cylinder and transferred into the filter tower. Using a vacuum pump (pressure <5psi), the sample water was filtered through until the

filter was nearly dry. Then the filter tower was thoroughly rinsed with 0.2 μ m filtered seawater from the collection site or a site with equal salinity and filtered through again until the filter was nearly dry. After removing the filter tower, the filter was removed from the base using sterilized forceps. The filter was then folded in half, placed in its respective labeled scintillation vial, and stored at -20°C until it was ready to be processed for chlorophyll *a* within 7 days.

The sample filters were processed using 4mL of 90% acetone added to each scintillation vial, which was then placed in a freezer overnight for 24 h prior to analysis. After 24 h, samples were extracted via pipette and placed into a glass scintillation vial. Each vial was wiped carefully with a Kimwipe before placed into a Trilogy fluorometer, which is calibrated semi-annually with liquid certified standard and daily with a solid standard and blank (see below). The parameters on the fluorometer were set to account for the volume of the acetone (4mL) as well as the volume of the water sample that was filtered (100 mL). As the samples were collected in triplicate, the average and standard deviation of the three samples reflected the concentration of total chlorophyll *a* for the given site on the given date. Triplicate filtered seawater blanks were run with each analytical run and the detection limit was defined as three-times the standard deviation of blank measurements. Procedures for quantifying chlorophyll *a* and QA/QC standards for the procedure were based on Parsons and Strickland (1963) and Parsons (2013). Upon completion of analyses, samples and reagents used during analyses were carefully disposed of in marked containers and safely stored until containers could be properly disposed.

RESULTS

Kelp bioextraction of N and C

During the 2023-2024 winter growing season, kelp deployed in late 2023 began growing robustly at all sites in February and March 2024 (Fig 2A). While the Oyster Bay sites needed to be removed in early April due to resuming marina operations, yields at all other sites peaked in May (Fig 2A). Final yields for the East River, Oyster Bay, Northport Harbor, and Mount Sinai Harbor were approximately 6, 2, 2, and 1.3 kg m⁻¹, respectively (Fig 2B), with the East River yields being significantly greater than all other sites and the Northport Harbor yields being significantly greater than Mount Sinai Harbor ($p < 0.05$; Tukey HSD; Fig 2B). Trends in 2025 generally paralleled those in 2024, but with higher yields at each site (Fig 3). The final kelp yields in 2025

for the East River, Oyster Bay, Northport Harbor, and Mount Sinai Harbor were approximately 14, 3, 4, and 3 kg m⁻¹, respectively (Fig 3B), with the East River yields being significantly greater than all other sites ($p < 0.05$; Tukey HSD; Fig 3B).

It should be noted that a parallel ongoing study conducted by the Gobbler Lab at these sites has found that substantially higher kelp yields can be achieved at the lower growth sites through the earlier outplanting of seedstock (i.e. November) produced using gametophyte cultures, as opposed to spore-derived seedstock which was used in this study (Appendix Figs 7, 8). As described in the Methods section above, the production of spore-derived seedstock is dependent on the collection of reproductive tissue (i.e. sorus tissue) from local wild kelp populations. In NY waters, reproductive tissue typically cannot be found until late October/early November, and spore-derived seedstock is therefore typically not ready for field deployment until December. More recently, however, techniques have been developed to culture kelp gametophytes year-round, allowing for the production of kelp seedstock at any time. Thus, with gametophyte-derived seedstock, kelp lines can be seeded earlier in November as soon as water temperatures are cool enough (i.e. < 15 °C). This means that the bioextraction potential of kelp farming is potentially even greater than reported here for kelp lines seeded in December with spore-derived seedstock.

The moisture content of kelp varied in both space and time during this project (Fig. 4). During both years, the moisture content of kelp was highest in Oyster Bay and lowest in Mount Sinai Harbor (Fig 4). In 2024, the moisture content of kelp in Oyster Bay was significantly higher than Mount Sinai and East River kelp ($p < 0.05$; Tukey HSD; Figure 4B). In 2025, the moisture content of kelp in Oyster Bay was significantly higher than all other sites ($p < 0.05$; Tukey HSD; Figure 4B). The high moisture content in Oyster Bay likely relates to its earlier removal date (April) than other sites (May); as the physiological condition of the kelp deteriorates in warmer water, they hold less water.

The nitrogen (N) content of kelp varied in both space and time during this project. During both years, the nitrogen content of kelp declined with time across all sites (Fig 5). The percent N content at all sites was above 3% during winter from December to February and then declined during spring from February to May (Fig. 5). In both years, the final nitrogen content of kelp followed the expected eutrophication gradient. In 2024, the highest levels of kelp nitrogen content were found in the East River (2.9%) and lower levels in Oyster Bay (1.6%), Northport Harbor

(2%) and Mount Sinai Harbor (1.2%) (Fig 6A), and the levels at each site were significantly different from each other ($p < 0.05$; Tukey HSD; Fig 6A). In 2025, kelp nitrogen percentages were slightly lower: the East River, Oyster Bay, Northport Harbor, and Mount Sinai Harbor percentages were 2.4%, 1.6%, 2%, and 1%, respectively, (Fig 6B), and the levels at each site were significantly different from each other ($p < 0.05$; Tukey HSD; Fig 6B).

The carbon (C) content of kelp was less varied than nitrogen. In 2024, final kelp carbon content was significantly higher in the East River (31%) compared to Oyster Bay (24%), Northport Harbor (26%), and Mount Sinai Harbor (27%) ($p < 0.05$; Tukey HSD; Fig 7A). In 2025, kelp carbon percentages were slightly lower; the East River, Oyster Bay, Northport Harbor, and Mount Sinai Harbor percentages were 24%, 21%, 25%, and 27%, respectively (Fig 7B), and the kelp carbon levels at Northport Harbor and Mount Sinai Harbor sites were significantly greater than from Oyster Bay ($p < 0.05$; Tukey HSD; Fig 7B).

The carbon and nitrogen (CN) content of blades and stipe were highly similar and did not follow a particular trend (Fig. 8). In some cases, the stipe of kelp had higher carbon and nitrogen content compared to the blades (Fig. 8). In other cases, the blades of kelp had higher carbon and nitrogen content compared to the stipes (Fig. 8). CN values for the whole combined sample typically fell between that for blades and stipes, and were closer to the CN value of the blades as blades comprise the majority of the biomass. The CN values for the whole combined sample were used for bioextraction calculations, as it is expected that all kelp biomass (blade and stipe) would be harvested for maximal bioextraction.

The total amount of N extracted by kelp (kelp bioextraction rates) at each site generally paralleled patterns regarding growth and N content, with the kelp grown at the East River site standing out as being more effective than all other sites in both years. For example, in 2024, the N extracted per meter of kelp line for the East River, Oyster Bay, Northport, and Mount Sinai Harbor was 21.8, 2.6, 4.5, and 1.4 g N per meter, respectively (Fig 9A). The 2025 growing season yields were significantly higher at 30.4, 3.3, 8.8, and 3.3 g N per meter, for East River, Oyster Bay, Northport, and Mount Sinai, respectively (Fig 9B).

The kelp bioextraction rates achieved during the study for both years were scaled to a per hectare scale assuming 1.5 meter spacing between lines, which results in sixty-seven 100-meter-long lines per hectare (i.e. 6,700 linear meters of kelp per hectare). If the nitrogen bioextraction

rates achieved in 2024 were extrapolated to a one-hectare kelp farm, the theoretical kelp farm would extract an estimated 145, 17, 30, 10 kg N per hectare for East River, Oyster Bay, Northport Harbor, and Mount Sinai Harbor, respectively (Fig 10A). In 2025, yields were substantially larger, so a theoretical one-hectare kelp farm would extract an estimated 204, 22, 59, and 22 kg N per hectare for East River, Oyster Bay, Northport, and Mount Sinai Harbor, respectively (Fig 10B).

Similarly, the amount of carbon extracted by kelp was significantly greater in the East River than all other sites in both 2024 and 2025 (Fig 11; ANOVA's log₁₀ transformed, $p < 0.001$, $F \text{ value} = 57.97$ for 2024; $p < 0.001$, $F \text{ value} = 19.86$ for 2025). In 2024, the C extracted per meter of line was 233.4, 39.9, 61.7, and 33.5 g C per meter, for the East River, Oyster Bay, Northport, and Mount Sinai Harbor, respectively (Fig 11A). The 2025 growing season yields were higher at 313.0, 43.3, 113.1, and 95.8 g C per meter, for East River, Oyster Bay, Northport, and Mount Sinai, respectively (Fig 11B).

Extrapolating from the 2024 results, a one-hectare kelp farm would theoretically extract an estimated 1,564, 267, 413, 224 kg C per hectare for East River, Oyster Bay, Northport, and Mount Sinai Harbor, respectively (Fig 11A). In 2025, yields were substantially larger, so scaling the kelp bioextraction rates to a theoretical one-hectare farm would extract an estimated 2,097, 290, 758, and 642 kg C per hectare for East River, Oyster Bay, Northport, and Mount Sinai Harbor, respectively (Fig 11B).

Ulva bioextraction of N and C

The growth rates of *Ulva* varied by season and year more so than by site (Fig 12). Differences between 2024 and 2025 partly reflect adjustment of methods as the duration of deployments was more variable in 2024 (two-to-four weeks) than in 2025 (mostly four weeks; Fig 12). Seasonally, growth rates generally paralleled temperatures with increasing growth rates during spring, steady and high growth rates in summer, and declining growth rates in autumn (Fig 12). In spring of 2025, growth rates were $< 10\%$ per day at each of the three sites where *Ulva* was cultivated, including East River, Oyster Bay, and Mount Sinai Harbor (*Ulva* was not cultivated in Northport Harbor) (Fig. 12). Peak summer growth rates were generally between 12-13% per day across all three sites (Fig 12). Averaged across the entire 2025 growing season, growth rates of *Ulva* did not significantly differ among sites (ANOVA, n.s.; Fig 13).

Like kelp, the moisture content of *Ulva* did not show strong temporal or spatial trends (Fig.

14). The moisture content varied from 78% to 86% across sites and years (Fig 14). The average moisture content was 82-84% in the East River, 83% in Oyster Bay, and 81-82% in Mount Sinai (Fig 14).

Similar to kelp, the nitrogen content of *Ulva* followed the expected west to east eutrophication gradient in LIS, with the highest tissue nitrogen levels found in the East River and lowest in Mount Sinai Harbor (Fig 15). While the East River stayed relatively consistent through the growing season around 3% nitrogen for both years, Oyster Bay and Mount Sinai seemed to show some seasonality with levels below 3% and even 2% early in the season (May), but levels exceeding 3.6% in later summer (Fig 15). On average, *Ulva* in the East River had more than 3% N for both years, *Ulva* in Oyster Bay was about 3% N in 2025 (was not cultivated in 2024), and the N content of *Ulva* in Mount Sinai Harbor was 2.9% in 2024 and 2% in 2025 (Fig. 15).

The carbon content of *Ulva* did not show strong spatial trends (Fig 16). Across locations and dates, the carbon content of *Ulva* varied from 20% to 30% (Fig. 16). On average, *Ulva* in the East River contained 26 – 29% carbon, *Ulva* in Oyster Bay was about 28% N, and the carbon content of *Ulva* in Mount Sinai was 26 - 27% (Fig. 16).

Unlike kelp, *Ulva* bioextraction is progressive over the course of a season as the *Ulva* was generally harvested every three-to-four weeks and then replaced and allowed to grow out again. *Ulva* bioextraction is a function of the *Ulva* growth rates and N content of the *Ulva* and thus followed the trends of those two parameters. Because the growth rates of *Ulva* followed temperature trends and were seasonal, the bioextraction of N and C showed strong seasonal trends (Figs. 17-18). For example, *Ulva* N bioextraction rates in 2025 for the East River, extrapolated to a theoretical one-hectare farm containing 4,000 surface bags per hectare (50 100-meter rows of 80 bags each), were estimated to be ~3 kg N per hectare in May, increasing progressively to 15 kg N per hectare in July, and then declining each month into October when yields drop to an estimated 2 to 5 kg N per hectare (Figure 17a). Under the same farm design assumptions, peak N bioextraction yields per site over a ~4-week period were estimated to be 16, 11, and 8 kg N per hectare for East River, Oyster Bay, and Mount Sinai, respectively (Fig. 17). Peak C bioextraction yield per site over a ~4-week period were estimated to be 129, 78, and 83 kg C per hectare for East River, Oyster Bay, and Mount Sinai, respectively (Fig. 18).

Comparing kelp and *Ulva* bioextraction of nitrogen

Despite its generally higher nitrogen content than kelp, especially at harvest time, this project found that kelp was significantly more efficient at nitrogen bioextraction than *Ulva* in the East River. In the East River site, extrapolating bioextraction rates to kilograms removed per hectare, estimated kelp bioextraction would be more than 204 kg of nitrogen per hectare per year whereas *Ulva* was only 51 kg of nitrogen per hectare per year (Fig. 19). This translates to 450 pounds of nitrogen per year and 112 pounds of nitrogen per year for kelp and *Ulva*, respectively, for a theoretical one-hectare kelp farm in the East River. Interestingly, for Oyster Bay and Mount Sinai Harbor, the bioextraction yields for *Ulva* exceeded kelp (Fig. 19). For Oyster Bay, an estimated 35 kg of nitrogen per hectare per year would be extracted by *Ulva*, and an estimated 22 kg of nitrogen per hectare per year would be extracted by kelp (Fig. 19). In Mount Sinai Harbor, an estimated 24 kg of nitrogen per hectare per year would be extracted by *Ulva*, and an estimated 22 kg of nitrogen per hectare per year would be extracted by kelp (Fig. 19).

Oyster performance and bioextraction of nitrogen and carbon

Oysters grew robustly at all sites during this study from July 2024 through October 2025 (Fig. 20). Regarding oyster length, oysters in Mount Sinai Harbor grew slightly faster than oysters in the East River (Fig 20). Oysters at both sites were deployed in July 2024 at a starting shell height of ~ 20mm, and by October 2025 (~15 months later) grew to ~ 86 mm in Mount Sinai Harbor and 79 mm in the East River. In the East River, oysters co-grown with *Ulva* in 2024 grew significantly faster than those grown without *Ulva* ($p < 0.05$; T-test; Fig. 20). Oysters were deployed in Oyster Bay in 2025 only and those oysters had slightly larger starting sizes than the oysters deployed in 2024 at Mount Sinai Harbor and the East River. As a consequence, the growth rates of the oysters deployed in Oyster Bay in 2025 were lower than that measured in 2024 for oysters deployed in the East River and Mount Sinai Harbor. At the start of the 2025 oyster growing season, however, the oysters in the East River and Mount Sinai Harbor were larger in size than the oysters just deployed to Oyster Bay, and the 2025 oyster growth rates were consequently higher in Oyster Bay than in the East River and Mount Sinai Harbor (Fig. 20).

Regarding oyster weight, growth patterns were somewhat similar to the length-based outcomes, with even greater differences between the East River and Mount Sinai Harbor oysters. Oysters in the East River and Mount Sinai Harbor began at < 1 g per oyster and grew to 60 g per

oyster in the East River and 90 g per oyster in Mount Sinai Harbor (Fig. 21). The weights of oysters co-grown with *Ulva* in the East River and Mount Sinai Harbor were not significantly different than those without (T-test, n.s. Fig. 21). For Oyster Bay, oysters began at a weight of ~ 2 grams per individual and grew to weights of 50 and 43 grams per individual when grown with and without *Ulva*, respectively (Fig. 21). While the difference was not significant ($p=0.16$; T-test), it again suggests that the co-culture of seaweed with oysters can benefit oyster growth rates.

Comparing final weights across sites, oysters grown in Mount Sinai Harbor achieved double the final dry tissue weight (3 g per oyster) achieved by the oysters in the East River (1.5 grams per individual; Fig. 22a), while the Oyster Bay oysters had an intermediate weight reaching 2.5 g per individual when grown with *Ulva*, but only 2.1 grams per individual without (20% less; Fig. 22a). Final shell weights followed a somewhat similar relationship; Mount Sinai Harbor oysters had the largest shell weight, followed by the East River oysters, and the Oyster Bay oysters being the lightest (Fig. 22b). Within sites, no significant differences were found in final tissue or shell weights between oysters cultivated with seaweeds and oysters cultivated without seaweeds (t-tests, n.s.; Fig 23).

Tissue nitrogen content did not significantly differ among sites (ANOVA, n.s.; Fig 24a), ranging between 7.5% to 8.7%. Shell nitrogen content was substantially lower than tissue nitrogen content, and did significantly differ between sites, averaging 0.12% in the East River and 0.19% in Mount Sinai Harbor ($t=1.73$, $p<0.05$; Fig 24b). Within sites, neither tissue nor shell nitrogen content differed significantly between oysters grown with and without seaweeds (Fig. 25).

Tissue carbon content ranged between 38% to 42% among sites, and was significantly higher in Oyster Bay than Mount Sinai Harbor with no other significant differences among sites (ANOVA, post-hoc Tukey tests, $F=3.74$, $p<0.05$; Fig. 26a). Shell carbon content was very similar between sites, averaging 12.2% in the East River and 12.4% in Mount Sinai Harbor (ANOVA, n.s.; Fig 26b). Within sites, no significant differences were found in either tissue or shell carbon content between oysters grown with and without seaweeds, with the only exception of tissue carbon in Oyster Bay where oysters grown without seaweed had significantly higher tissue carbon content (t-tests, $p<0.05$ for significant difference; Fig. 27).

Based on the final dry tissue and shell weights at harvest and the carbon and nitrogen (CN) content of the oyster tissue and shell at each site, we calculated that harvested oysters contained

0.17, 0.24, and 0.32 g of nitrogen per oyster, and 4.9, 4.1, and 7.4 g of carbon per oyster, at the East River, Oyster Bay, and Mount Sinai Harbor sites, respectively. Extrapolating these carbon and nitrogen bioextraction rates, assuming an annual harvest of 800,000 oysters per hectare per year (see Oyster Monitoring in Methods section), oysters would achieve estimated annual nitrogen bioextraction rates of 135 kg, 185 kg, and 260 kg N per hectare, and estimated annual carbon bioextraction rates of 3,883 kg, 3,232 kg, and 5,897 kg C per hectare, at the East River, Oyster Bay, and Mount Sinai Harbor sites, respectively (Fig 29).

Comparing oyster, kelp, and Ulva bioextraction of nitrogen and carbon

In both Oyster Bay and Mount Sinai Harbor, estimates of nitrogen and carbon bioextraction through oyster farming far exceeded that of both kelp and *Ulva* farming (Fig 29). In the East River, however, kelp farming achieved the highest annual nitrogen bioextraction rates, and oyster farming had the highest carbon bioextraction rates (Fig 29). Combining the bioextraction services of all three species, we estimate that a one-hectare kelp-*Ulva*-oyster farm could theoretically remove 390 kg, 242 kg, and 306 kg of nitrogen per year, and 6,418, 3,815, and 6,849 kg of carbon per year, in the East River, Oyster Bay, and Mt. Sinai Harbor respectively (Fig 30).

Water quality – dissolved nitrogen data

Dissolved nitrogen data displayed clear temporal and largely expected spatial trends, with some exceptions (Fig. 31, 32). Temporally, nitrate levels were higher during the cooler months and lower in the warmer months, with all sites displaying peaks during November and December and levels being lower thereafter (Fig 31). Across sites, the expected west-to-east eutrophication gradient among sites generally held, with Northport Harbor standing out as an exception (Fig. 31, 32). Across the kelp season, the *Ulva* season, and the full study period, nitrate levels were highest in the East River, followed by Oyster Bay, and then Mount Sinai Harbor (Fig. 32). Northport Harbor had levels that were nearly double those of the East River and 3 – 4 times higher than Oyster Bay and Mount Sinai (Fig 32). While this trend was partly driven by the Northport Harbor data only being for the first half of the year, even during the kelp season, the trends were nearly identical (Fig. 32). The sampling location in the back of Northport Harbor is near the discharge site for a sewage treatment plant and groundwater nitrogen levels in this region are known to be extremely high due to dense aggregations of onsite septic systems

(SCWMP, 2020), accounting for the high levels constantly present during this study.

Temperatures

During this two-year project, temperatures followed an expected pattern getting as low as below zero in winter (more so in 2024 than in 2025) and rising to or above 25°C in summer (Figure 33). Sensors from each location had malfunctions for some periods over the two years, yielding minor data gaps. Temperatures were similar across sites and similar between locations with and without seaweeds within sites.

Effects of seaweeds on dissolved oxygen and pH

Across all experimental deployments of kelp and *Ulva*, paired dissolved oxygen (DO) and pH sensors were deployed next to the seaweed deployments and at nearby control locations without seaweeds, allowing for discernment of the effects of seaweeds on these two water quality parameters. The first of these comparisons was during the 2024 kelp deployment in Northport Harbor with sensors recording data from late April through May when kelp biomass growth was at its peak (Fig. 34, 35). Regarding DO, the area with kelp (experimental) reached higher DO maximums compared to the control site but also slightly lower minimums on some dates during the later stages of the deployment (Fig. 34). Notably, the control sites violated the NYSDEC chronic DO minimum (4.8 mg/L) on May 6th while the kelp site did not (Fig. 34). The chronic dissolved oxygen (DO) standard in New York State is a regulatory benchmark set for certain marine waters to protect their designated uses, such as shellfishing, fishing, and recreation. For Class SA, SB, and SC waters, the chronic DO standard is 4.8 mg/L, with an allowable excursion to not less than 3.0 mg/L (i.e. acute DO minimum) for certain periods of time (NYSDEC 2008). This standard applies year-round and is continuously enforceable. Similar patterns were observed with pH (Fig. 35). The pH sensors were deployed and functional for a longer period (April 11 – May 23; Fig. 35). During this time, pH progressively declined for both sites (Fig. 35), a common observation for spring as waters warm and community respiration rates increase (Wallace et al., 2014; Wallace and Gobler, 2021). Throughout the period when both the control and kelp pH sensors were recording, the kelp site had higher pH levels; typically by ~0.3 units (Fig. 35).

The next comparison during the 2024 kelp season was in the East River where pH sensors were logging data from April 30th through June 2nd (Fig. 36). The DO sensors malfunctioned during this deployment. During this time, pH progressively declined for both sites (Fig. 36), a common

observation for spring as waters warm and community respiration rates increase (Wallace et al., 2014; Wallace and Gobler, 2021). Throughout the deployment, the kelp site had higher pH levels; typically by ~0.05 units (Fig. 36). While pH differences were mild during the first half of the deployment, the differences between sites were more apparent in late May as the control site reached nocturnal pH minimums of < 7.3 while the kelp site stayed above this level (Fig. 36).

During the summer of 2024, sensors successfully logged DO and pH during the Mount Sinai Harbor deployment of *Ulva* (Fig. 37, 38). Regarding DO, the *Ulva* logger recorded data from July until October, whereas the control logger was functioning properly from August through October (Fig. 37). During the periods that both loggers recorded DO data, the *Ulva* site experienced daily peaks in DO above 10 mg/L, while the control site did not (Fig. 37). There were also important differences in the nocturnal DO minimums, especially in September when the control site consistently violated the NYSDEC acute DO minimum (3.0 mg/L) but the *Ulva* site did not (Fig. 37). This acute DO minimum level also aligns with the LIS Partnership's definition of hypoxia at 3.0 mg/L (LIS Partnership 2025). Regarding pH, differences between the control and *Ulva* site were smaller than DO and changed over the course of the deployment with the control and *Ulva* each having higher pH at different periods (Fig. 38). Importantly, however, during the previously noted hypoxic event that occurred in mid-September, the control site reached nocturnal pH minima of less than 7.3 while the *Ulva* site did not (Fig. 38).

During the 2025 kelp season, there were successful deployment of DO and/or pH sensors at all four experimental sites. At Mount Sinai Harbor, differences in pH and DO between the kelp (inside) and control (outside) sites were relatively mild (Fig. 39, 40). Regarding DO, levels were indistinguishable during February and early March (Fig. 39). During the second half of March, however, the control site displayed substantially lower DO than the kelp sites (Fig. 39). Regarding pH, levels were quite similar inside and outside of the kelp deployment; while levels seemed higher outside of the kelp in early February, levels became more similar in March with the sensor inside the kelp showing higher peaks and lower minimum pH values (Fig. 40).

For the kelp deployment in Oyster Bay in 2025, paired DO measurements were made from November through February (Fig. 41). During this time, DO levels rose from November through January as temperatures declined, with DO levels starting at around 8 mg/L and peaking at more than 13 mg/L (Fig. 41). DO levels then declined as temperatures rose from late January through

February (Fig. 41). Superimposed on this expected pattern that was driven by temperature-drive saturation, from late December through the end of the sampling period, the kelp site had higher levels of DO than the control site (Fig. 41). The pH record for this deployment went from early December through late March (Fig. 42). The pH displayed seasonal changes paralleling temperature likely driven by carbon dioxide saturation that is known to control pH levels (Fig. 42). While pH at the control site was nearly identical or higher than the kelp site for December and January, once the kelp began to grow robustly in February into March, the kelp site maintained pH values that were higher than the control site (Fig. 42).

For the 2025 kelp experiments in Northport Harbor and the East River, sensors successfully logged pH data only (Fig. 43, 44). The Northport pH sensors were active from December through May and showed that pH levels were consistently elevated within the experimental kelp deployments compared to the control sites (Fig. 43). Similar to the 2024 kelp deployment, pH differences in the East River were more muted (Fig. 44). Sensors were logging from March through June, with the control site having higher pH at the start of the deployment (Fig. 44). During April and early May, pH levels were about the same between the two sites (Fig. 44). Finally, from late May through early June, the experimental kelp deployment site had higher pH compared to the control site (Fig. 44).

The final DO-pH readings for the project were made for the summer/fall 2025 *Ulva* deployments during which there were paired measurements in Mount Sinai Harbor (Fig. 45, 46) and pH only for Oyster Bay (Fig. 47). In Mount Sinai Harbor, DO was logged from June to July and again in September (Fig. 45). In June, DO levels were similar between the experimental *Ulva* site and the control site. In July, the control site displayed bouts of hypoxia in early July, while the *Ulva* site did not (Fig. 45). In September, differences were less clear, with the *Ulva* site having slightly lower DO on average than the control site (Fig. 45). Regarding pH at Mount Sinai Harbor, the *Ulva* site seemed to have higher maximum levels and some lower minimums (Fig. 46). On average, pH levels were higher within the experimental *Ulva* treatment compared to the control (Fig. 46). This was most obvious from late July through September when, with the exception of some nightly excursions, the *Ulva* site had consistently higher pH compared to the control site (Fig. 46). Finally, during the Oyster Bay deployment, the pH sensors were logging from early June through early October (Fig. 47). Differences in pH between the *Ulva* and control site changed during this time (Fig. 47). From June through mid-August, levels were quite similar between sites,

on average (Fig. 47). From mid-August through early October, however, the levels of pH within the experimental *Ulva* site were consistently higher than the control site by a sizeable margin with a close to 0.25 unit difference maintained for much of the period (Fig. 47).

Dynamics of chlorophyll a

During the 2025 deployments of *Ulva*, chlorophyll *a* levels were measured from June through October in the sites with and without *Ulva* (Fig. 48). For both the East River and Mount Sinai Harbor sites, chlorophyll *a* levels were relatively low (1 – 6 µg/L; Fig. 48). There were no clear patterns with regard to one site having higher levels than another (Fig. 48). The levels at Oyster Bay were higher (5 – 14 µg/L), but there were also no substantial differences between the treatment and control (Fig. 48).

DISCUSSION

During this study, *Saccharina latissima* (sugar kelp/kelp), *Ulva* spp., and *Crassostrea virginica* (Eastern oysters) all grew well at all locations. Among the sites, the East River was the most productive site of bioextraction by kelp and *Ulva* but the poorest site for the growth of oysters which grew fastest in Mount Sinai Harbor followed by Oyster Bay. If bioextraction rates achieved during the study were extrapolated to a theoretical one-hectare farm, cultivation of each species at all sites would be capable of the bioextraction of an estimated hundreds of kilograms of nitrogen and thousands of kilograms of carbon per hectare per year. As discussed below, this represents significant portions of total nitrogen loads. Beyond bioextraction, oysters and seaweeds were found to provide several important ecosystem services, including the raising of pH and DO. Collectively, these findings bring critical new insight to the understanding of bivalve and seaweed bioextraction of nitrogen and carbon from Long Island Sound water bodies.

Seaweed bioextraction

The observed spatial variability in nitrogen (N) removal rates—with *Saccharina latissima* (kelp) dominating bioextraction in the East River and *Ulva* spp. demonstrating superior performance in Oyster Bay and Mount Sinai Harbor—underscores the complexity of applying macroalgal (seaweed) aquaculture as a uniform nutrient remediation strategy. The results confirm the substantial N removal capacity of both species in New York waters (Kim et al., 2015),

validating their utility in initiatives like the Nutrient Bioextraction Initiative under the Long Island Nitrogen Action Plan (LINAP), now the Long Island Watershed Program (LIWP) and its Long Island Watershed Action Agenda (LIWAA) (NYSDEC, 2019, 2025). However, the superior performance of one species over the other in different water bodies necessitates an interpretation focused on site-specific environmental characteristics and inherent physiological differences between the macroalgal genera.

The discrepancy likely stems from key physiological differences related to nutrient form, temperature, and water flow. *Saccharina latissima* is a brown alga typically regarded as a winter species (Kim et al. 2015), optimized for growth and N uptake during the cooler periods (December–May). Conversely, *Ulva* spp., a green alga, is known to flourish at relatively higher temperatures and light intensities, often peaking in summer months. Given that the East River is a highly energetic, deep, and tidal strait, the cooler, high-flow conditions may have favored the kelp's robust morphology and large thallus surface area, enhancing its ability to capitalize on the sustained, tide-induced nutrient import (Gobler et al., 2006).

In contrast, the superior performance of *Ulva* spp. in the more enclosed, shallower Long Island embayments (Oyster Bay, Mount Sinai Harbor) suggests these sites exhibited conditions more favorable to the opportunistic green alga. *Ulva* readily assimilates various forms of dissolved inorganic nitrogen (DIN) and organic nitrogen, showing a strong capacity to dominate in nutrient-enriched, shallow waters (Valiela et al., 1997). Furthermore, *Ulva*'s strong capacity for N and phosphorus uptake and robust resilience to environmental changes, particularly elevated temperatures, makes it an ideal biofilter in environments prone to high nitrogen loading (Chapin et al., 2004). The higher temperatures and potentially fluctuating nutrient pulses characteristic of these smaller harbors and bays during the growing season may have provided the optimal conditions for *Ulva*'s rapid growth and high nitrogen-to-biomass conversion efficiency (Valiela et al., 1992, 1997). In addition, the slower growth of kelp in these sites suggests that kelp thrives in high N environments, more so than *Ulva*. These findings emphasize that successful bioextraction deployment requires *in situ* small-scale testing coupled with a species selection model that accounts for the site's unique hydrodynamic, thermal, and nutrient regime rather than applying a single-species approach across an entire region.

Impacts of seaweeds on pH and dissolved oxygen

Aquacultured macroalgae, such as kelp and *Ulva*, may buffer carbonate chemistry to the benefit of nearby aquacultured bivalves. Previous laboratory studies have demonstrated that primary productivity by macroalgae via photosynthesis can promote the growth and survival of calcifying organisms, even under acidified conditions, by increasing pH and Ω (Wahl et al., 2018; Young and Gobler, 2018). Continuous pH measurements from the field experiment reported here demonstrated that pH values were higher within kelp lines and *Ulva* deployments compared to the control site, presumably due to increased primary productivity at the seaweed sites. While the control site pH during this study was lower than the open ocean global average (8.1), it was consistent with levels measured in many estuaries which experience acidification due to eutrophication (Wallace et al., 2014; Cai et al., 2017; Wallace and Gobler, 2021; Wallace et al., 2021). In a manner similar to the observations here, Xiao et al. (2021) reported a 0.10 pH unit increase within a *Saccharina japonica* (kombu) aquaculture area compared to a control site. Young et al. (2022) reported up to a 0.3 pH unit increase when sugar kelp was deployed on a New York oyster farm. Daytime primary productivity by seaweeds has been shown to significantly increase pH compared to non-seaweed sites across horizontal and vertical gradients on diel (daily) and even seasonal timescales (Delille et al., 2000; Delille et al., 2009; Hofmann et al., 2011). Conversely, respiration during the night by macrophyte (seaweed) assemblages release CO₂ (Hofmann et al., 2011; Cornwall et al., 2013), thereby reducing pH and possibly lowering the potential for calcification during the night (Saderne et al., 2015), a pattern reflected in some of the continuous pH measurements. Regardless, previous studies have shown that oysters grown with kelp or *Ulva* can have significantly higher growth rates compared to control sites (Young and Gobler, 2018, 2022; Sylvers et al., 2025). Beyond seaweeds raising pH and buffering carbonate chemistry, kelp detritus has been shown to be a potential food source for bivalves (Duggins et al., 1989; Duggins and Eckman, 1997; Levinton and Shumway, 2002). Hence, kelp may enhance the growth of bivalves by mitigating acidification and, potentially, by enhancing the nutritional status of bivalves.

Eutrophication can act as a driver of acidification and hypoxia in coastal zones (Wallace et al., 2014; Baumann et al., 2015; Wallace and Gobler, 2021), which can negatively affect the growth and survival of bivalves (Gobler et al., 2014; Stevens and Gobler, 2018). Some species of seaweeds, including *Ulva* and kelp, experience enhanced growth under elevated nutrient

conditions (Xu et al., 2011; Young and Gobler, 2016; Young et al., 2021) and grow robustly when aquacultured in eutrophic estuaries (Kim et al., 2015; Jiang et al., 2020). Given that harmful algal blooms (HABs) flourish in eutrophic zones, (Anderson et al., 2002; Anderson et al., 2008), the application of seaweeds in aquaculture to remove excess nutrients may reduce the intensity of HABs, which may indirectly benefit bivalves that are directly harmed by such events (Shumway, 1990). Kelp and *Ulva* may also benefit nearby bivalves by directly reducing densities of HAB species. Various species of red, green, and brown macroalgae have been shown to directly reduce HAB-forming microalgae through allelopathy, which is the production and release of biochemicals that inhibit growth (Wang et al., 2007; Tang and Gobler, 2011; Tang et al., 2015; Sylvers and Gobler, 2021, 2023, 2025; Benitt et al., 2022).

This study has additional broad implications for ecosystems where kelp and bivalves are aquacultured together. The harvest of aquacultured macroalgae, such as *Ulva* and kelp, represents the direct removal of sequestered carbon and nitrogen (Marinho-Soriano et al., 2009; Chung et al., 2011), rather than the return of these elements back into the ecosystem through the eventual degradation of the macroalgae (Bricker et al., 2008). Beyond nutrient assimilation by seaweeds, aquacultured bivalves have the capacity to remove excess nutrients by harvesting of the bivalves as well as by denitrification of particulate organic nitrogen (PON) transferred to the sediment surface via biodeposition (Newell, 2004; Shpigel, 2005; Ayvazian et al., 2021). Additionally, suspension-feeding activity by bivalves can reduce phytoplankton biomass in the water column and increase water clarity and light penetration to the benthos, which benefits seagrass (Newell, 1988; Wall et al., 2008). Finally, cultivated bivalves, kelp, and associated aquaculture gear can provide forage and breeding habitats, as well as a predation refuge for marine life (Walls et al., 2016; O'Brien et al., 2018; Theuerkauf et al., 2021; Armbruster et al., 2024; Wong, 2025).

Oyster performance and bioextraction

The contrasting growth patterns observed between the calcifying oysters and the non-calcifying primary producers (kelp, *Ulva*) highlight a crucial divergence in physiological responses to the distinct physicochemical conditions across the three sites where oysters were co-cultivated with seaweeds in the East River, Oyster Bay, and Mount Sinai Harbor. Specifically, the strong inverse relationship between oyster growth (slowest: East River) and seaweed growth (fastest: East

River) suggests that the dominant limiting factor in the East River is an acidification challenge rather than a limitation in trophic resources given the algal chlorophyll *a* (oyster's food source) levels were similar between Mount Sinai Harbor and the East River. The significantly lower pH recorded in the East River directly implicates coastal ocean acidification (OA) as the likely source of stress for the bivalves. It is well-established that reduced seawater pH impairs the calcification processes in marine shelled molluscs, reducing shell thickness, increasing fragility, and demanding higher energetic expenditure for shell maintenance (Beniash et al., 2010; Gazeau et al., 2013). This diversion of metabolic energy toward acid-base regulation and shell repair limits the energy available for somatic growth, potentially explaining the observed stunting in the lowest pH environment (Dickinson et al., 2012).

Conversely, kelp and *Ulva* growth, which is fueled by dissolved inorganic carbon and nutrients, appears to benefit from the East River environment. While chlorophyll *a* levels were similar across sites, the vigorous current velocity and high turbulence typical of the East River likely enhance the delivery of essential nutrients (particularly nitrate) across the algal boundary layer, thereby facilitating faster growth despite potential hydrodynamic stress (Hurd, 2000; Kregting et al., 2016). Kelp growth rates are known to increase significantly in areas with intermediate to strong currents, as the continuous flow replenishes the nutrients necessary for rapid blade elongation (Gerard & Mann, 1979; Kregting et al., 2016). This creates a scenario where the high flow environment simultaneously benefits the primary producer (kelp) by optimizing nutrient flux while detrimentally affecting the calcifying filter feeder (oyster) by providing an environment with water chemistry unfavorable for shell accretion. These findings underscore the need for species-specific site selection in restoration and aquaculture efforts, particularly when facing gradients of coastal acidification.

Seaweed and oyster bioextraction as a nitrogen mitigation strategy

As part of the Long Island Nitrogen Action Plan (LINAP), now part of the Long Island Watershed Program (LIWP) as of 2025, bioextraction by seaweeds and oysters has become a nitrogen mitigation strategy under serious consideration. During the first decade of LINAP, Nassau and Suffolk Counties developed Nine Element Plans for watershed management that both quantified nitrogen loading rates and also determined the nitrogen mitigation goals by which nitrogen loading rates should be curbed in order improve water quality in differing water bodies.

Located within Nassau and Suffolk County, bioextraction effectiveness can be assessed in Oyster Bay and Northport Harbor as the nitrogen loading rates and mitigation goals for these water bodies are known from these Nine Element Plans. Mount Sinai Harbor was found to have good water quality by the Suffolk County Nine Element Plan with no harmful algal blooms nor hypoxia, and thus nitrogen reductions were not required (reduction goal of 0%).

In addition, for Oyster Bay, bivalve and seaweed aquaculture was already considered as a hypothetical nitrogen mitigation strategy, allowing for a direct comparison with modeling efforts within the Nassau County Nine Element Plan (NCNEP, 2022). Combining the bioextraction services of all three species studied here, we estimate that a one-hectare kelp-*Ulva*-oyster farm would be capable of removing 390 kg, 242 kg, and 306 kg of nitrogen per year, in the East River, Oyster Bay, and Mt. Sinai Harbor, respectively. This total bioextraction in the East River exceeded prior estimates made by the Nassau County Nine Element Plan for seaweed-oyster aquaculture (380 kg of nitrogen per hectare per year), while rates for Mount Sinai Harbor and Oyster Bay fell short of that mark.

Oyster Bay has an annual nitrogen loading rate of 107,000 kg per year and the Nassau County Nine Element Plan set a goal to reduce this load by 60% (NCNEP, 2022; Table 1). The Nassau County Nine Element Plan determined that 1% of embayment surface waters could be dedicated to aquaculture (a conservative estimate, with up to 5% by some Nassau communities) with likely no negative ecological nor socioeconomic consequences. For Oyster Bay, this is about 0.11 square kilometers or 11 hectares, and the Nassau County Nine Element Plan estimated that oyster and seaweed aquaculture could remove about 4700 kg of nitrogen per year in this space. This was based on the assumptions that oysters, kelp, and the red seaweed, *Gracilaria tikvahiae*, could extract 215, 112, and 56 kg per hectare per year, respectively (NCNEP, 2022). The extrapolated nitrogen removal by oysters and kelp based on results of this field study in Oyster Bay was estimated to be 185 kg N and 22 kg N per hectare per year, respectively, well *short of the* Nine Element Plan estimates, and estimated *Ulva* bioextraction in Oyster Bay was 35 kg N per hectare per year. Based on the extrapolated results of the field study, the total Oyster Bay kelp-*Ulva*-oyster bioextraction of nitrogen was estimated to be 242 kg N per hectare per year, a rate that could yield an estimated 2,662 kg nitrogen across the 11 hectares that would be dedicated to aquaculture (Table 1). This value is within the range of what was projected by the Nine Element (i.e. 4.4%) and would remove about 3% of the total N load to this system (Table 1).

For Northport Harbor and Northport Bay, the nitrogen loading rates are 76,318 and 28,536 kilograms nitrogen per year, respectively, and the Suffolk County Nine Element Plan has determined that these water bodies require a 72% and 37% nitrogen reduction, respectively (SCSWS, 2020; Table 1). Their watersheds are 2.14 km² and 7.3 km², respectively, and thus 0.02 and 0.07 km² of their surface waters could be dedicated to aquaculture, which is the equivalent of 2 and 7 hectares, respectively. While we were unable to calculate a precise rate since oysters and *Ulva* were not cultivated in Northport Harbor, if we assume the Oyster Bay rates, this would remove about 558 and 1,953 kg N per hectare per year, respectively. This is 0.7% and 6.8% of the total nitrogen load entering Northport Harbor and Northport Bay, respectively. While the projected Northport Harbor bioextraction reductions are small compared to the reduction goal of 72%, the reduction in Northport Bay accounted for 18.5% of the nitrogen reduction required by the Suffolk County Nine Element Plan, suggesting that bioextraction could be a viable approach for nitrogen mitigation in that system. This difference makes sense as Northport Harbor is one of the most hypereutrophic systems on Long Island as evidenced by it having higher nitrate levels than the East River. In contrast, Northport Bay has a more modest nitrogen loading rate and the aquaculture of bivalves and seaweeds was found to be more effective. While these calculations only considered dedicating up to 1% of surface waters for aquaculture, the open waters of Northport Bay could likely accommodate more than that (>7 hectares) for aquaculture, and therefore could potentially achieve even greater removal percentages. Regardless, these findings suggest bioextraction would be more impactful in moderately eutrophic systems, and less so in the severely eutrophic systems. It is also worth noting that aquaculture activities are more likely to be permissible in moderately eutrophic systems, but might be prohibited due to shellfishing bans in more severely eutrophic systems. Hence, it might be most important to target bioextraction for regions where it would be permitted and most impactful.

Conclusions and future prospects

This study has demonstrated that bioextraction of nitrogen by oysters and seaweeds is a viable nitrogen mitigation approach. Extrapolating bioextraction rates to a theoretical one-hectare farm, the combined aquaculture of two seaweeds and oysters was found to potentially remove an estimated 250 – 440 kilograms of nitrogen per hectare per year. For some ecosystems, this

theoretical farm could remove more than 6% of the total nitrogen load, which is higher than the maximal rate previously estimated for Nassau County water bodies (4%) and nearly 20% of the total N reductions needed for Northport Bay. In addition, there are options for expanding nitrogen bioextraction via aquaculture. As stated above, more than 1% of some surface waters could be dedicated to aquaculture with likely no negative ecological nor socioeconomic consequences. In addition, there are almost certainly opportunities to improve bioextraction by *Ulva*. While oyster and, to a lesser extent, kelp aquaculture has been refined over the years, studies of *Ulva* aquaculture have been rare and, to our knowledge, this was the first attempt to grow *Ulva* for the purposes of removing nitrogen in NY waters. There remains a series of important open questions regarding *Ulva*, including the temporal and spatial dynamics of different *Ulva* species across New York waters, the bioextraction performance of these different species in different locations, and the ideal cultivation approaches for maximizing bioextraction by each species at different locations. Resolving these questions should allow for vast improvement in *Ulva* bioextraction, making this an even more viable nitrogen mitigation approach for New York coastal waters.

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TABLES AND FIGURES

Table 1. Nitrogen loads, reduction goals, and potential bioextraction in three Long Island Sound embayments on the north shore of Long Island, including Oyster Bay, Northport Harbor, and Northport Bay. Estimates of seaweed and oyster bioextraction were extrapolated from the results of this study assuming 1% of the embayments surface water were used for aquaculture.

System	Total N load (kg N/yr)	N reduction needed (%)	N reduction needed (kg)	N removal by seaweeds and oysters (kg N/yr)	Percent of total N load	Percent of N removal needed
Oyster Bay	107,000	60%	64,200	2,662	2.5%	4.1%
Northport Harbor	76,318	72%	54,949	558	0.7%	1.0%
Northport Bay	28,536	37%	10,558	1,953	6.8%	18.5%

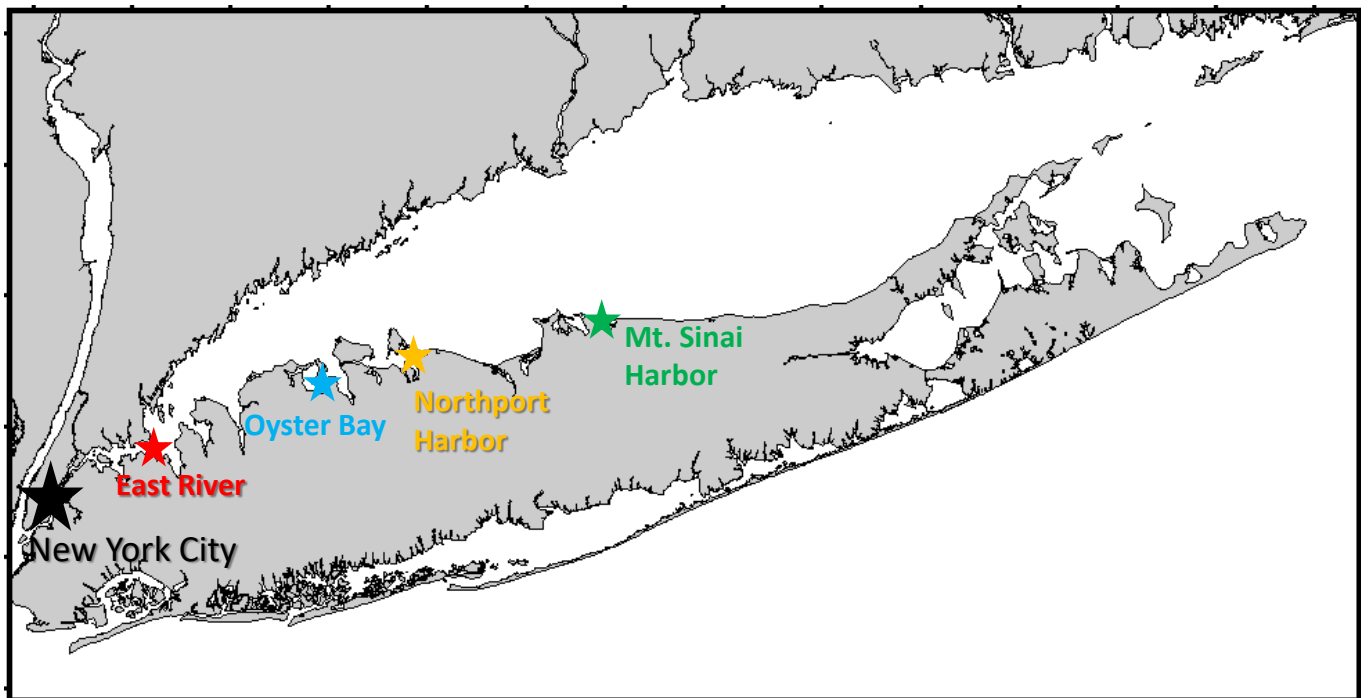


Fig 1. Map of study locations, including, from west to east, East River (red), Oyster Bay (blue), Northport Harbor (orange), and Mount Sinai Harbor, abbreviated as Mt. Sinai Harbor (green).

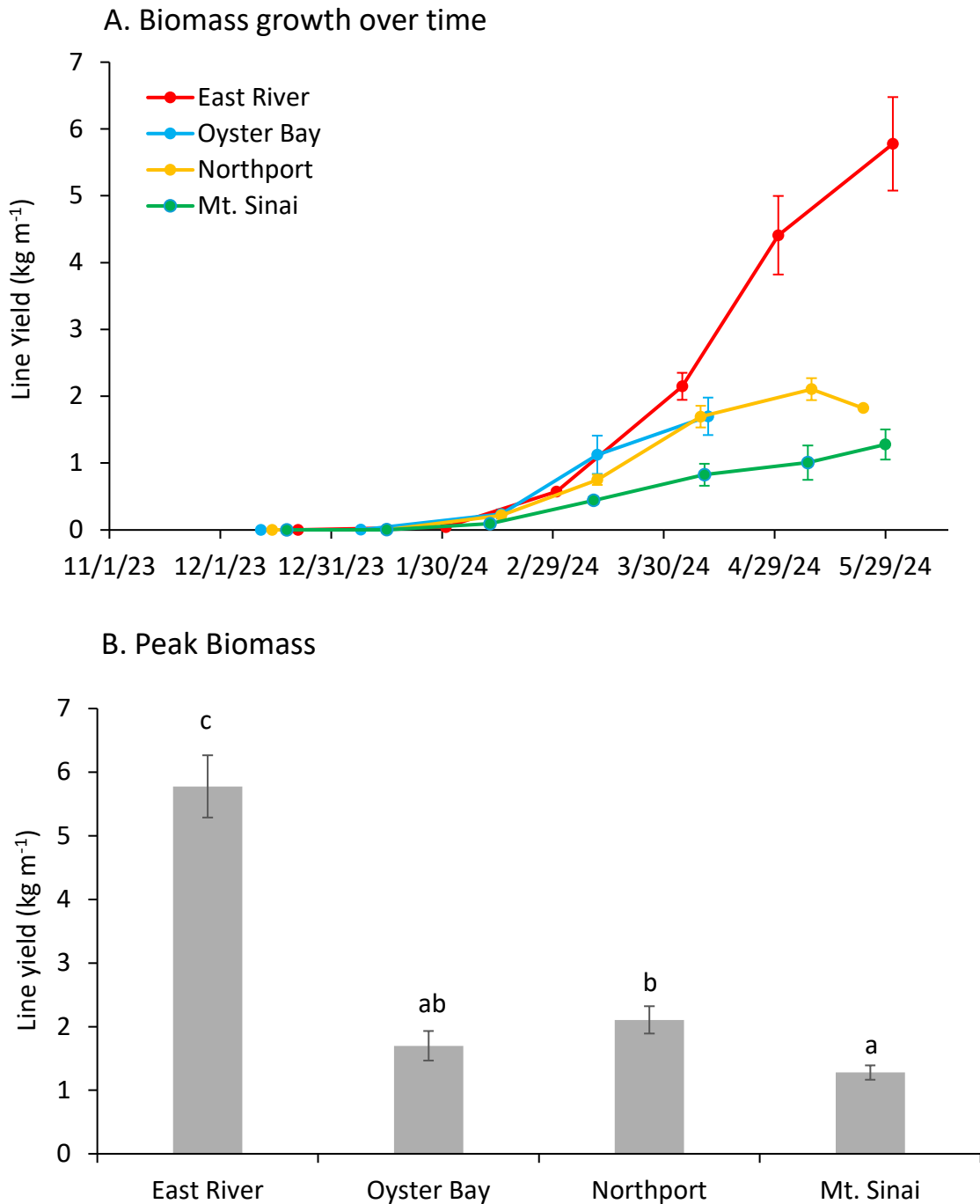


Fig 2. Comparison among sites, including East River, Oyster Bay, Northport Harbor (i.e. Northport), and Mount Sinai Harbor (i.e. Mt. Sinai), of (A) kelp biomass growth over time, and (B) peak biomass yields, along spore-derived kelp lines during the 2024 growing season. Highest line yields across all sites were achieved at the East River site on 5/31/24 with average line yields of 5.8 kg m⁻¹. Kelp lines in Oyster Bay were harvested early on 4/11/24 and most likely did not reach peak biomass. An ANOVA (log₁₀ transformation, $p < 0.001$, F value=36.99) was conducted on peak biomass and letters above bars denote significant differences between treatments (Tukey, $p < 0.05$).

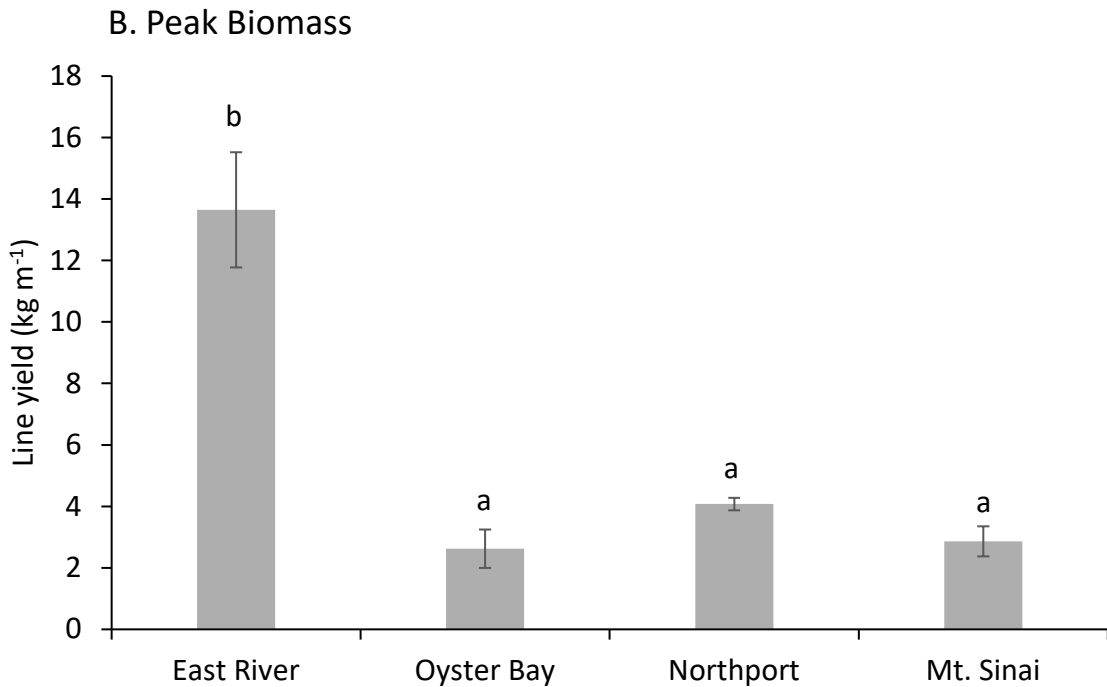
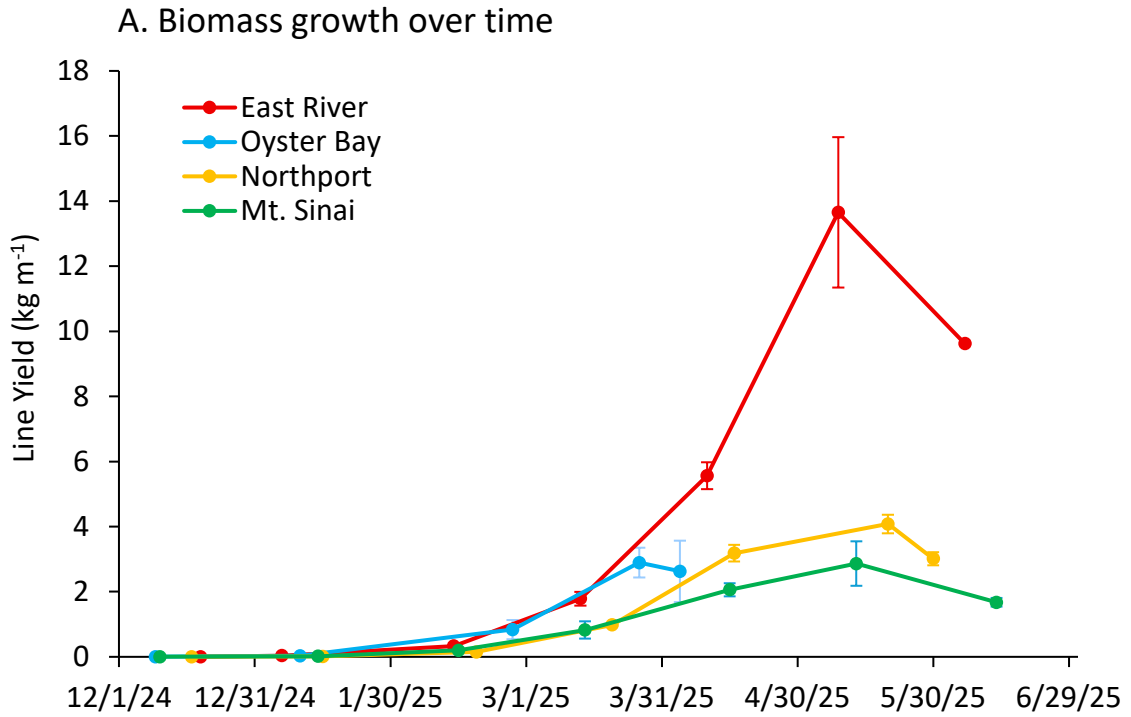


Fig 3. Comparison among sites of (A) kelp biomass growth over time, and (B) peak biomass yields, along spore-derived kelp lines during the 2025 growing season. Highest line yields across all sites were achieved at the East River site. Kelp lines in Oyster Bay were harvested early on 4/4/25 and most likely did not reach peak biomass. An ANOVA (log₁₀ transformation, $p < 0.001$, F value=20.68) was conducted on peak biomass and letters above bars denote significant differences between treatments (Tukey, $p < 0.001$).

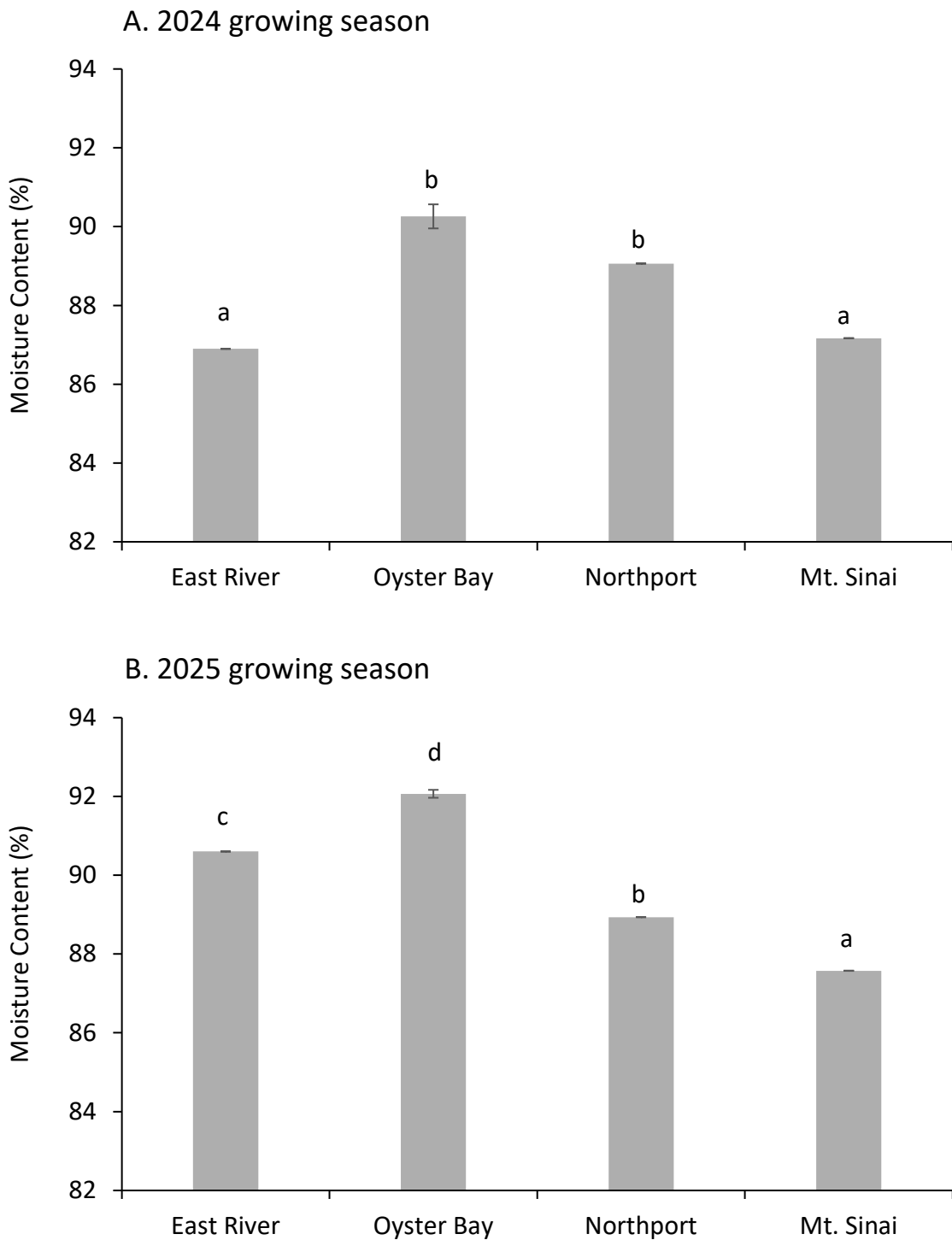


Fig 4. Comparison of kelp tissue moisture content among sites for the **(A)** 2024 and **(B)** 2025 kelp growing seasons. An ANOVA ($p < 0.001$, F value=24.13) was conducted on 2024 moisture content and letters above bars denote significant interactions between treatments (Tukey, $p < 0.01$ for all significant interactions). A GLM ($p < 0.001$) was conducted on 2025 moisture content and letters above bars denote significant differences between treatments (Tukey, $p < 0.001$).

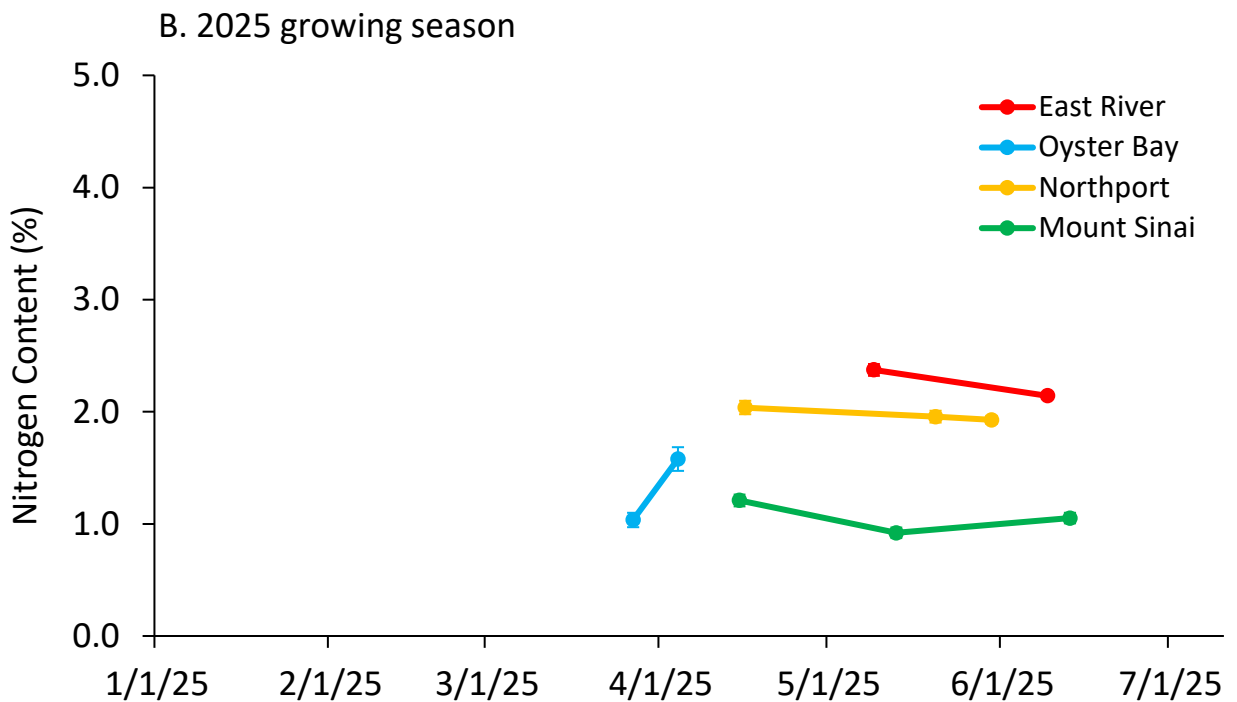
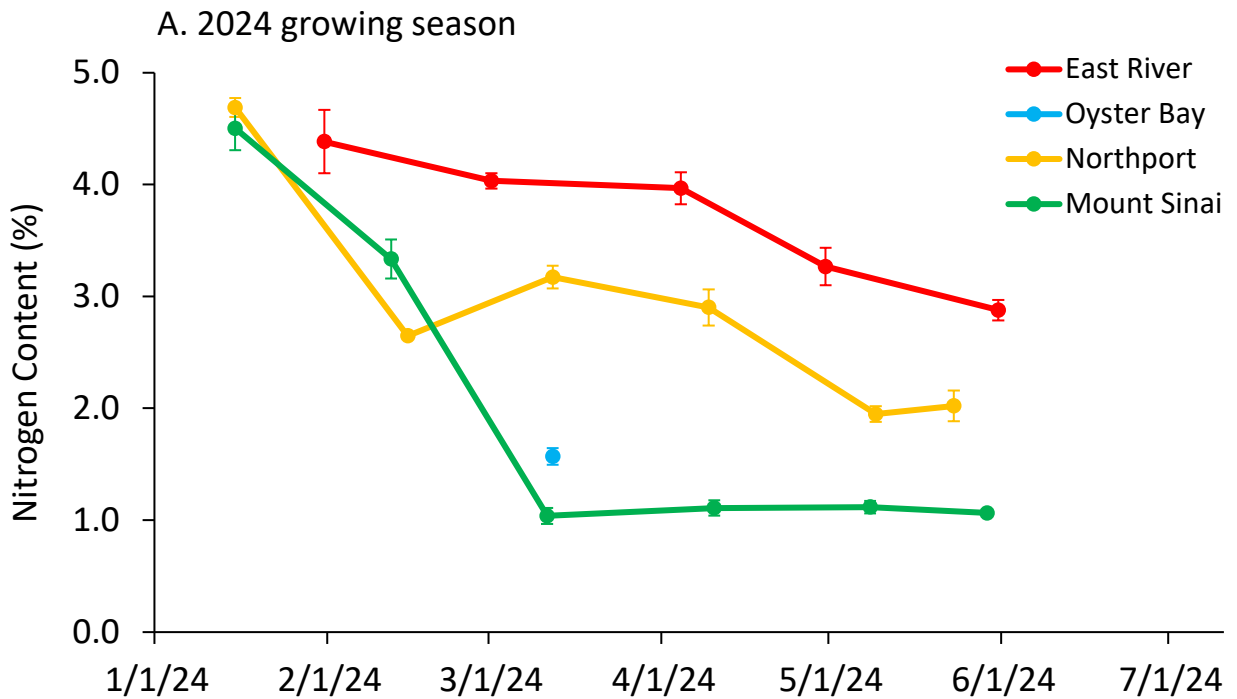


Fig 5. Nitrogen content of kelp tissue over the course of the (A) 2024 and (B) 2025 growing season at four locations in Long Island Sound, including: 1) East River (red), 2) Oyster Bay (blue), 3) Northport Harbor (orange), and 4) Mount Sinai Harbor (green). The nitrogen content is expressed as a percentage of the tissue biomass (dry weight). Error bars are standard errors.

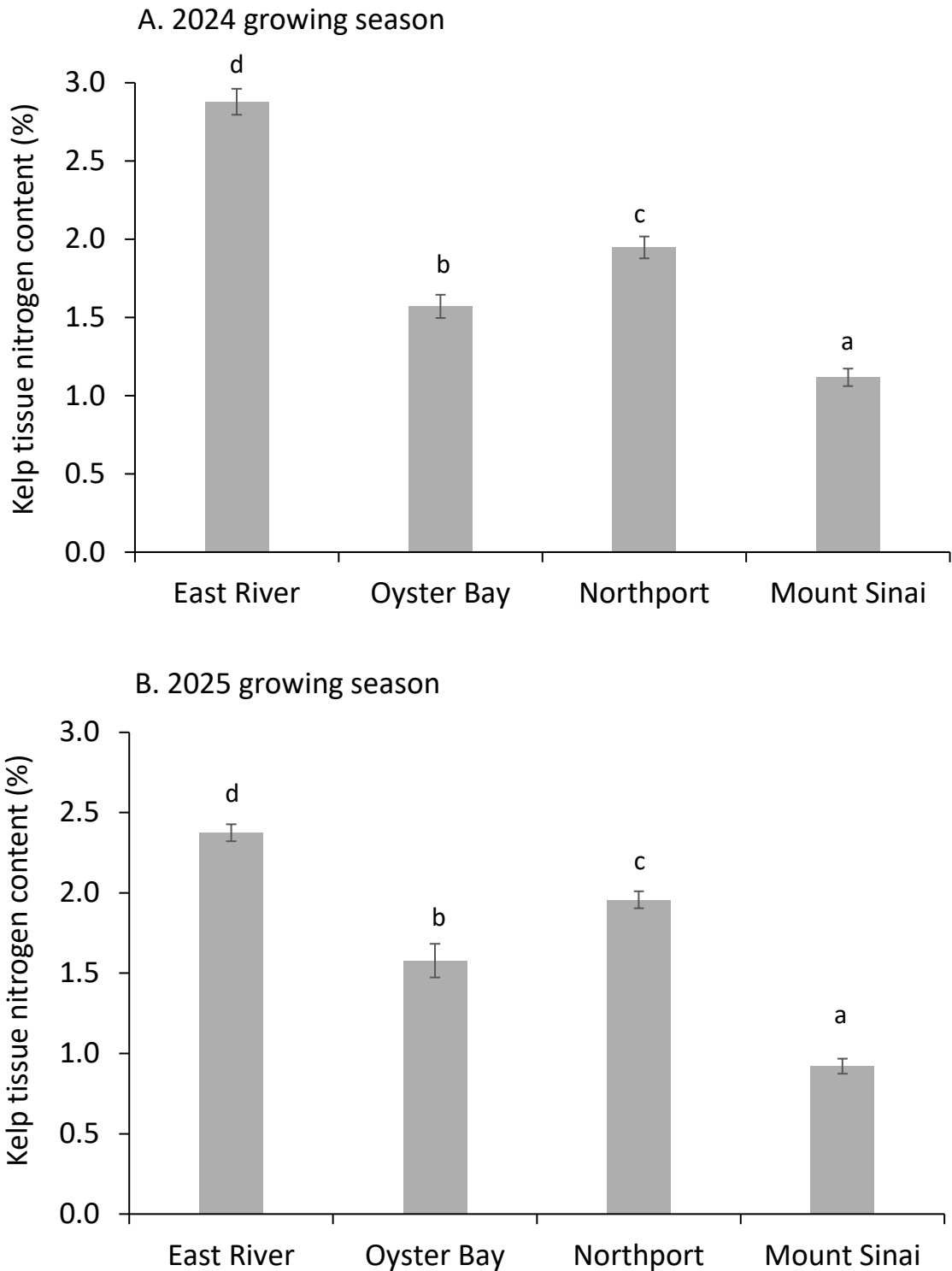


Fig 6. Comparison of kelp tissue nitrogen content at peak biomass among sites for the (A) 2024 and (B) 2025 kelp growing seasons. ANOVA's ($p < 0.001$, F value=104.3 for 2024; $p < 0.001$, F value=129.3 for 2025) were conducted on 2024 and 2025 nitrogen content and letters above bars denote significant differences between treatments (Tukey, $p < 0.05$ for all significant differences in 2024 and $p < 0.01$ for all significant differences in 2025).

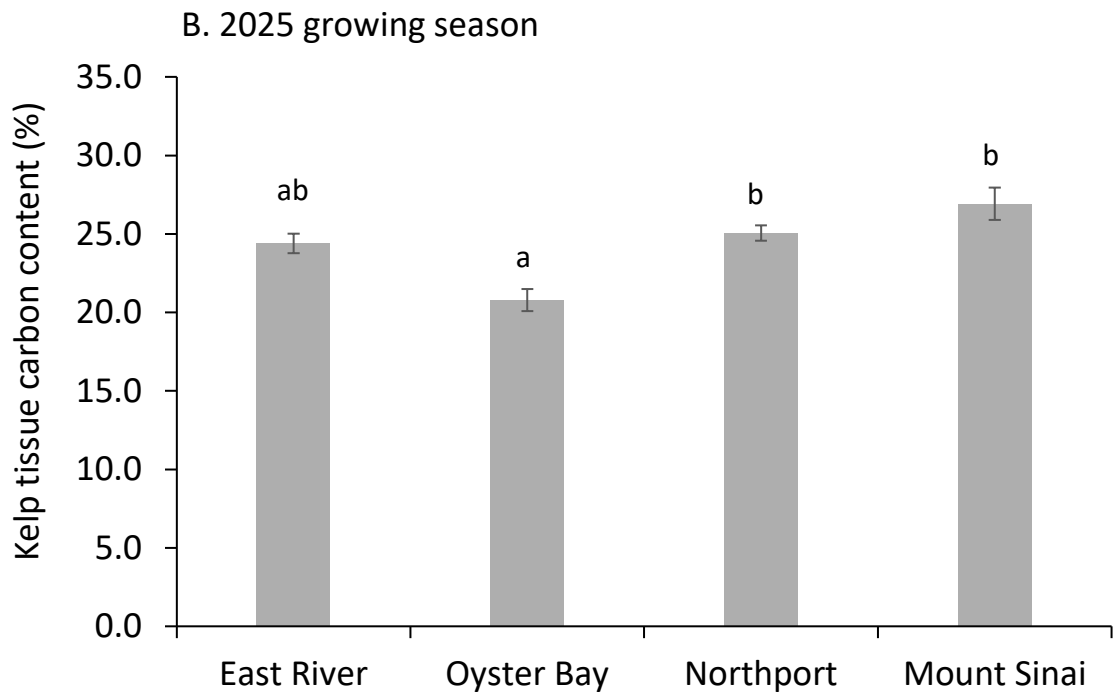
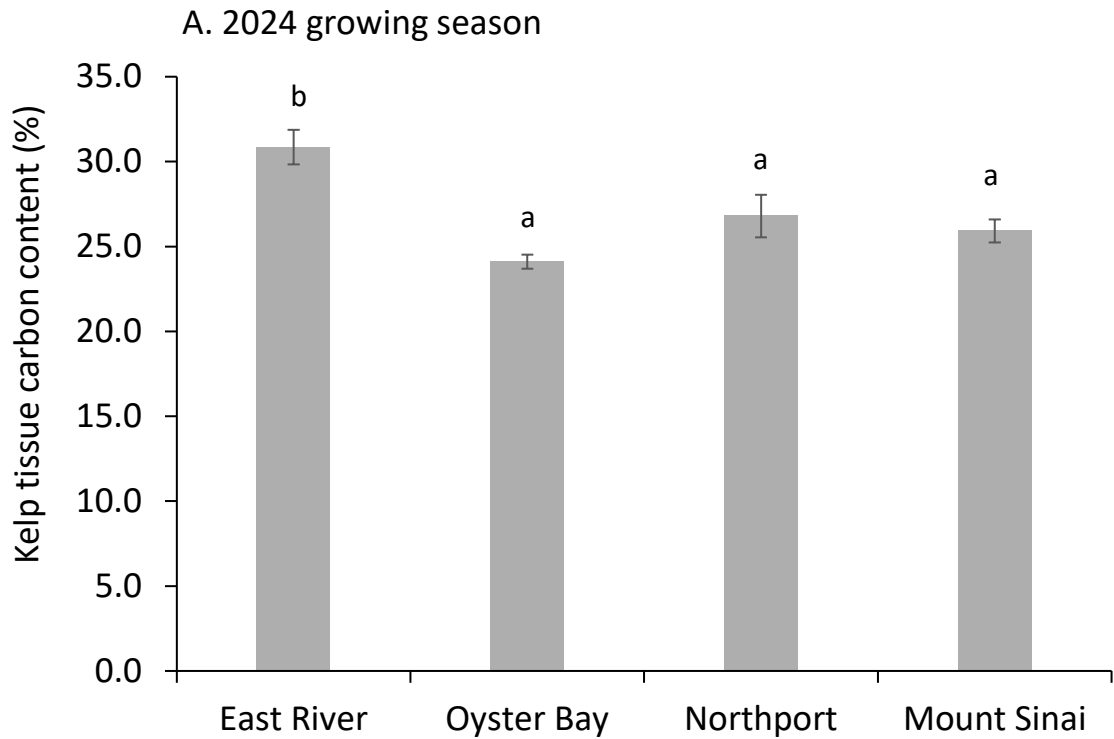


Fig 7. Comparison of kelp tissue carbon content at peak biomass among sites for the **(A)** 2024 and **(B)** 2025 kelp growing seasons. ANOVA's ($p < 0.001$, F value=11.88 for 2024; $p < 0.01$, F value=7.355 for 2025) were conducted on 2024 and 2025 carbon content and letters above bars denote significant differences between treatments (Tukey, $p < 0.05$ for all significant differences in both years).

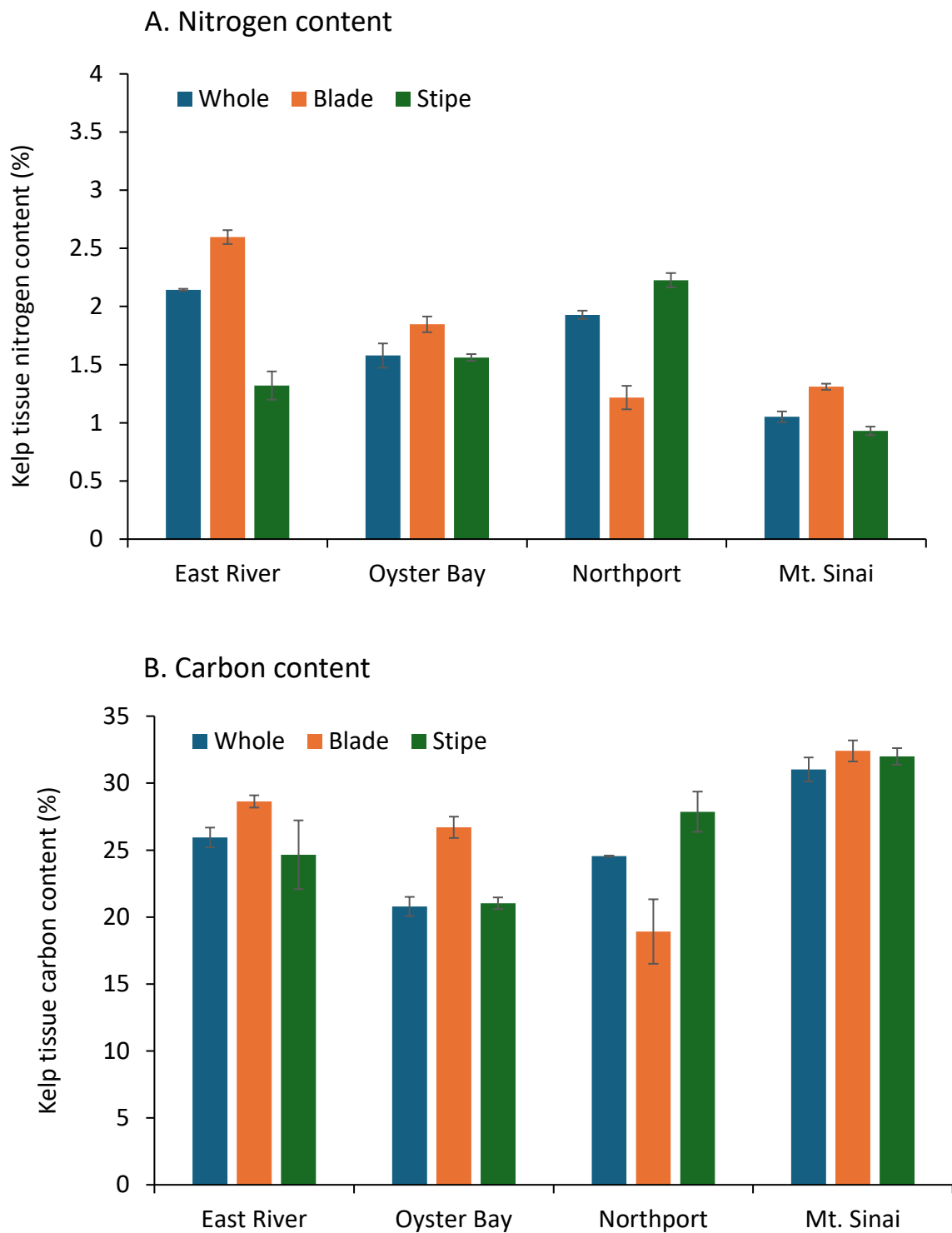


Fig 8. Comparison of (A) nitrogen and (B) carbon content between blade tissue, stipe tissue, and whole tissue (blade + stipe) of kelp tissue on the final harvest date at each site in the 2025 kelp growing season. CN content data from spore-derived and gametophyte-derived kelp tissue is combined within each site. CN content of whole kelp samples were used in bioextraction calculations.

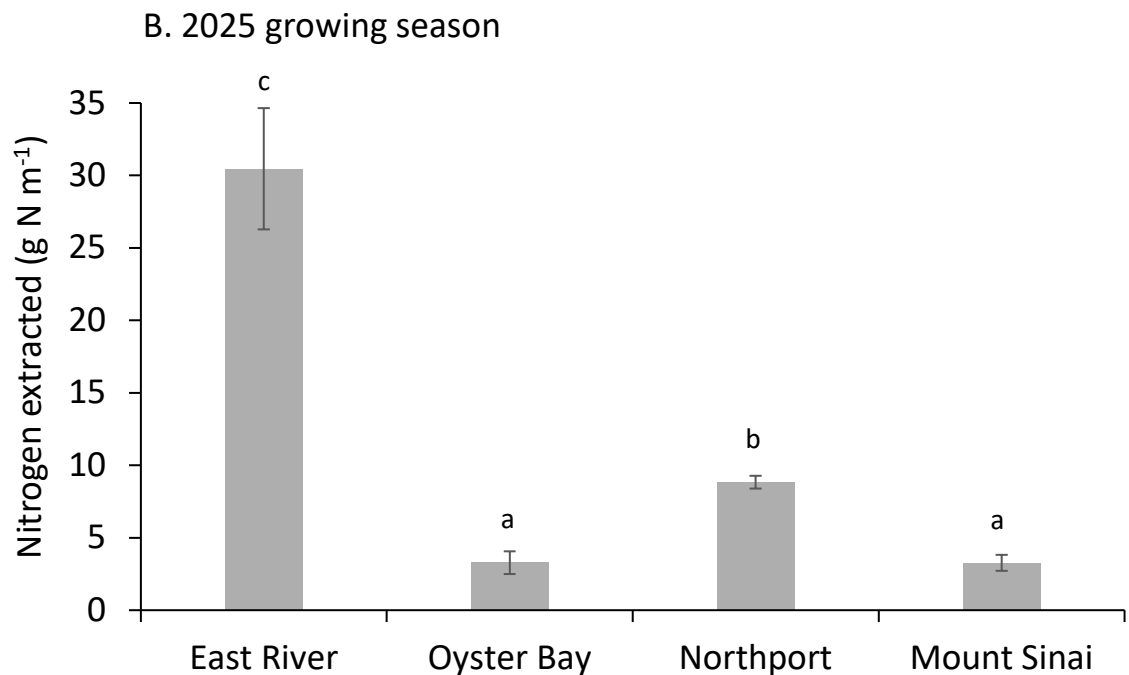
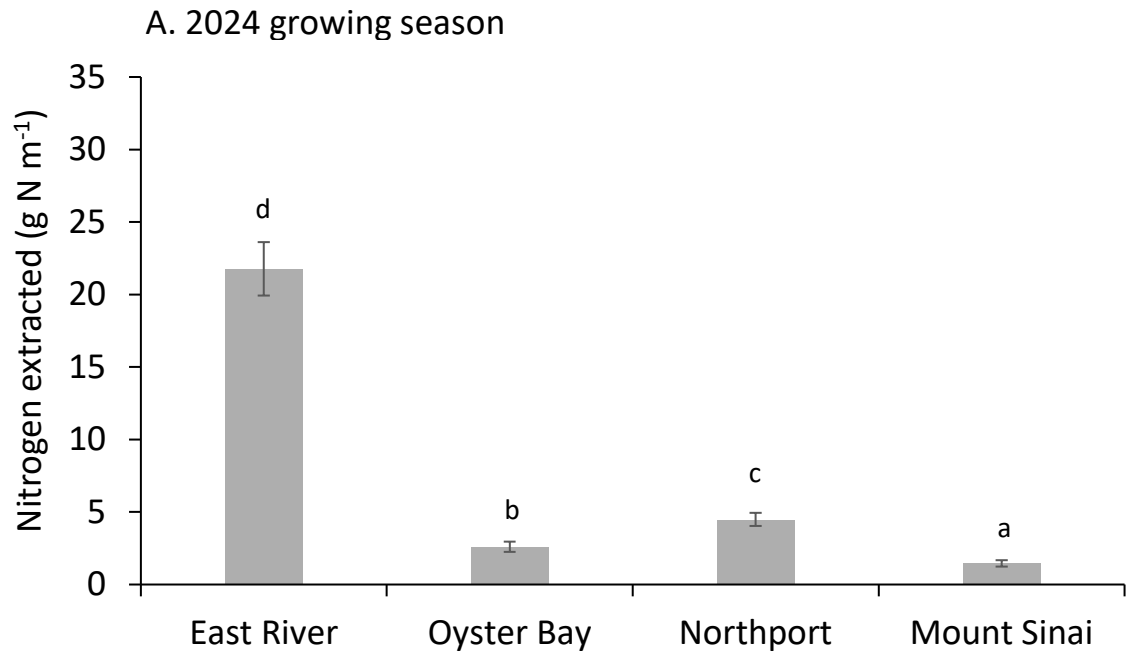


Fig 9. Nitrogen extracted per meter of kelp line at the four study sites in **(A)** 2024 and **(B)** 2025. The nitrogen extracted per meter of line was calculated by multiplying the dry weight kelp biomass per meter of line by the percent nitrogen content in the dry kelp tissue. Kelp lines in Oyster Bay were harvested early on 4/4/25 and most likely did not reach peak biomass. ANOVA's (log₁₀ transformed, $p < 0.001$ and F value=98 for 2024; $p < 0.001$ and F value=41.01 for 2025) were conducted on extracted nitrogen (g N m⁻¹) in 2024 and 2025, letters above bars denote significant differences between treatments (Tukey, $p < 0.05$ for all significant differences in 2024 and $p < 0.01$ for all significant differences in 2025).

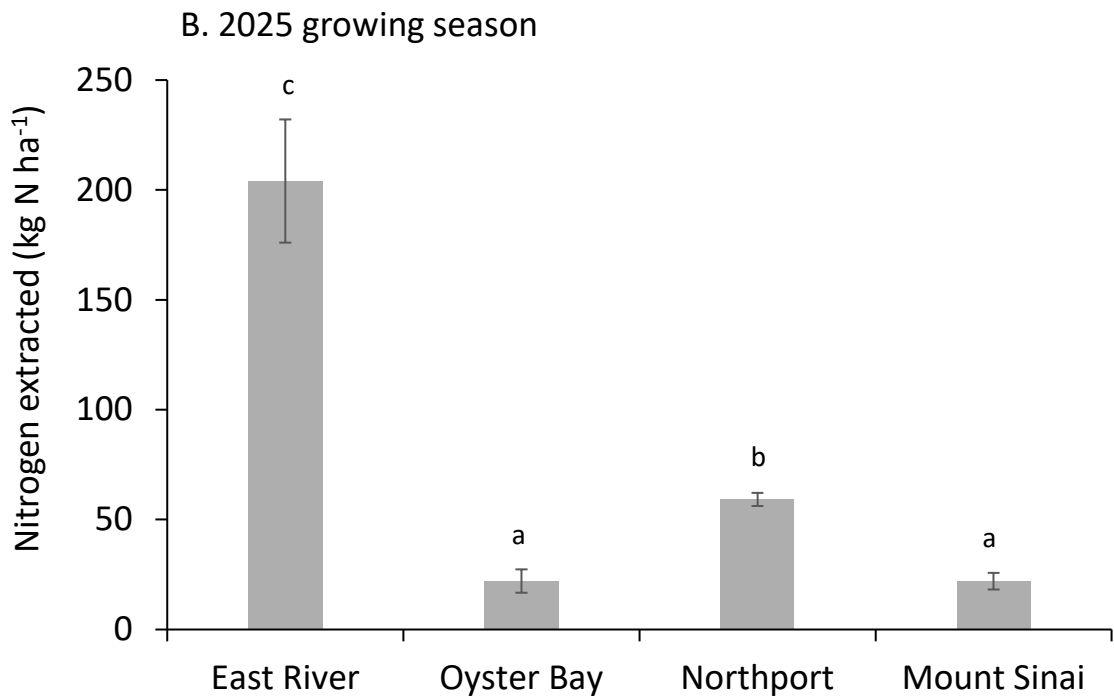
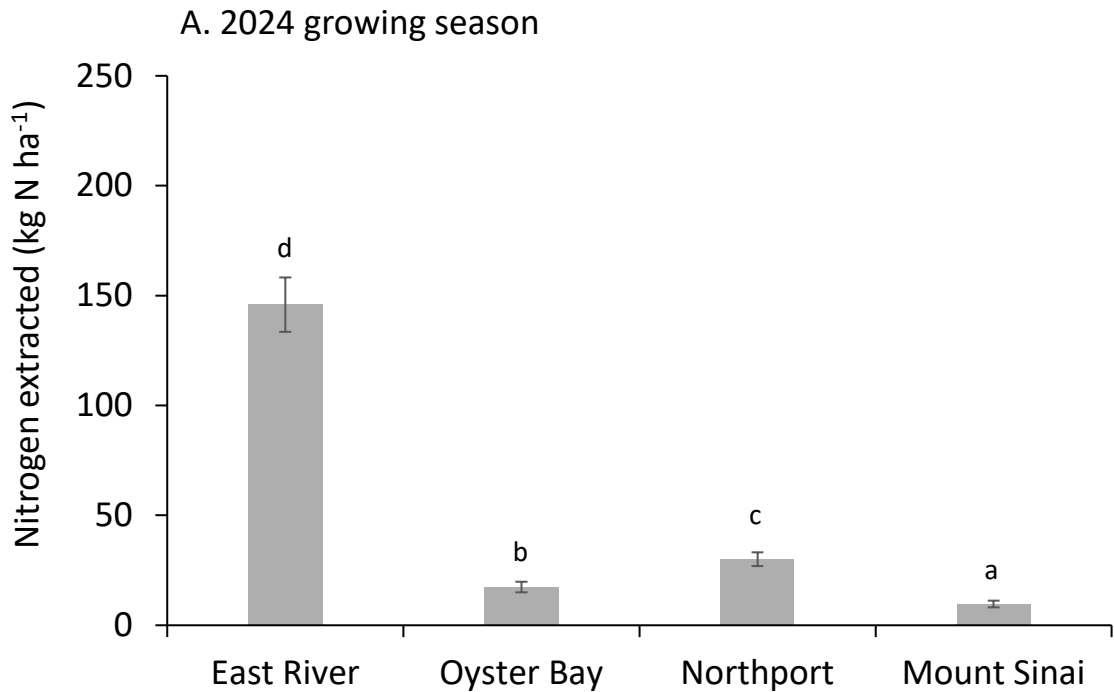


Fig 10. Estimated nitrogen extracted per hectare at peak kelp biomass at the four study sites in **(A)** 2024 and **(B)** 2025. The nitrogen extracted per hectare was extrapolated from the line yields, assuming that a hectare could support 6,700 linear meters of kelp line. Kelp lines in Oyster Bay were harvested early on 4/4/25 and most likely did not reach peak biomass. ANOVA's (log₁₀ transformed, $p < 0.001$ and F value=98 for 2024; $p < 0.001$ and F value=41.01 for 2025) were conducted on extracted nitrogen (g N m^{-2}) in 2024 and 2025, letters above bars denote significant differences between treatments (Tukey, $p < 0.05$ for all significant differences in 2024 and $p < 0.01$ for all significant differences in 2025).

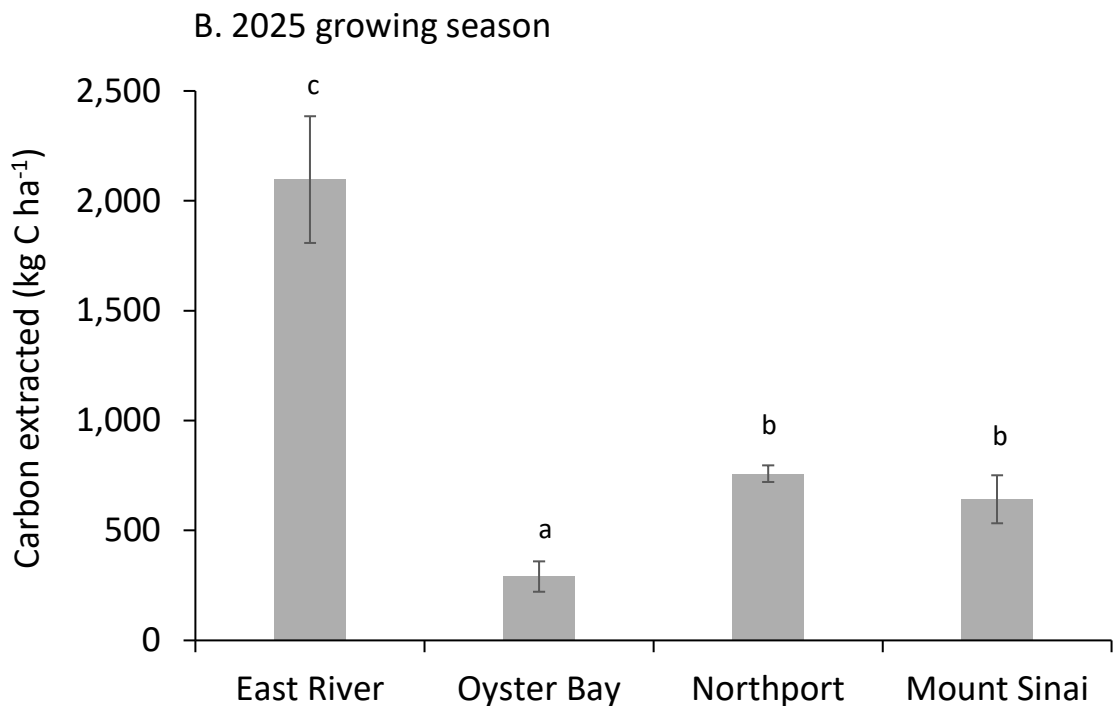
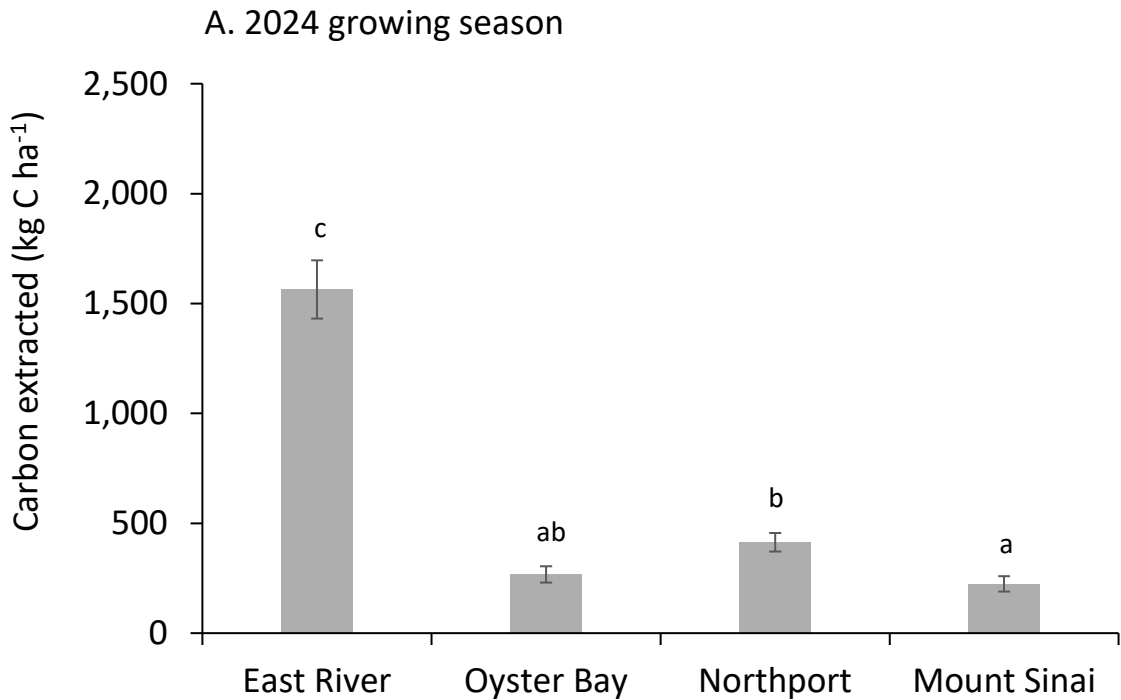


Fig 11. Estimated carbon extracted per hectare at peak kelp biomass at the four study sites in **(A)** 2024 and **(B)** 2025. The carbon extracted per hectare was extrapolated from the line yields, assuming that a hectare could support 6,700 linear meters of kelp line. Kelp lines in Oyster Bay were harvested early on 4/4/25 and most likely did not reach peak biomass. ANOVA's (log₁₀ transformed, $p < 0.001$ and F value=57.97 for 2024; $p < 0.001$ and F value=19.86 for 2025) were conducted on extracted carbon (g N m⁻¹) in 2024 and 2025, letters above bars denote significant differences between treatments (Tukey, $p < 0.05$ for all significant differences in 2024 and 2025).

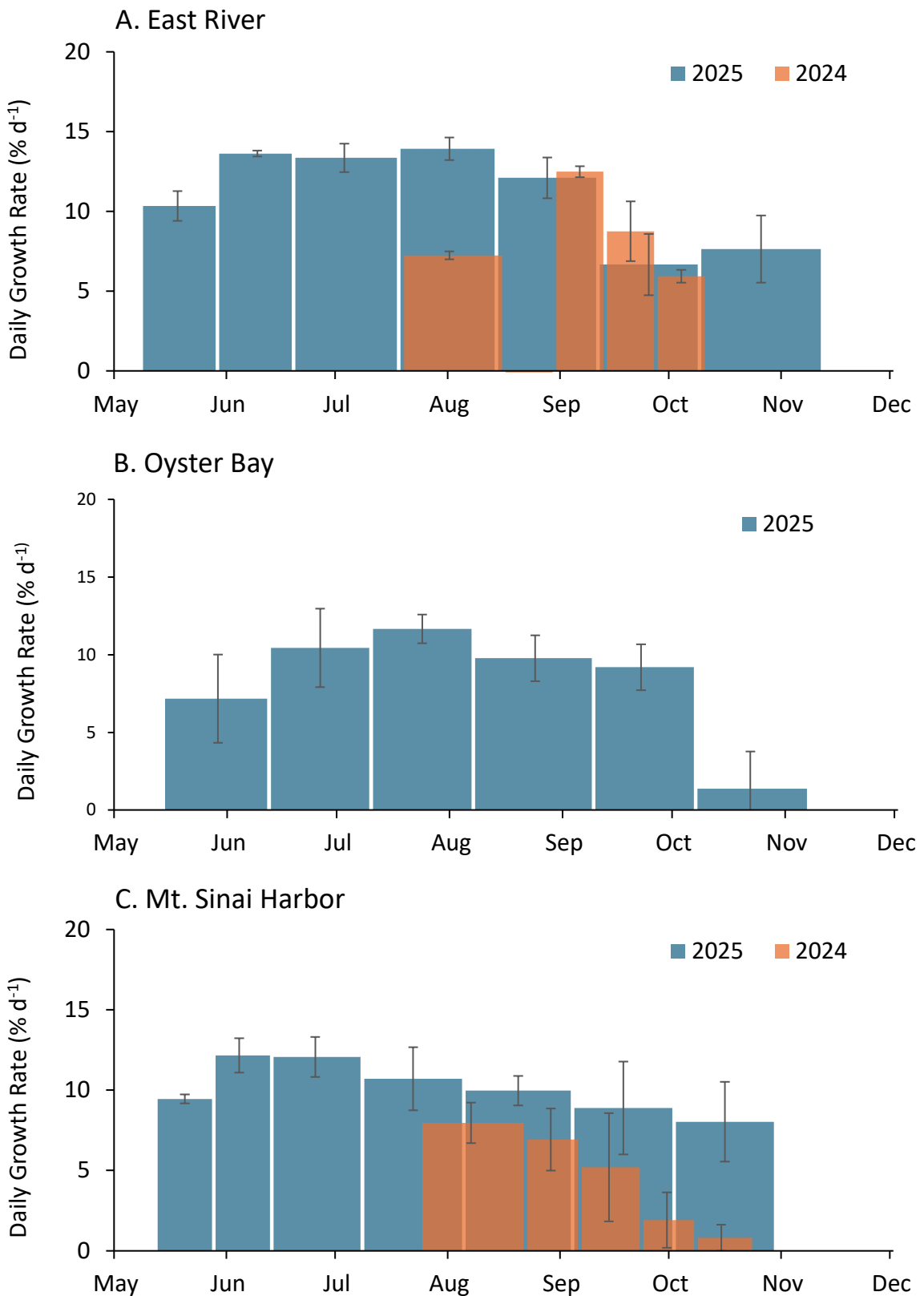


Fig 12. The growth rate (% d⁻¹) of *Ulva* cultivated in floating bags during each cultivation period in the 2024 (orange bars) and 2025 (blue bars) growing seasons, at each of three study sites, including (A) East River, (B) Oyster Bay (2025 only), (C) Mt. Sinai Harbor. *Ulva* biomass was harvested from the bags every two to four weeks, and the width of bars covers the time period over which the daily growth rate was measured.

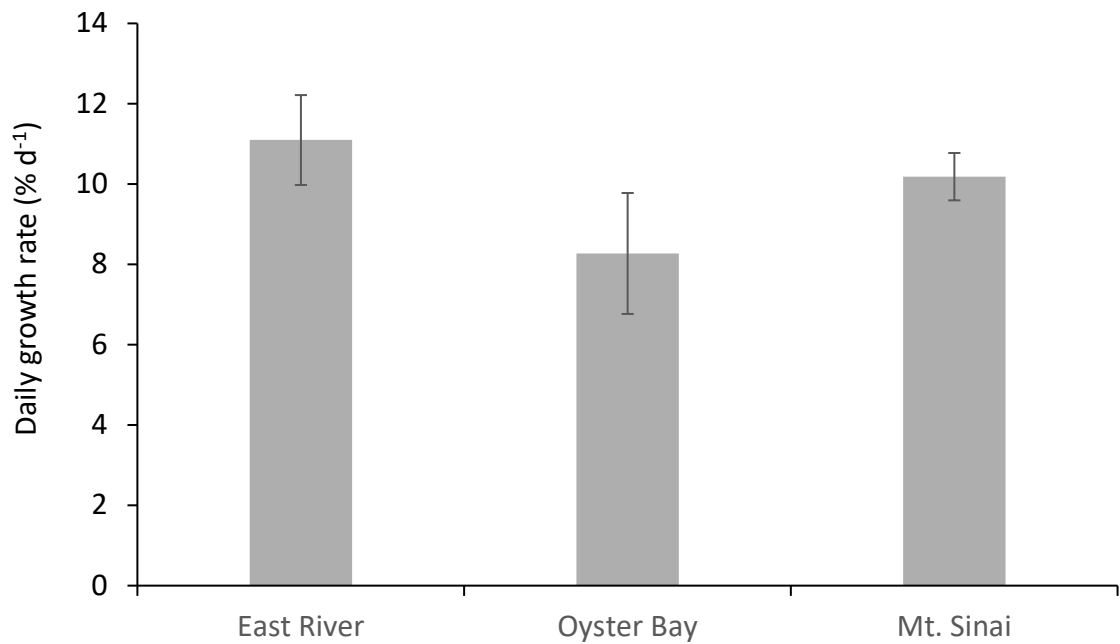


Fig 13. Comparison among study sites of the average daily growth rate of Ulva (% d⁻¹) over the course of the entire 2025 Ulva growing season (May-November). No significant differences were found among the three sites (ANOVA, F=1.66, n.s.).

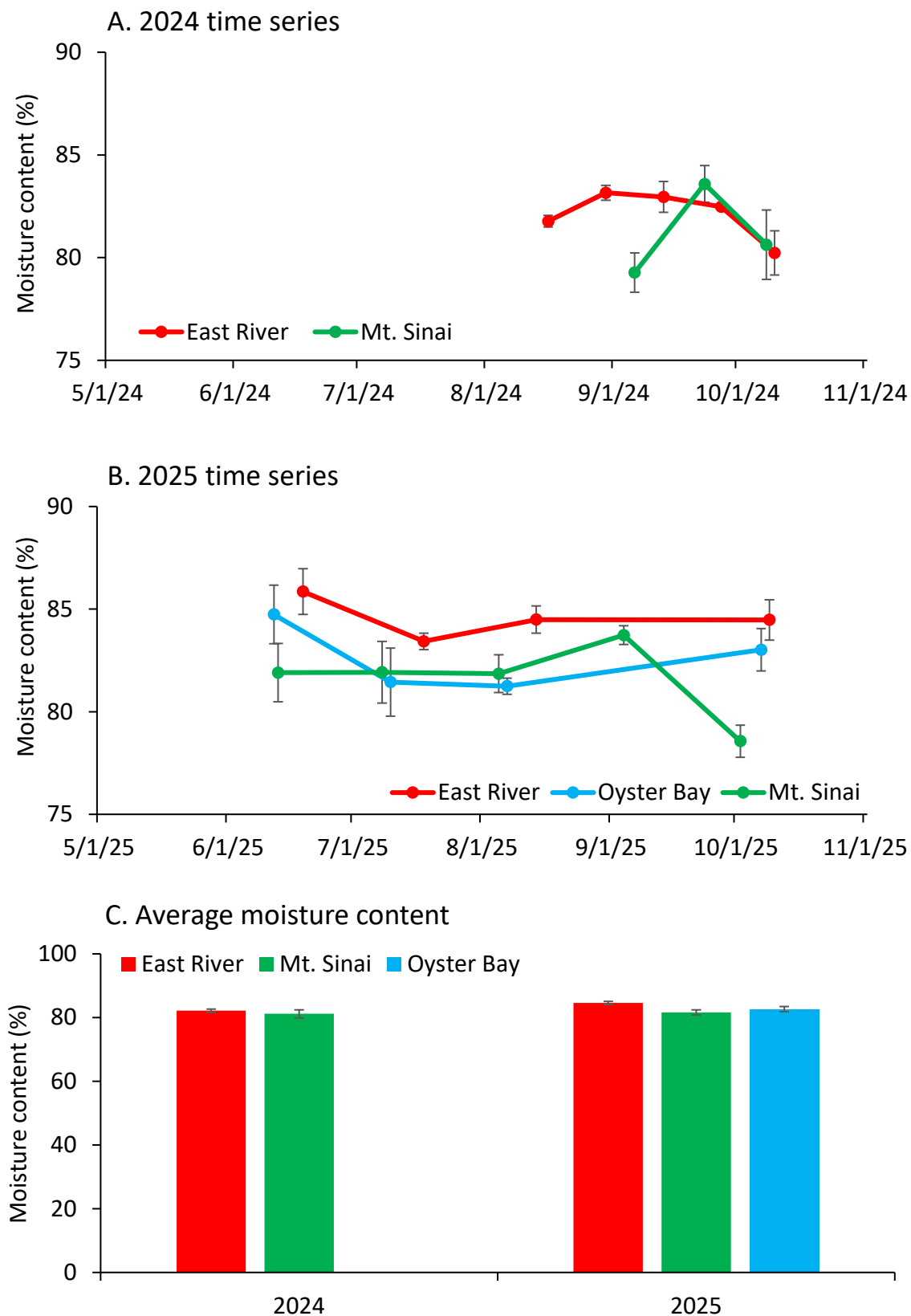


Fig 14. Comparison among sites of the moisture content in *Ulva* tissue over time during the (A) 2024 and (B) 2025 growing seasons, and (C) the average moisture content of *Ulva* tissue at each site during each year. *Ulva* tissue moisture content was measured every two to four weeks when tissue was harvested. No significant differences were found in tissue moisture content among sites in 2024 or 2025 (ANOVAs, n.s.).

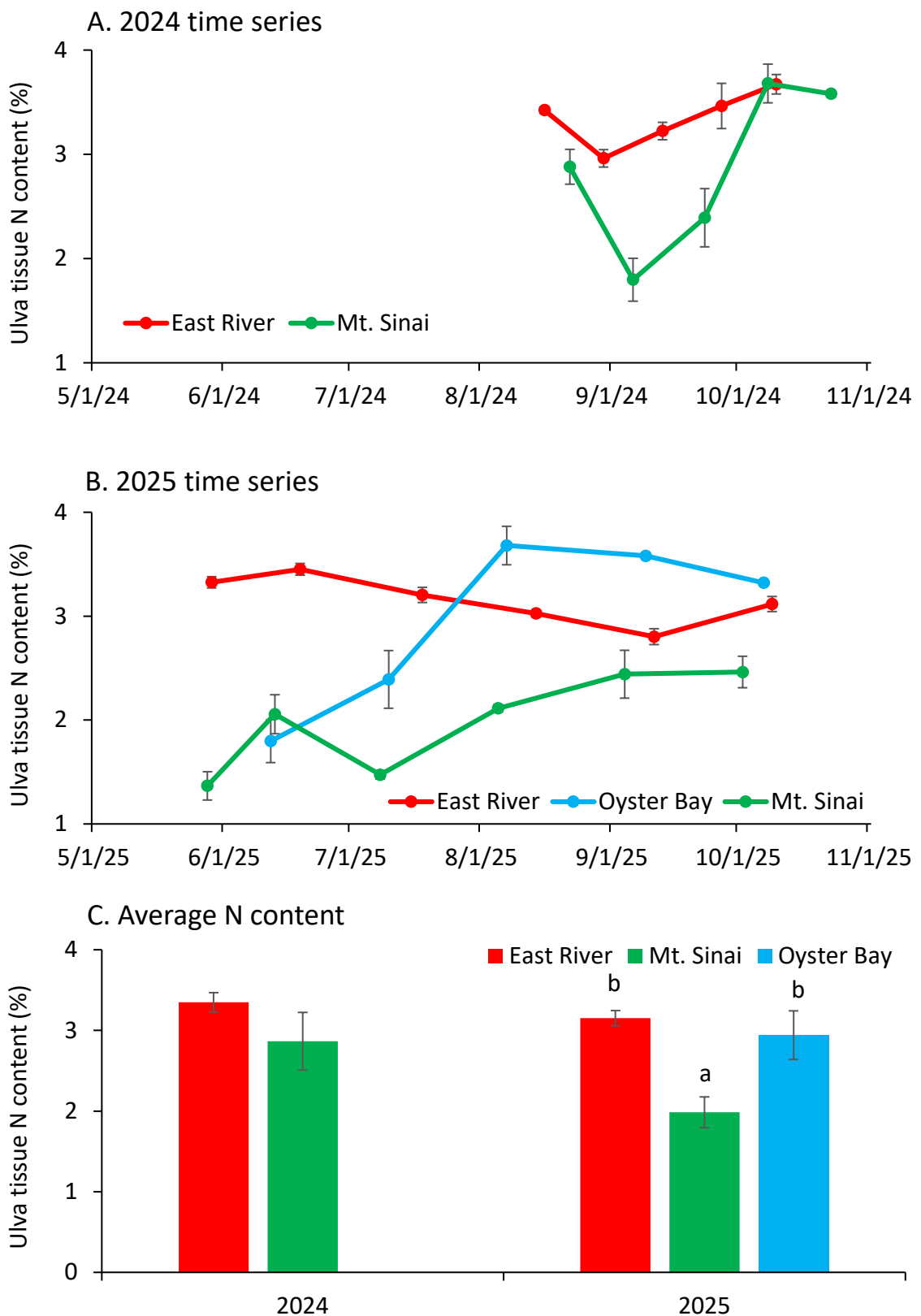


Fig 15. Comparison among sites of the nitrogen content in *Ulva* tissue over time during the (A) 2024 and (B) 2025 growing seasons, and (C) the average nitrogen content at each site during each year. *Ulva* nitrogen content was measured every two to four weeks when tissue was harvested. ANOVA's ($f=1.65$, n.s. for 2024; $f=8.555$, $p<0.01$ for 2025) were conducted on moisture content in 2024 and 2025. Letters above bars denote significant differences between treatments (Tukey, $p<0.05$ for all significant differences).

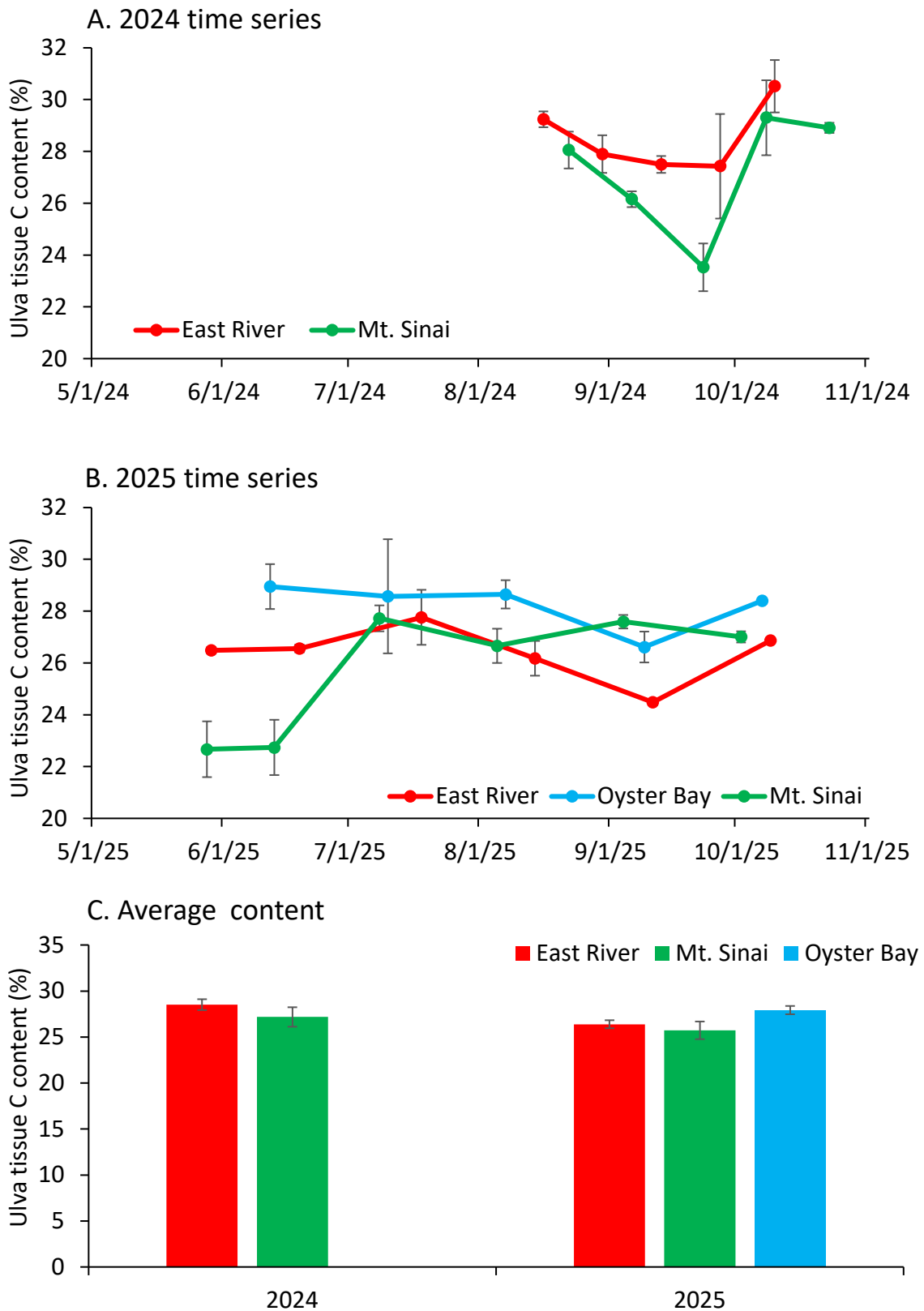


Fig 16. Comparison among sites of the carbon content in *Ulva* tissue over time during the **(A)** 2024 and **(B)** 2025 growing seasons, and **(C)** the average carbon content at each site during each year. *Ulva* carbon content was measured every two to four weeks when tissue was harvested. No significant differences were found in *Ulva* tissue carbon content among sites in 2024 or 2025 (ANOVAs, n.s.).

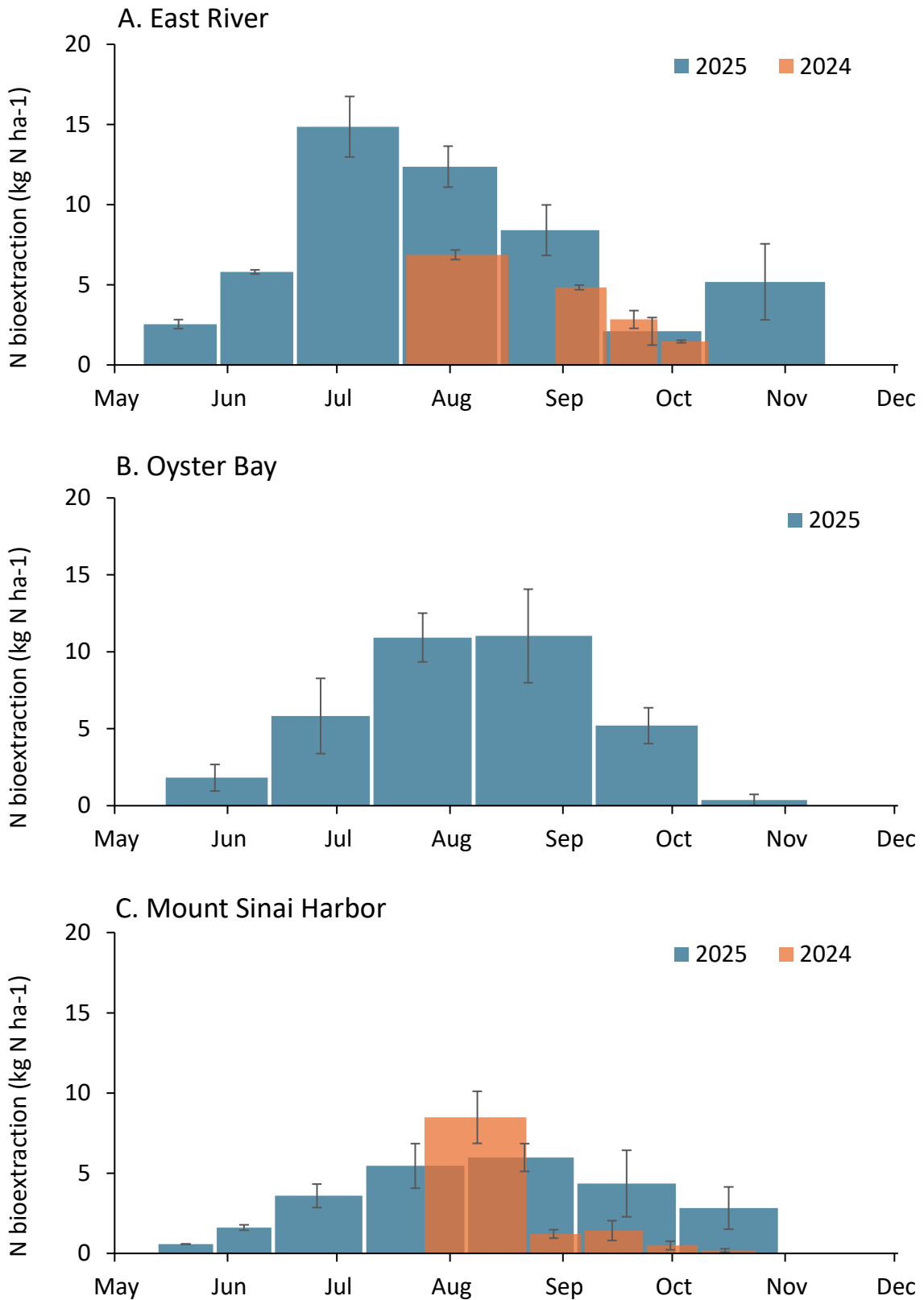


Fig 17. Nitrogen extracted per hectare by *Ulva* during the 2024 (orange bars) and 2025 (blue bars) growing seasons, at (A) East River, (B) Oyster Bay (2025 only), (C) Mt. Sinai Harbor. *Ulva* biomass was harvested from experimental bags every two to four weeks. The width of bars along the x-axes encompasses the time period of each cultivation round. Nitrogen extracted from the experimental bags was extrapolated to a per ha basis assuming a farm design of 4000 bags per ha.

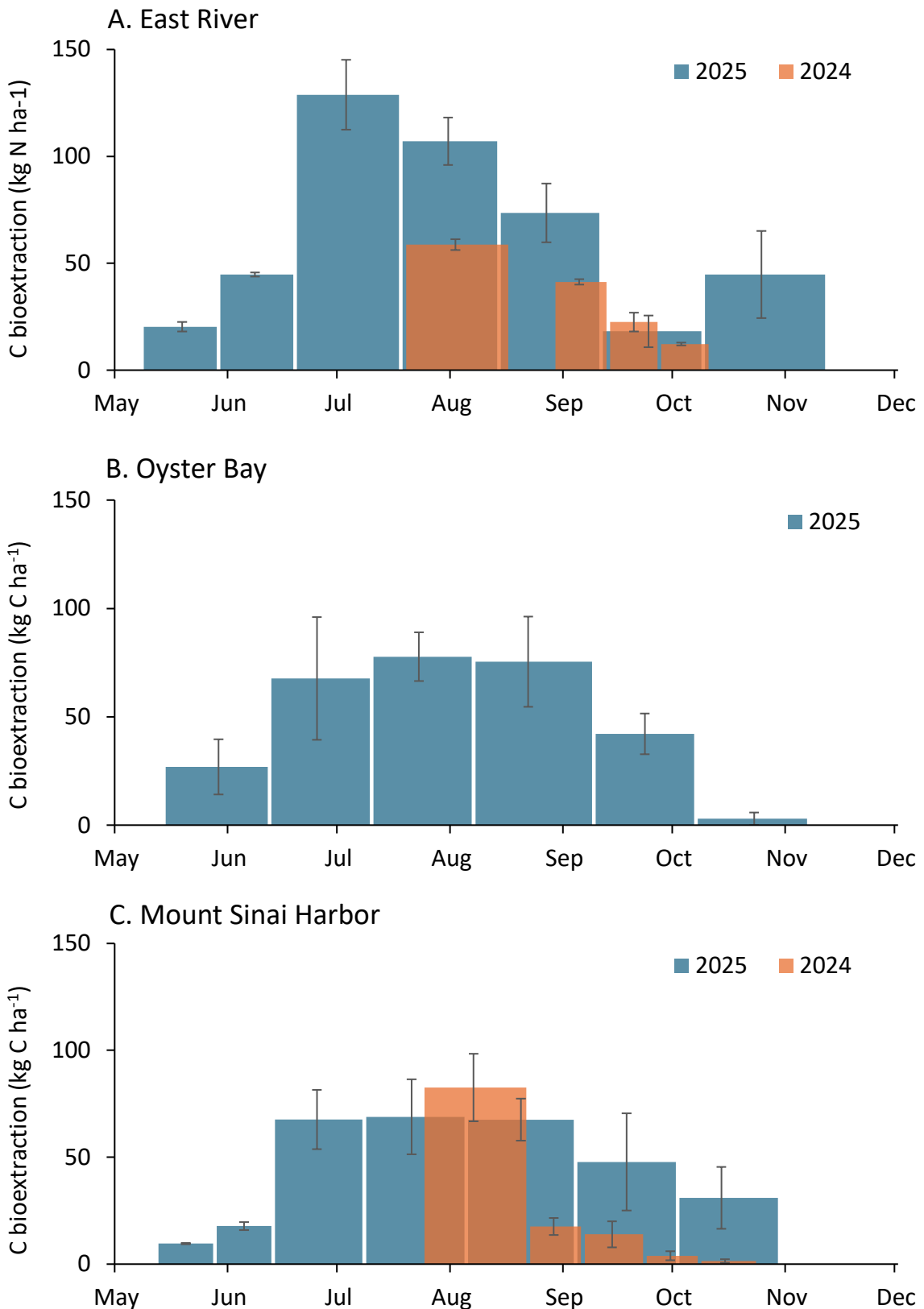


Fig 18. Carbon extracted per hectare by *Ulva* during the 2024 (orange bars) and 2025 (blue bars) growing seasons, at (A) East River, (B) Oyster Bay (2025 only), (C) Mt. Sinai Harbor. *Ulva* biomass was harvested from experimental bags every two to four weeks. The width of bars along the x-axes encompasses the time period of each cultivation round. Carbon extracted from the experimental bags was extrapolated to a per ha basis assuming a farm design of 4000 bags per ha.

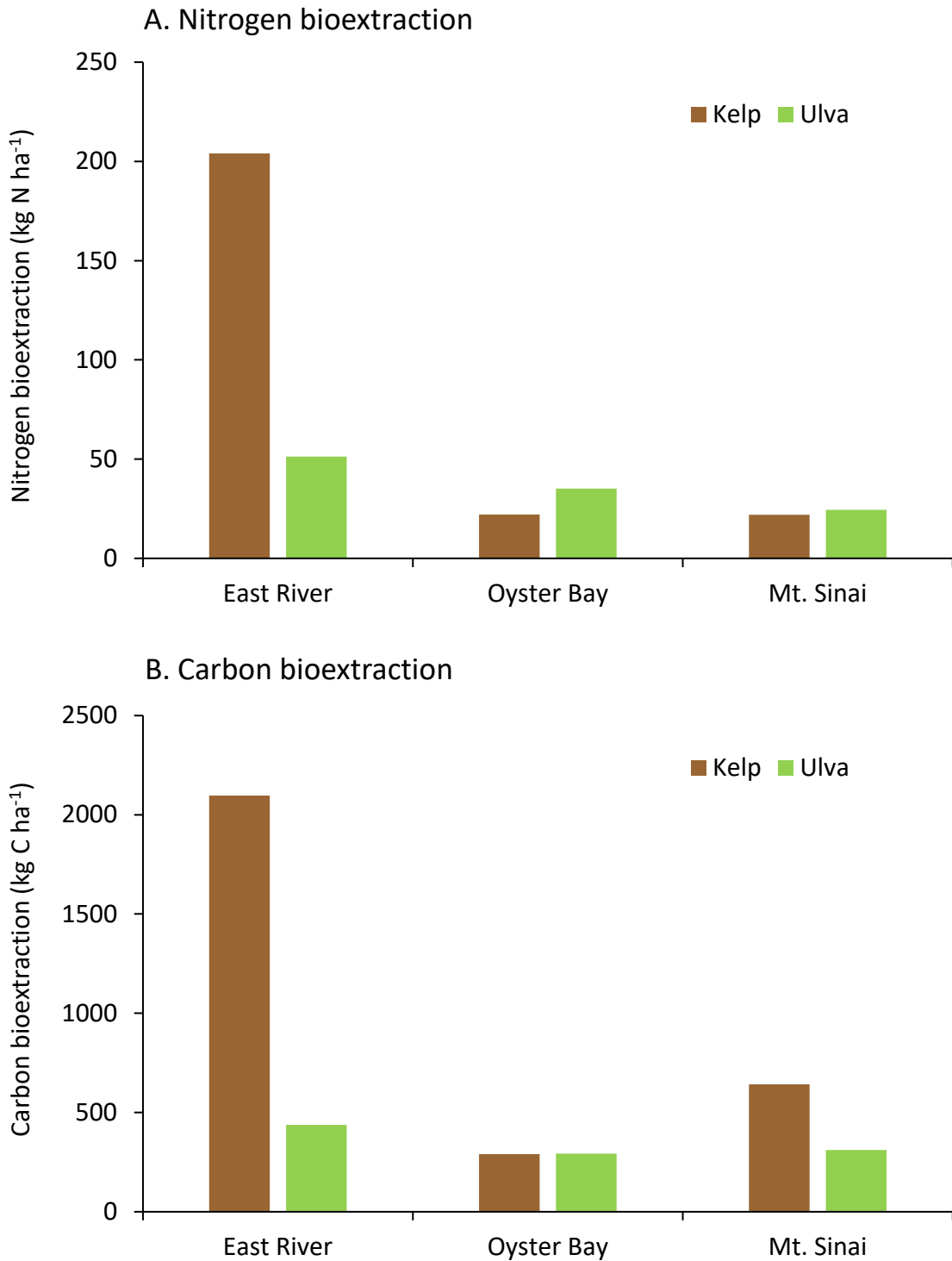


Fig 19. Comparison of the annual (A) nitrogen and (B) carbon extracted per hectare by Ulva versus kelp at three sites (East River, Oyster Bay, and Mt. Sinai Harbor), based on seaweed growth and CN data collected during 2025. Kelp was cultivated from December 2024 to May 2025 with CN bioextraction calculated from a single harvest at peak biomass, with the exception of Oyster Bay where kelp was harvested in April prior to peak biomass. Ulva was cultivated from May to November 2025, with Ulva biomass harvested every three to four weeks, and annual CN bioextraction calculated by summing the CN extractions from each harvest. Farm design assumptions include 6,700 linear meters of kelp line per hectare, and 4,000 floating bags of Ulva per hectare.

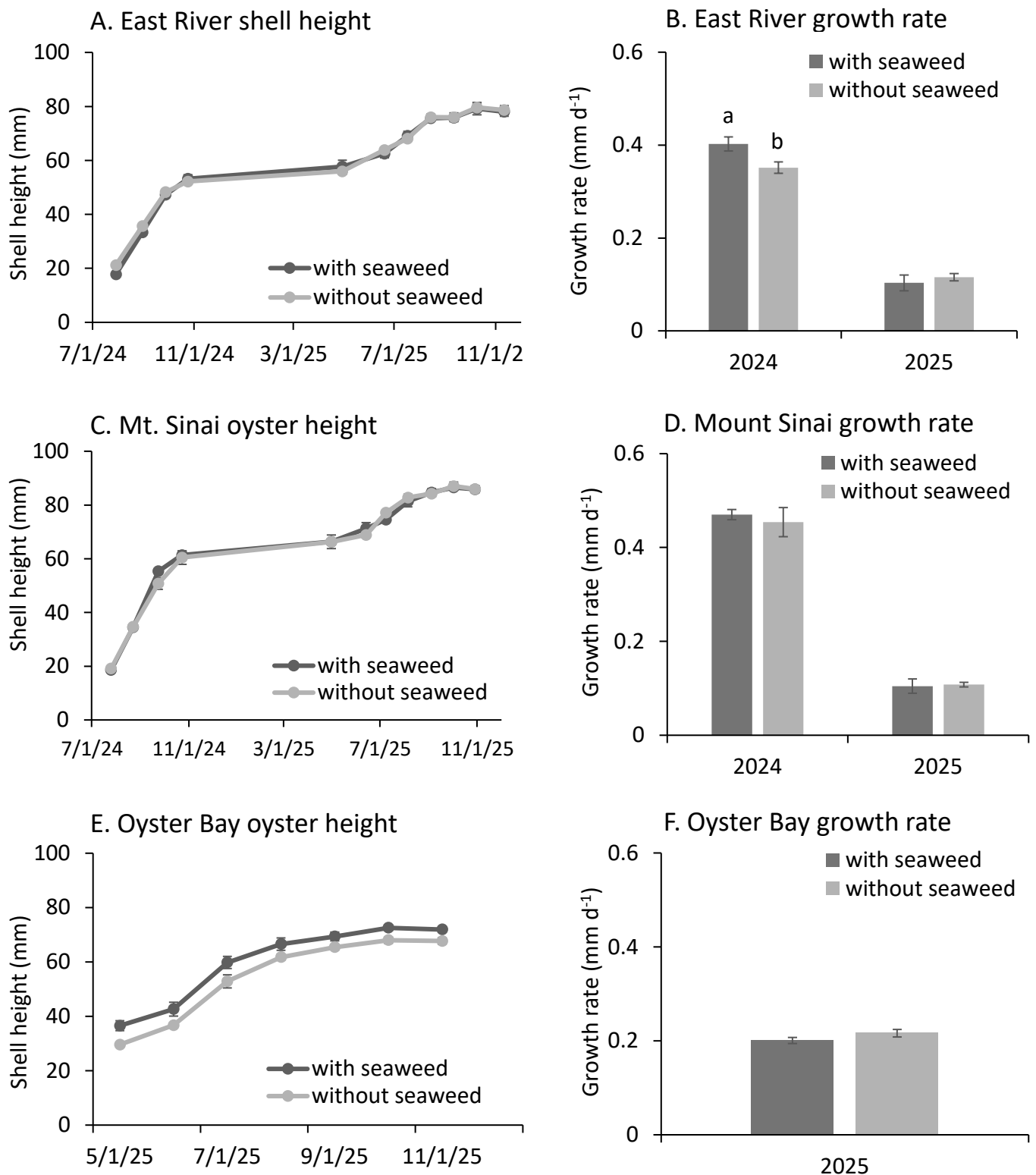


Fig 20. Comparison of shell growth between oysters cultivated with and without seaweeds among three sites including (A-B) East River, (C-D) Oyster Bay, and (E-F) Mt. Sinai Harbor. Panels on the left (A, C, E) compare shell height over time. Panels on the right (B, D, F) compare growth rates in each growing season. In East River and Mt. Sinai the same cohort of oysters were grown over ~15-month period that encompassed two growing seasons (2024, 2025). In Oyster Bay, a separate cohort was cultivated during the 2025 growing season (May-Nov.). Oysters cultivated with *Ulva* grew significantly faster than oysters cultivated without *Ulva* in the East River in 2024 (t-test, $p < 0.05$). No significant differences were found, however, within other sites and years.

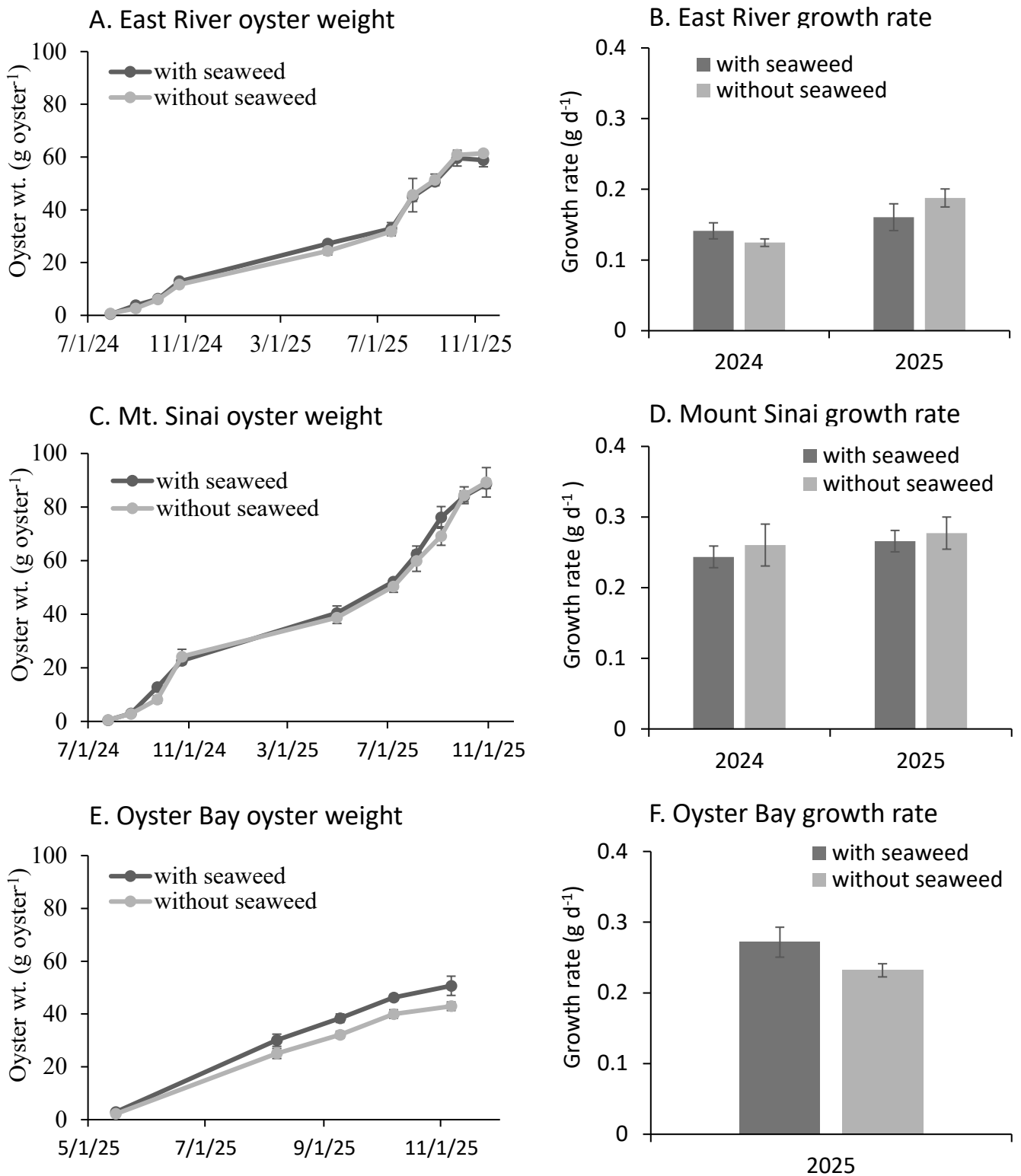
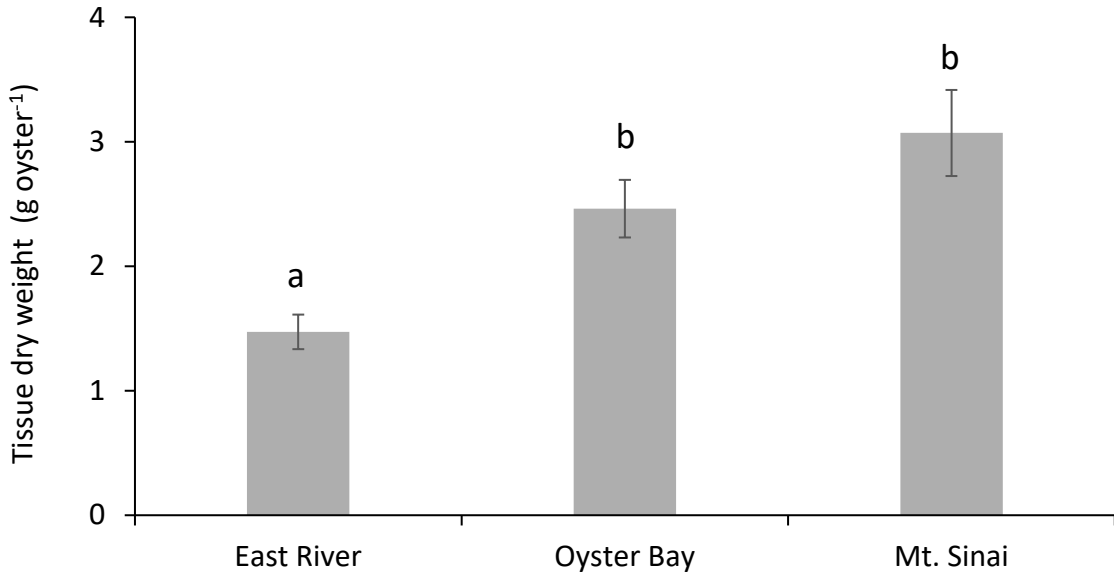


Fig 21. Comparison of weight-based growth (i.e. total oyster weight) between oysters cultivated with and without seaweeds among three sites including (A-B) East River, (C-D) Mount Sinai Harbor, and (E-F) Oyster Bay. Panels on the left (A, C, E) compare oyster weight over time. Panels on the right (B, D, F) compare growth rates in each growing season. In East River and Mount Sinai Harbor the same cohort of oysters were grown over ~15-month period that encompassed two growing seasons (2024, 2025). In Oyster Bay, a separate cohort was cultivated during the 2025 growing season (May-Nov.). Within each site and year, no significant differences were found in weight-based growth rates between oysters grown with and without seaweeds (t-tests, n.s.).

A. Oyster tissue dry weight



B. Oyster shell dry weight

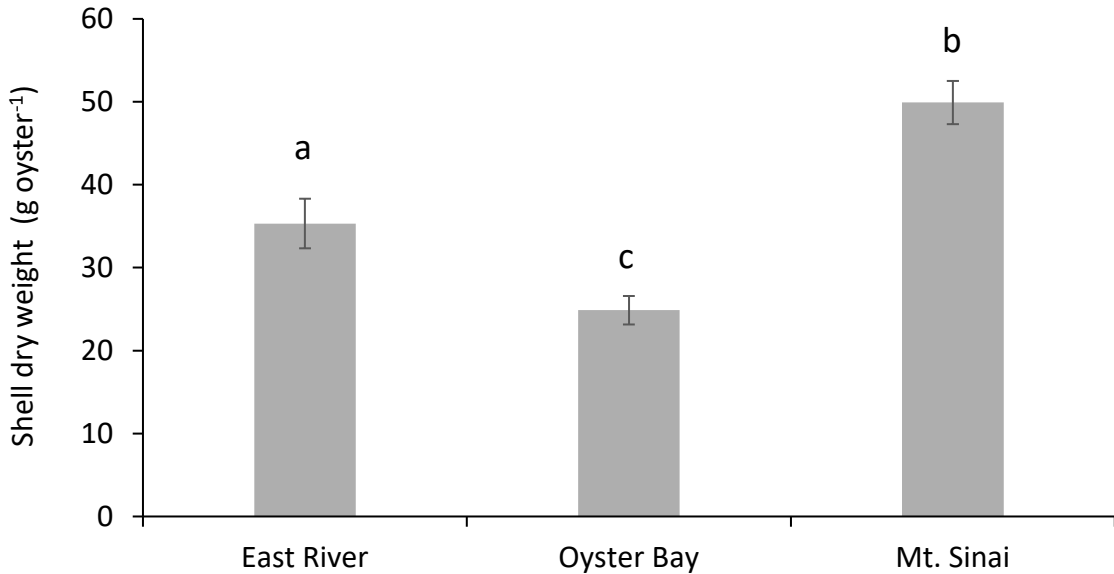


Fig 22. Comparison among sites of oyster (A) dry tissue and (B) dry shell weights at final harvest (late October-early November 2025), for oysters cultivated with seaweeds. The same cohort of oysters was cultivated in the East River and Mt. Sinai Harbor for ~15 months from July 2024 to November 2025. A separate cohort of oysters was cultivated in Oyster Bay for the 2025 growing season, for ~6 months from May through October. ANOVA's were conducted ($p < 0.001$ and $F \text{ value} = 168.8$ for dry tissue; $p < 0.001$ and $F \text{ value} = 25.6$ for dry shell), and letters above bars denote significant differences between sites (Tukey, $p < 0.05$ for all significant interactions).

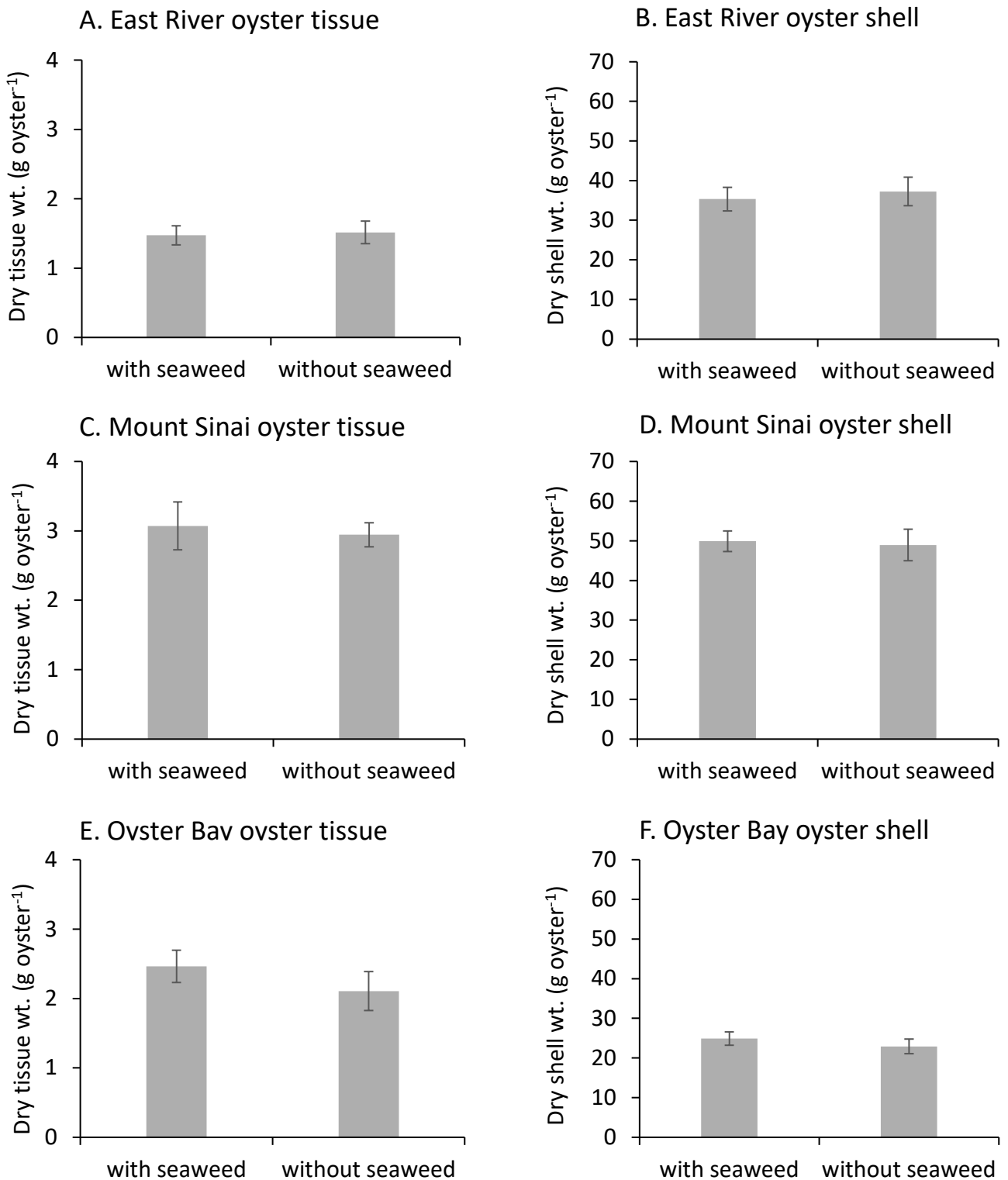


Fig 23. Comparison of oyster tissue and oyster shell dry weights at final harvest between oysters cultivated with and without seaweeds at three sites including (A-B) East River, (C-D) Mount Sinai Harbor, and (E) Oyster Bay. Panels on the left (A, C, E) compare oyster tissue dry weights. Panels on the right (B, D, F) compare oyster shell dry weights. The same cohort of oysters was cultivated in the East River and Mount Sinai Harbor for ~15 months from July 2024 to November 2025. A separate cohort of oysters was cultivated in Oyster Bay for the 2025 growing season, for ~6 months from May through October. Within each site, no significant differences were found in either tissue or shell dry weight between oysters grown with and without seaweeds (t-tests, n.s.).

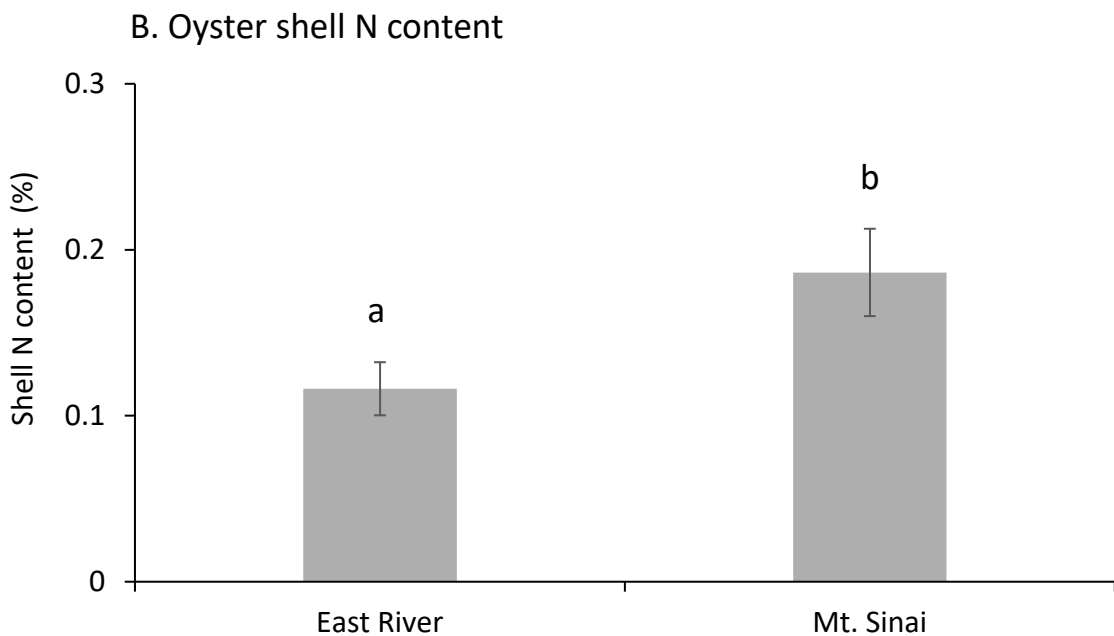
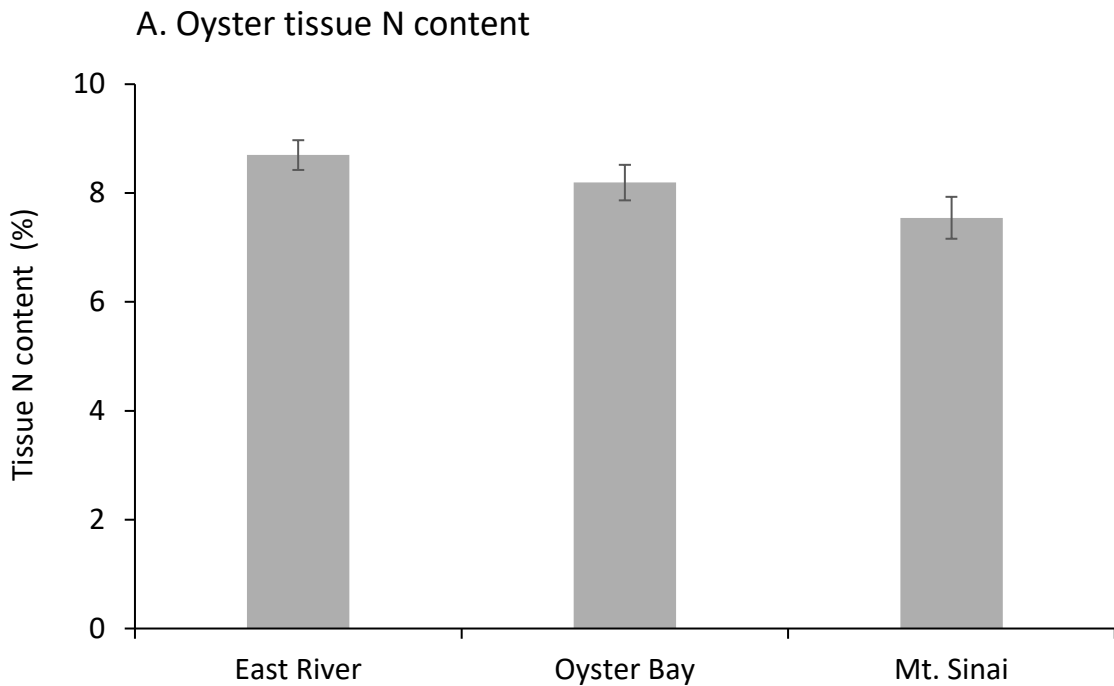


Fig 24. Comparison among sites of oyster (A) tissue nitrogen content and (B) shell nitrogen content, for oysters cultivated with seaweeds. No significant differences were found among sites in tissue N content (ANOVA, $F=3.05$, n.s.). Shell nitrogen content was significantly higher in oysters cultivated in Mount Sinai than the East River (t-test, $t=1.73$, $p<0.05$).

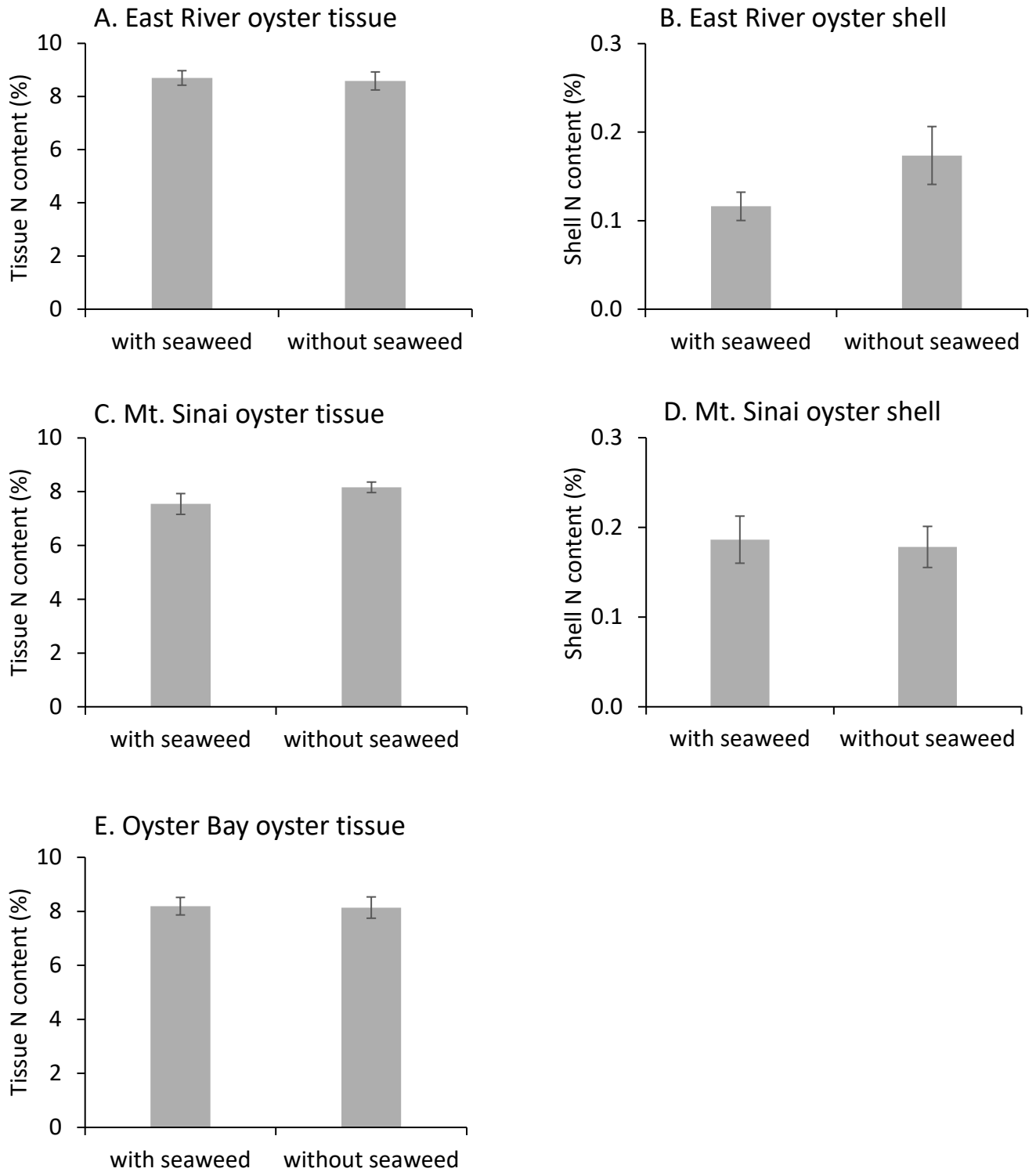


Fig 25. Comparison of oyster tissue and oyster shell nitrogen content between oysters cultivated with and without seaweeds at three sites including (A-B) East River, (C-D) Mt. Sinai Harbor, and (E) Oyster Bay. Panels on the left (A, C, E) compare oyster tissue nitrogen content. Panels on the right (B, D) compare oyster shell nitrogen content. Within each site, no significant differences were found in either tissue or shell nitrogen content between oysters grown with and without seaweeds (t-tests, n.s.).

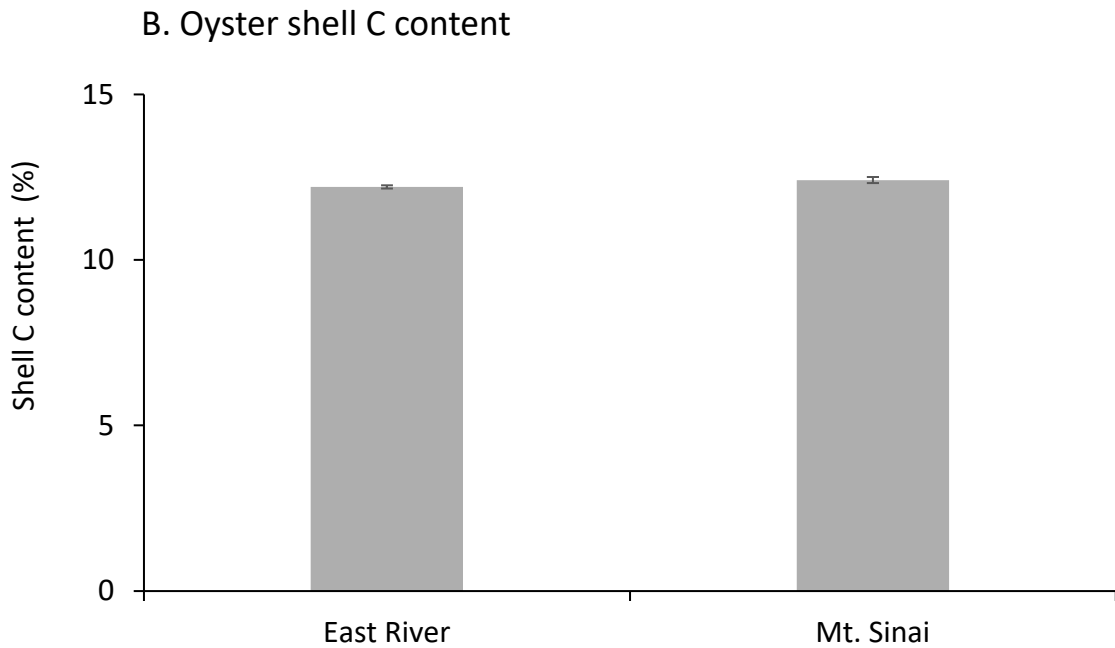
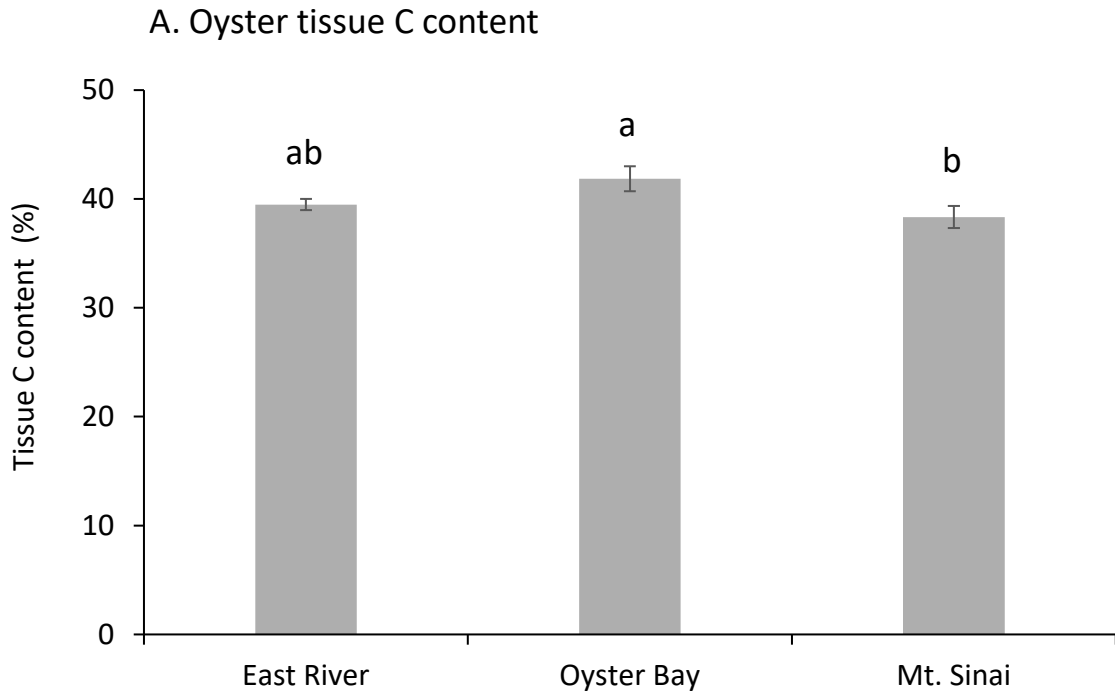


Fig 26. Comparison among sites of oyster (A) tissue carbon content and (B) shell carbon content, for oysters cultivated with seaweeds. Tissue carbon content was significantly higher in oysters cultivated in Oyster Bay than Mt. Sinai Harbor, with no other differences between sites (ANOVA with post-hoc Tukey test, $F=3.74$, $p<0.05$). No significant differences were found among sites in shell carbon content (t-test, n.s.).

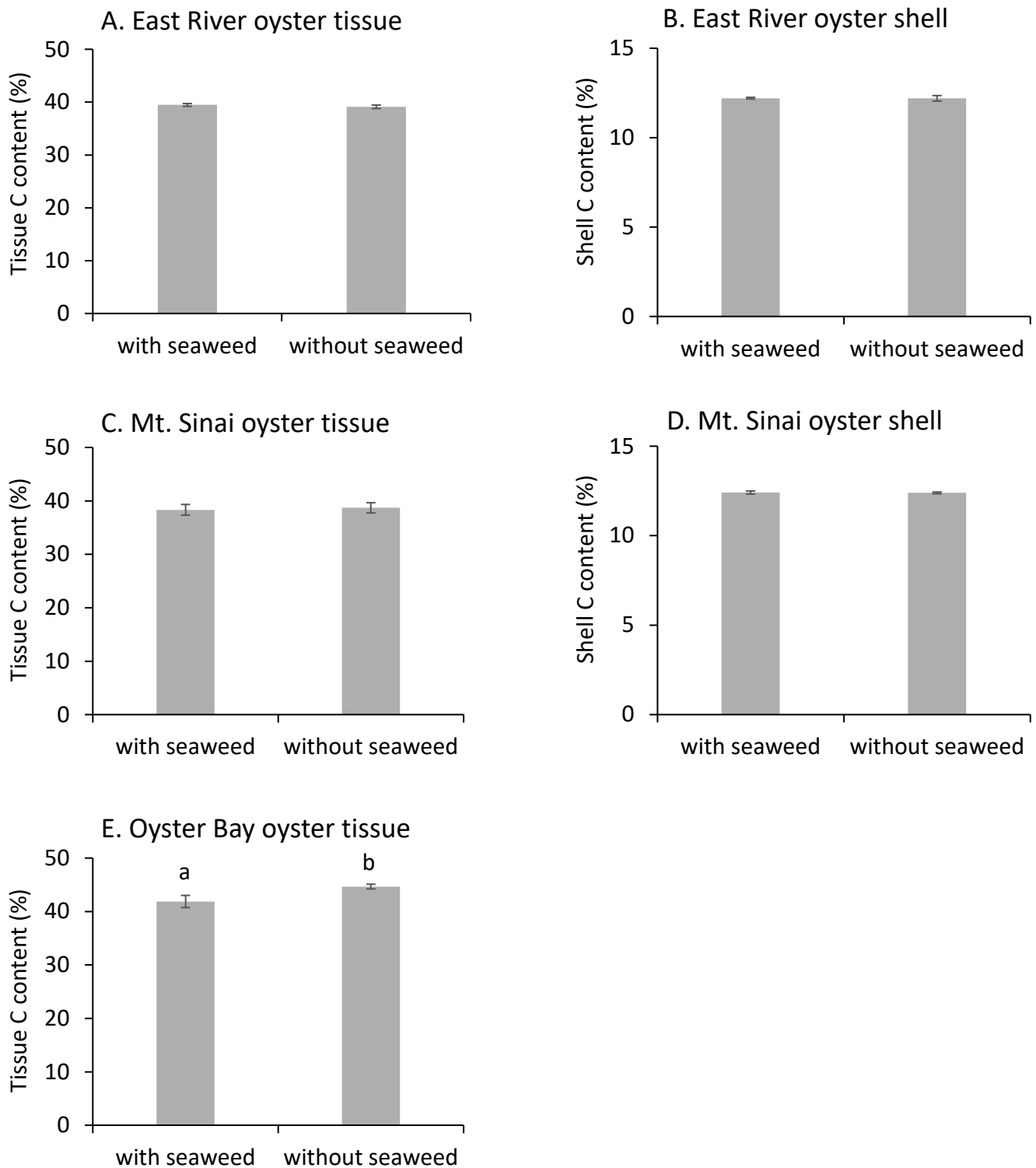


Fig 27. Comparison of oyster tissue and oyster shell carbon content between oysters cultivated with and without seaweeds at three sites including (A-B) East River, (C-D) Mt. Sinai Harbor, and (E) Oyster Bay. Panels on the left (A, C, E) compare oyster tissue carbon content. Panels on the right (B, D) compare oyster shell carbon content. Within each site, no significant differences were found in either tissue or shell carbon content between oysters grown with and without seaweeds, with the only exception of tissue carbon in Oyster Bay where oysters grown without seaweed had significantly higher tissue carbon content (t-tests, $p < 0.05$ for significant difference).

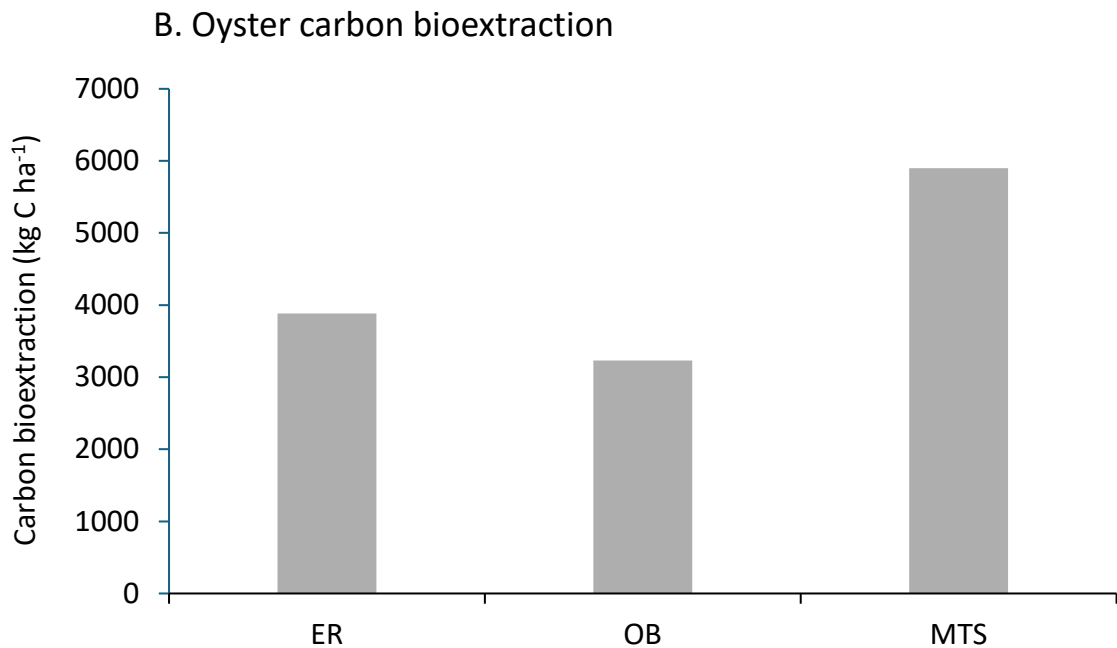
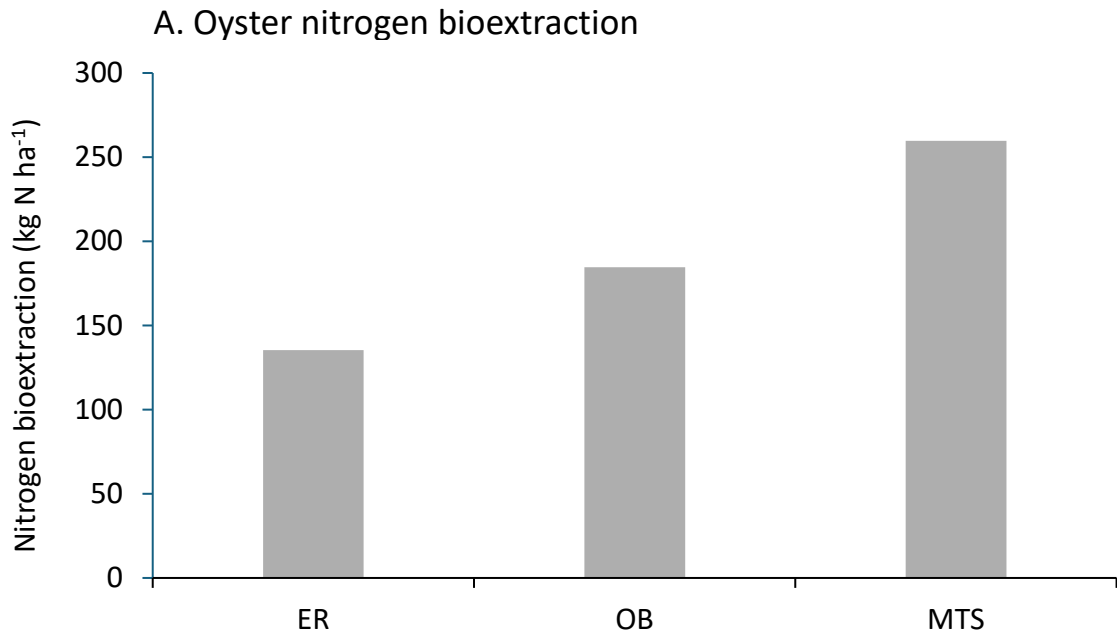


Fig 28. Comparison among sites of estimated annual oyster (A) nitrogen bioextraction and (B) carbon bioextraction, for oysters cultivated with seaweeds. The same cohort of oysters was cultivated in the East River and Mt. Sinai Harbor for ~15 months from July 2024 to November 2025. A separate cohort of oysters was cultivated in Oyster Bay for the 2025 growing season, for ~6 months from May through October. The average CN content per oyster at harvest at each site was calculated based on the tissue and shell mass and CN content data collected. CN bioextraction was extrapolated to a per hectare basis by assuming 800,000 oysters harvested per hectare per year (i.e. 200 oysters from each of 4,000 floating bags).

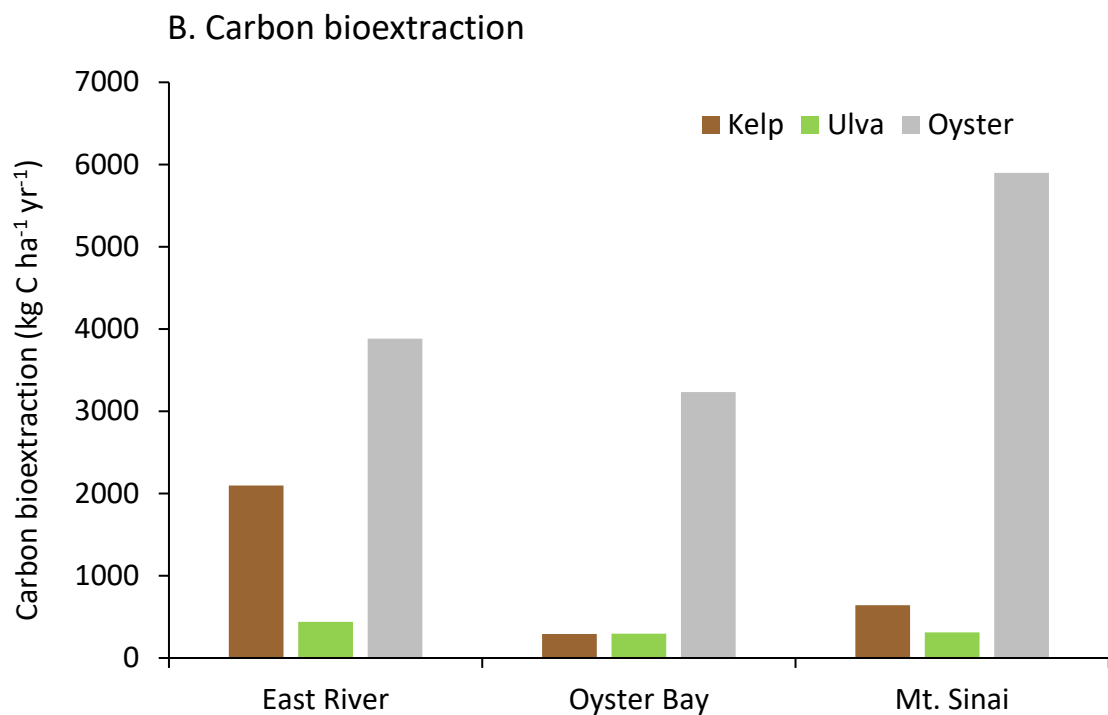
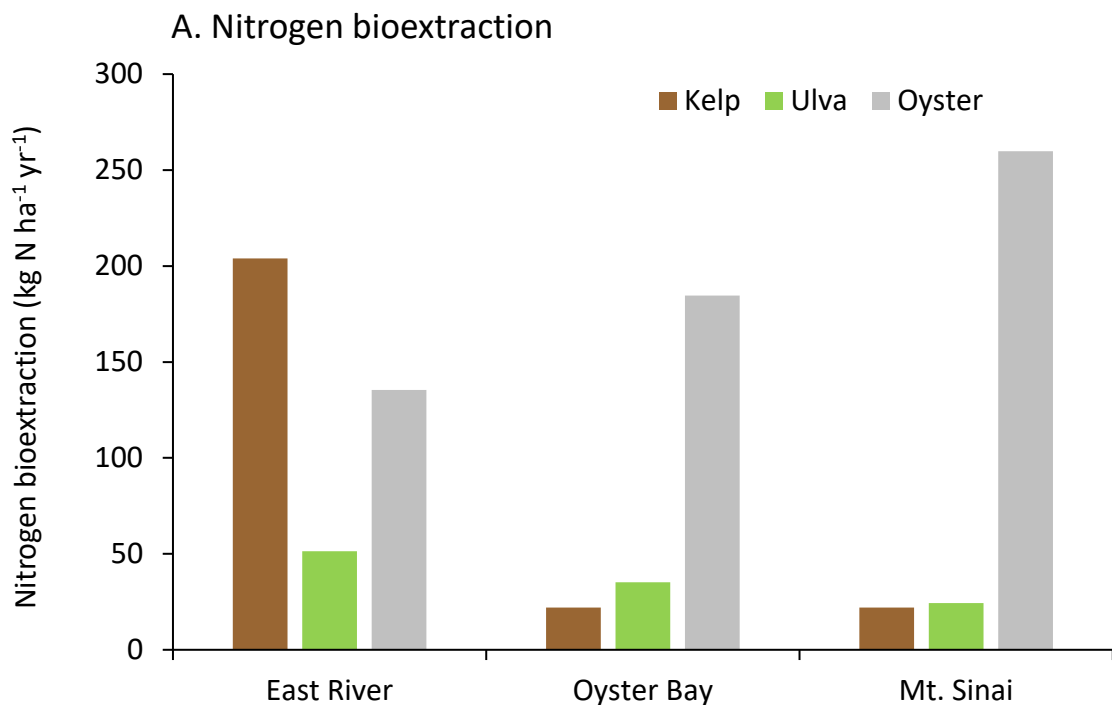


Fig 29. Estimated annual (A) nitrogen and (B) carbon extracted per hectare through Ulva, kelp, and oyster cultivation at three sites (East River, Oyster Bay, and Mt. Sinai Harbor). Estimates are based on seaweed and oyster growth and CN data collected in this study and extrapolated to a per hectare basis under the following assumptions: 6,700 linear meters of kelp line per hectare, and 4,000 floating bags of Ulva and oyster in co-cultivation per hectare, with 200 oysters harvested per bag per year (800,000 oysters per year).

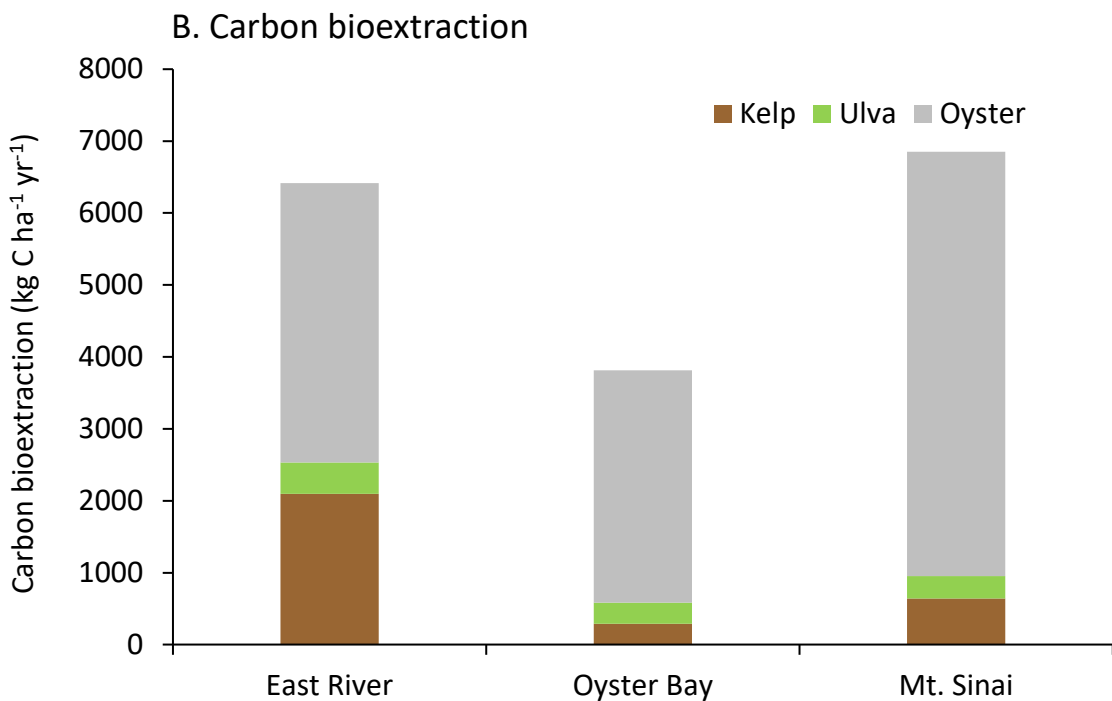
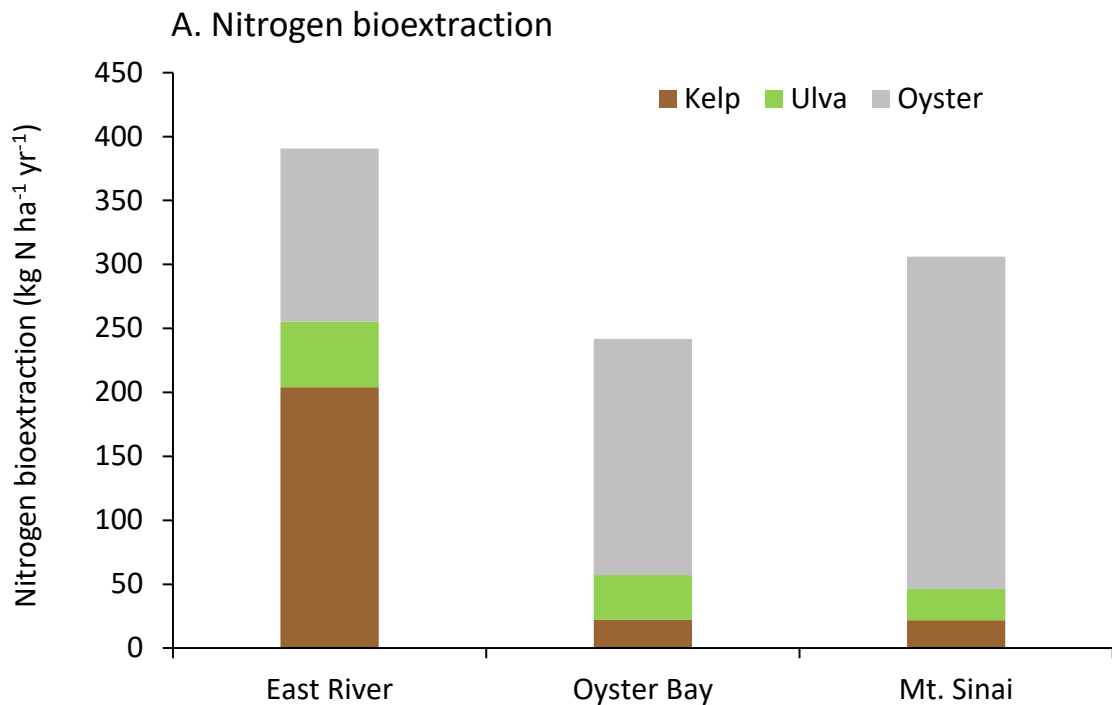


Fig 30. Estimated annual (A) nitrogen and (B) carbon extracted per hectare through Ulva, kelp, and oyster cultivation at three sites (East River, Oyster Bay, and Mt. Sinai Harbor). Estimates are based on seaweed and oyster growth and CN data collected in this study and extrapolated to a per hectare basis under the following assumptions: 6,700 linear meters of kelp line per hectare, and 4,000 floating bags of Ulva and oyster in co-cultivation per hectare, with 200 oysters harvested per bag per year (800,000 oysters per year).

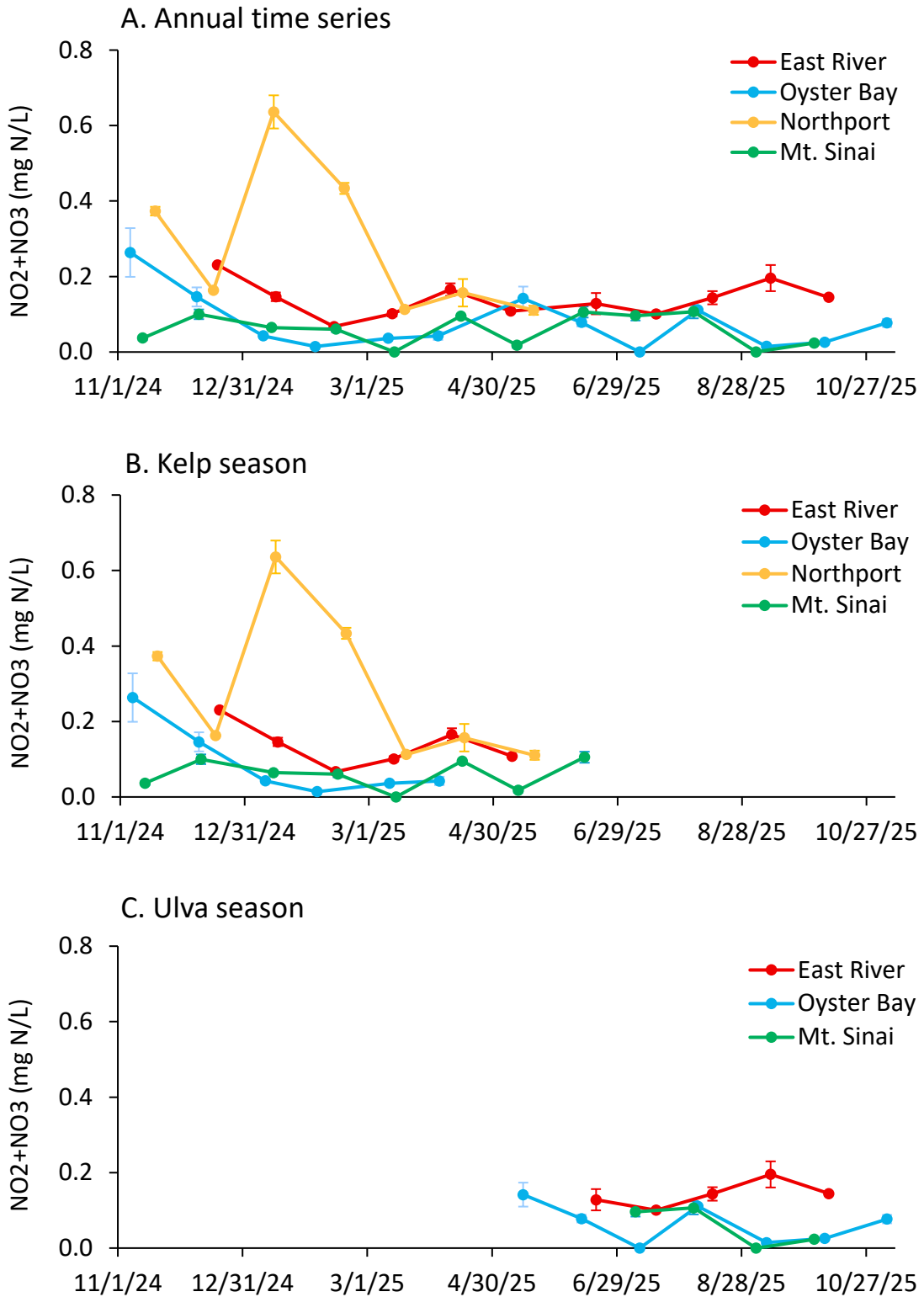


Fig 31. Comparison of dissolved nitrate concentrations among the four study sites over (A) one-year period from November 2024 to November 2025, (B) kelp season (November 2024-May/June 2025), (C) Ulva season (may/June 2025-October/November 2025).

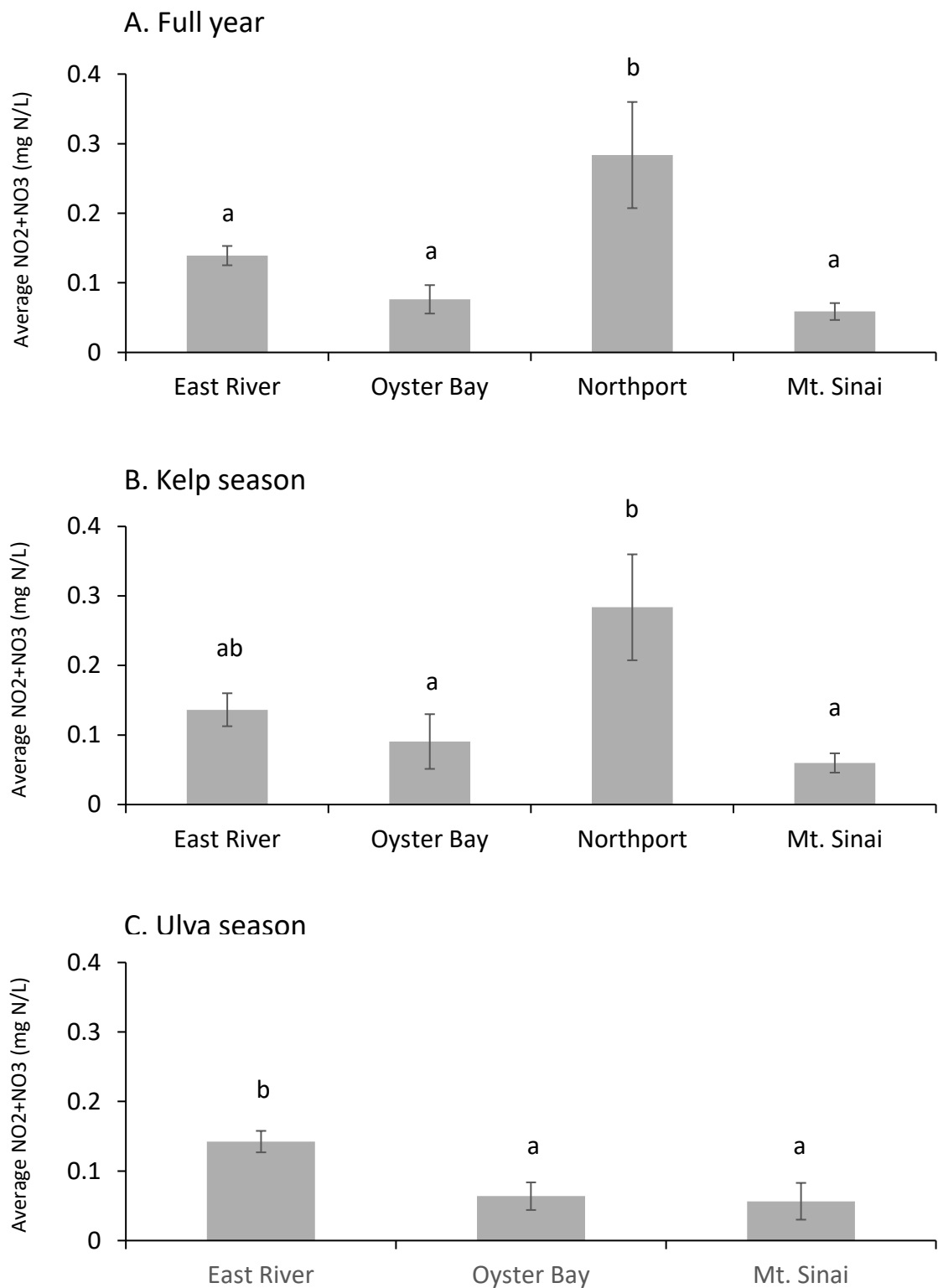


Fig 32. Comparison of dissolved nitrate concentrations at the four study sites averaged from monthly measurements taken over (A) one-year period from November 2024 to November 2025, (B) kelp season (November 2024-May/June 2025), (C) Ulva season (May/June 2025-October/November 2025). GLM's ($p < 0.001$, t value = 4.866 for full year; $p < 0.01$, t value = 2.839 for kelp season) were conducted for full year and kelp season nitrate concentrations, and an ANOVA ($p < 0.05$, f value = 4.972) was conducted on nitrate concentrations from the ulva season. Letters above bars denote significant interactions between treatments (Tukey, $p < 0.01$ for all significant interactions for the full season and $p < 0.05$ for all significant interactions for the kelp and ulva seasons.).

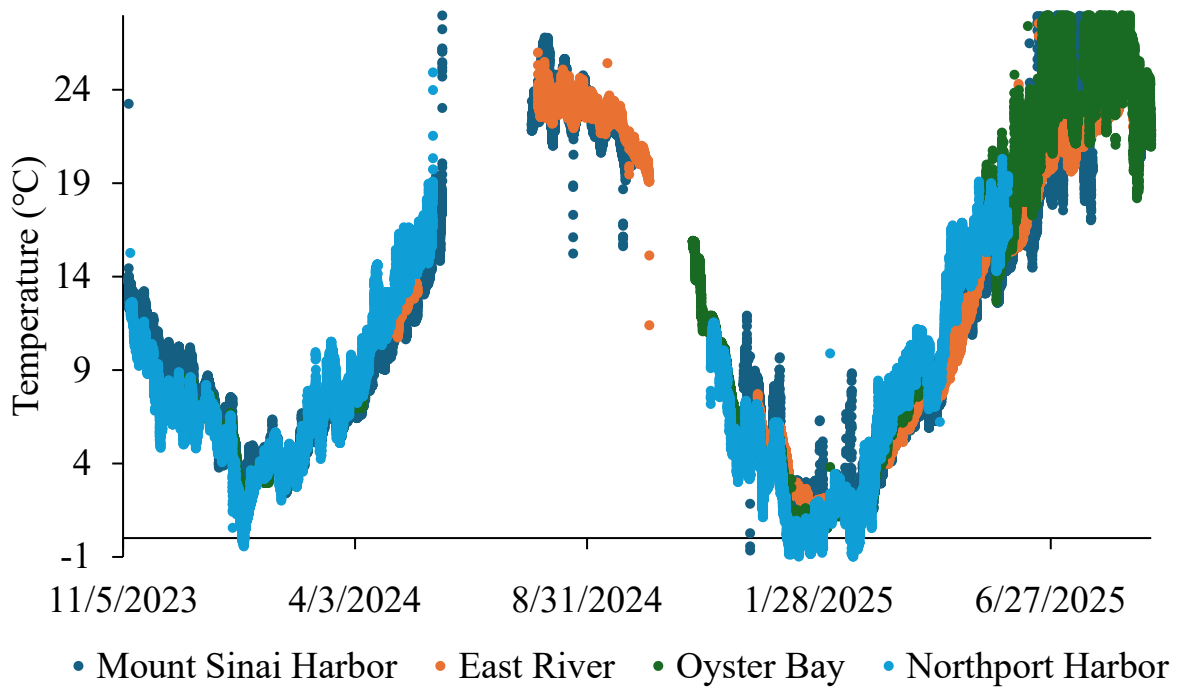


Figure 33. Mount Sinai Harbor, East River, Oyster Bay, and Northport Harbor experiments continuous temperature (°C) readings at 10-minute intervals were recorded from November 2023 until November 2025. Gaps in data reflect no experiment in progress or sensor servicing.

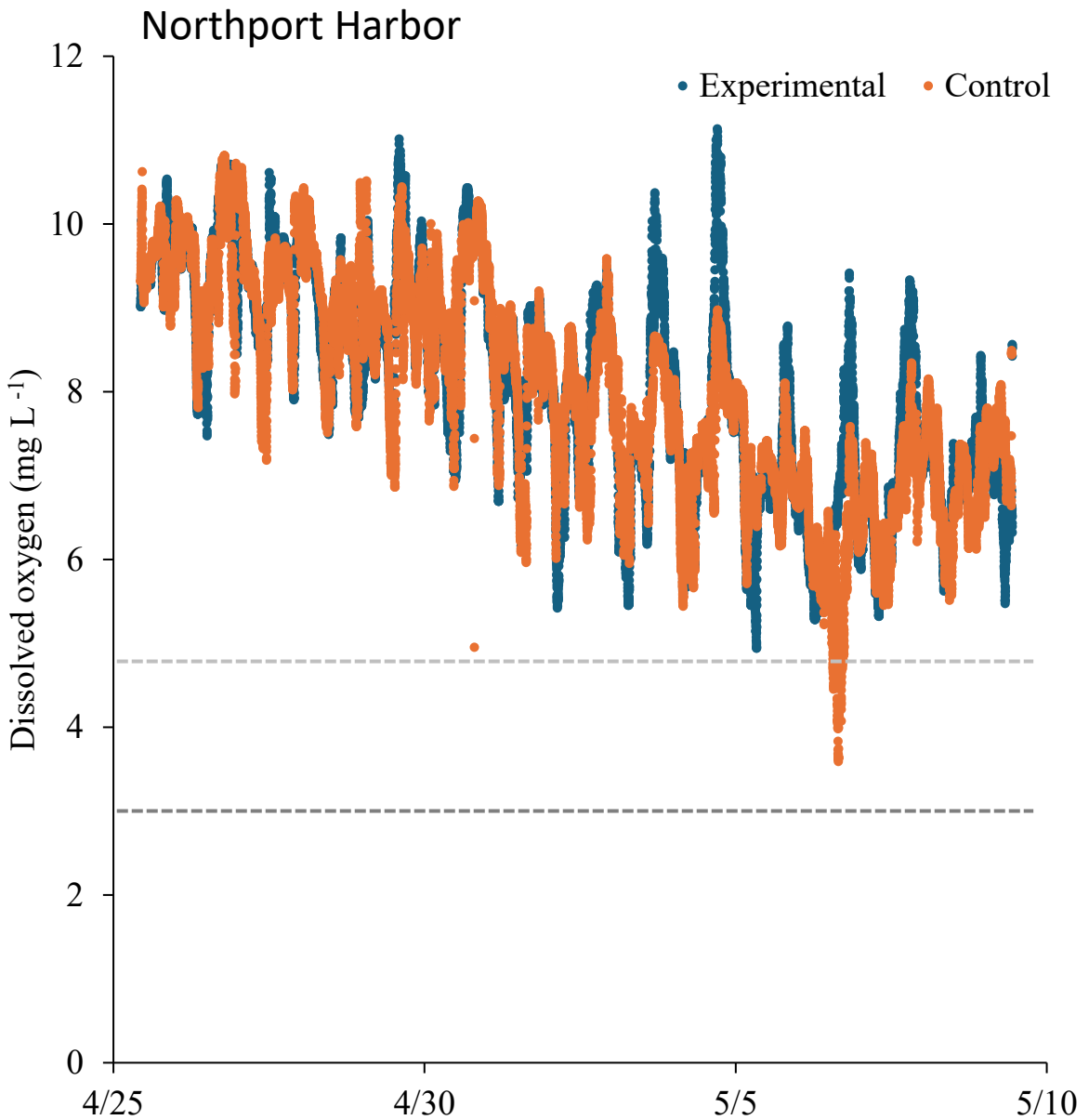


Figure 34. Kelp 2024: Northport Harbor experimental vs. control continuous dissolved oxygen (mg L^{-1}) readings at 10-minute intervals. Dotted lines mark chronic (dark gray) and acute (light gray) dissolved oxygen levels recognized by the NYSDEC at 4.8 mg/L and 3.0 mg/L , respectively.

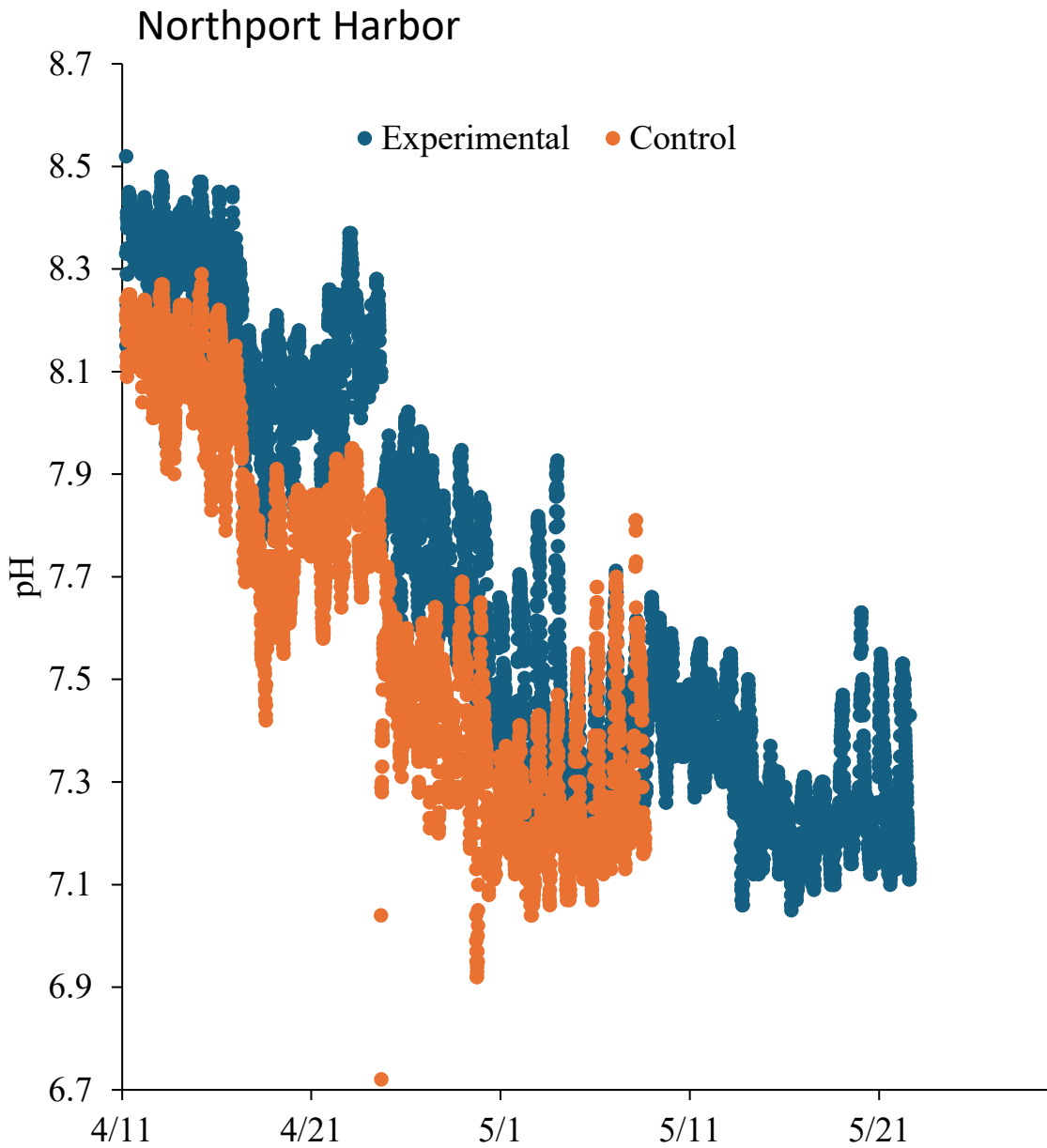


Figure 35. Kelp 2024: Northport Harbor experimental vs. control continuous pH readings at 10-minute intervals.

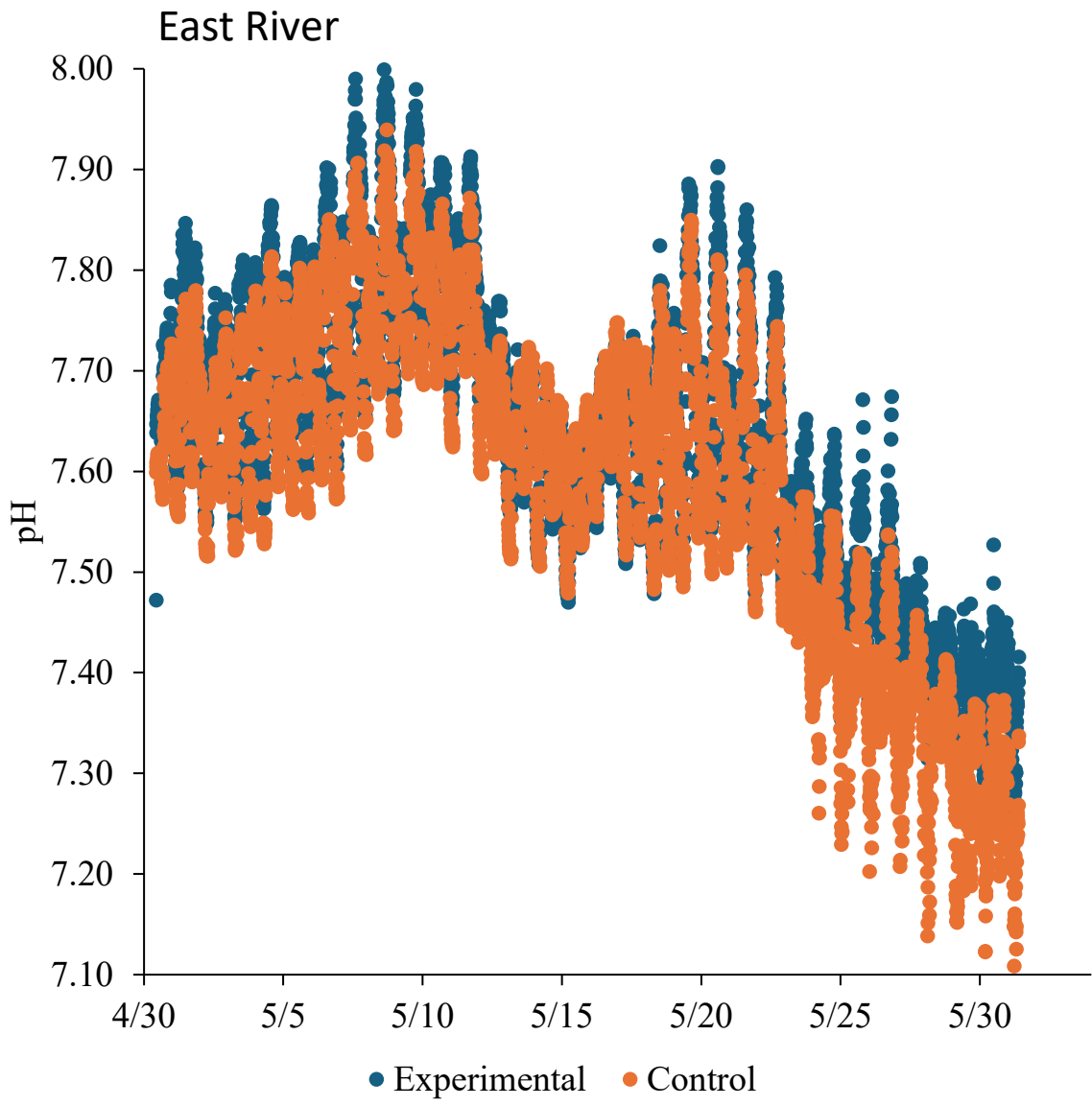


Figure 36. Kelp 2024: East River experimental vs. control continuous pH readings at 10-minute intervals.

Mount Sinai Harbor

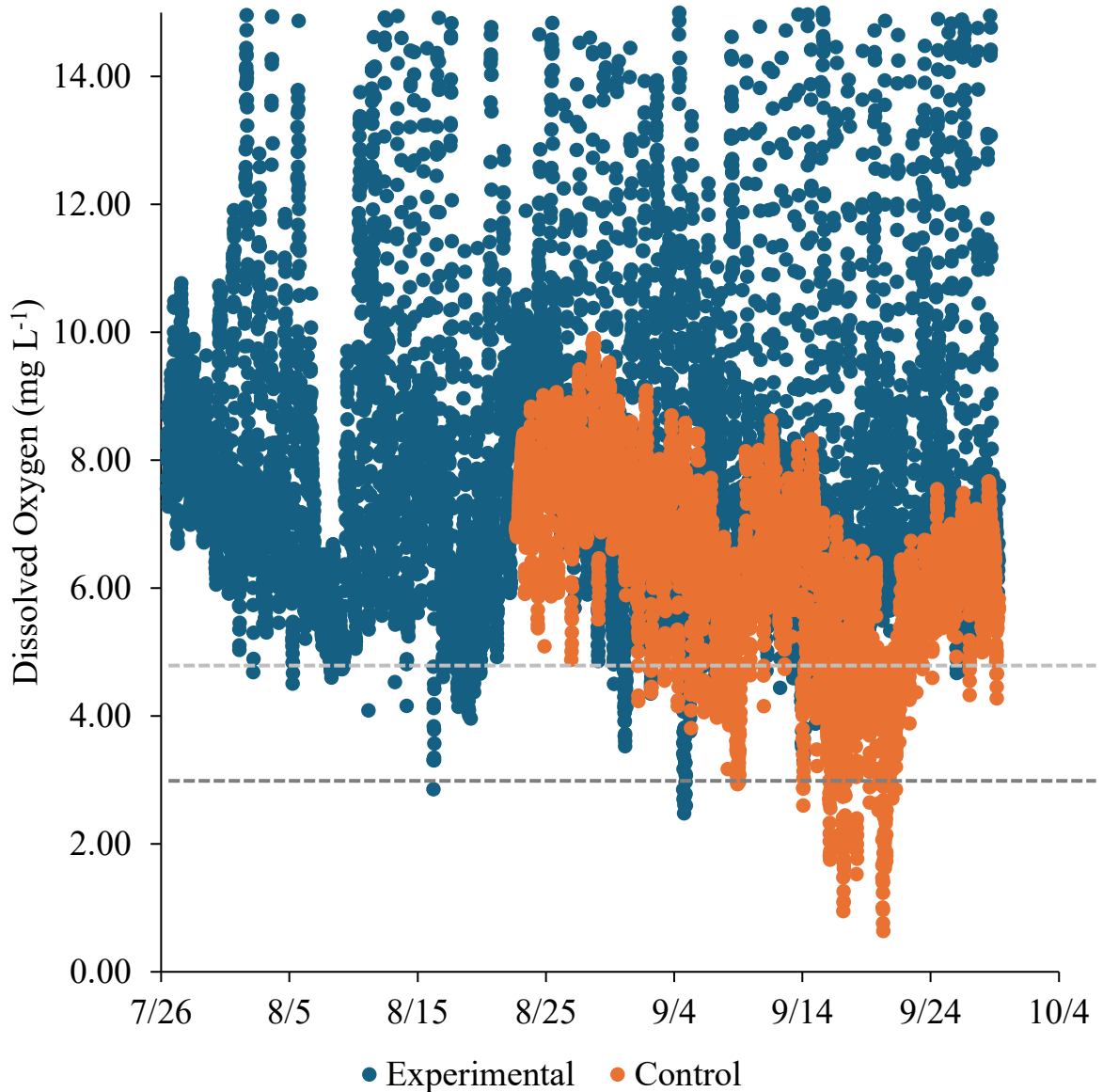


Figure 37. Summer 2024: Mount Sinai Harbor experimental vs. control continuous dissolved oxygen (mg L^{-1}) readings at 10-minute intervals. Dotted lines mark chronic (dark gray) and acute (light gray) dissolved oxygen levels recognized by the NYSDEC at 4.8 mg/L and 3.0 mg/L, respectively.

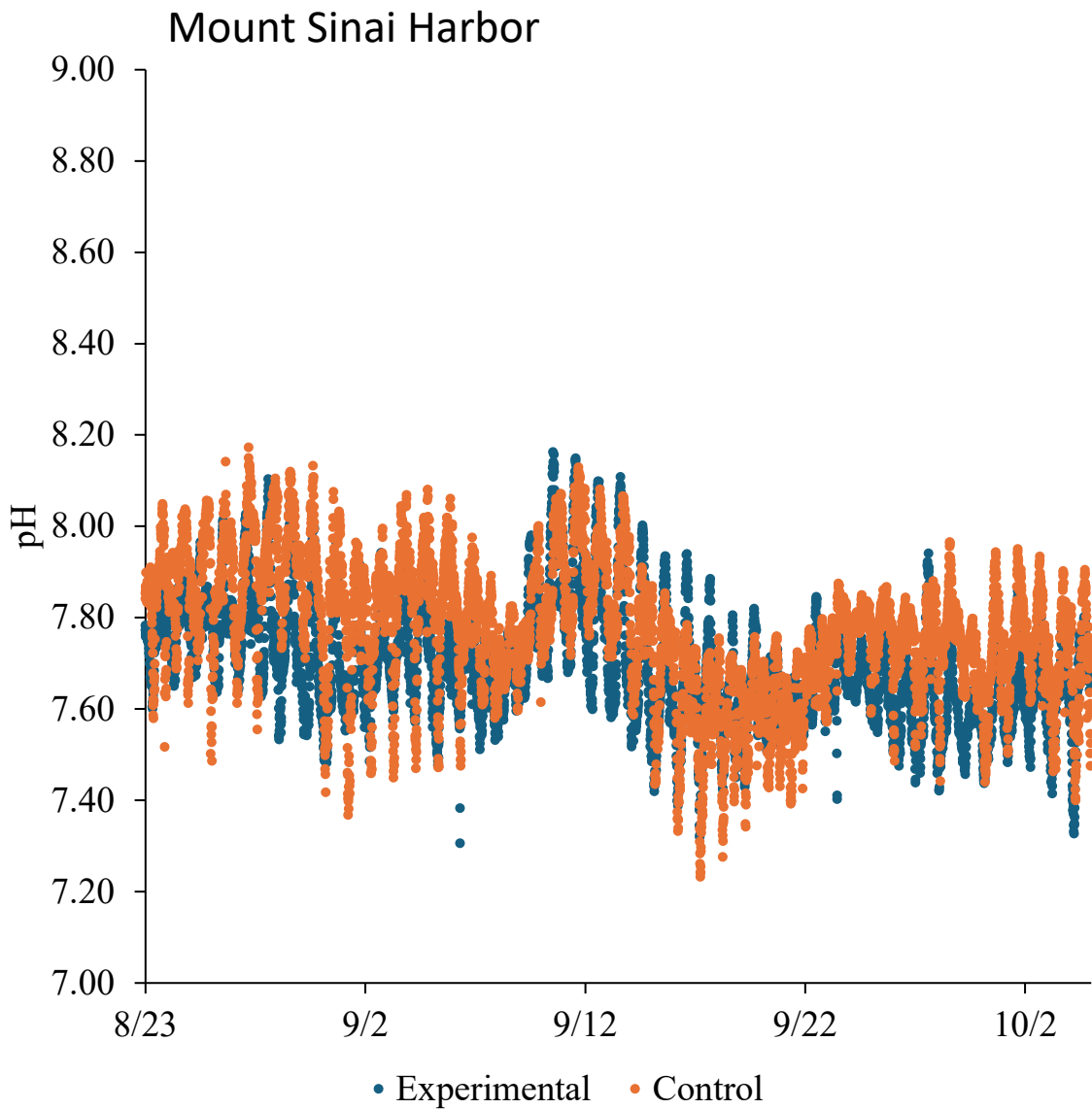


Figure 38. Summer 2024: Mount Sinai Harbor experimental vs. control continuous pH readings at 10-minute intervals.

Mount Sinai Harbor

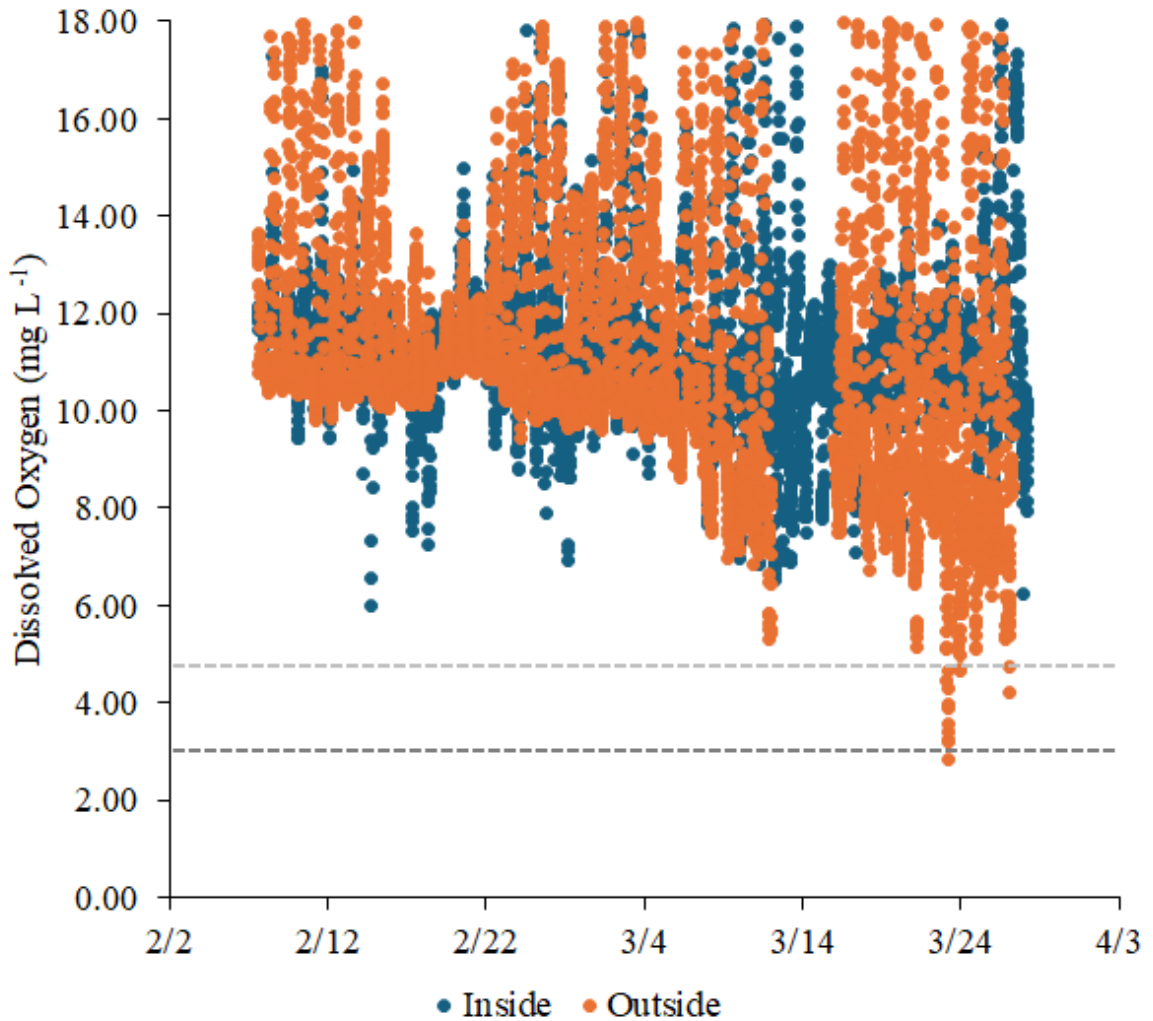


Figure 39. Kelp 2025: Mount Sinai Harbor experimental vs. control continuous dissolved oxygen (mg L⁻¹) readings at 10-minute intervals. Dotted lines mark chronic (dark gray) and acute (light gray) dissolved oxygen levels recognized by the NYSDEC at 4.8 mg/L and 3.0 mg/L, respectively.

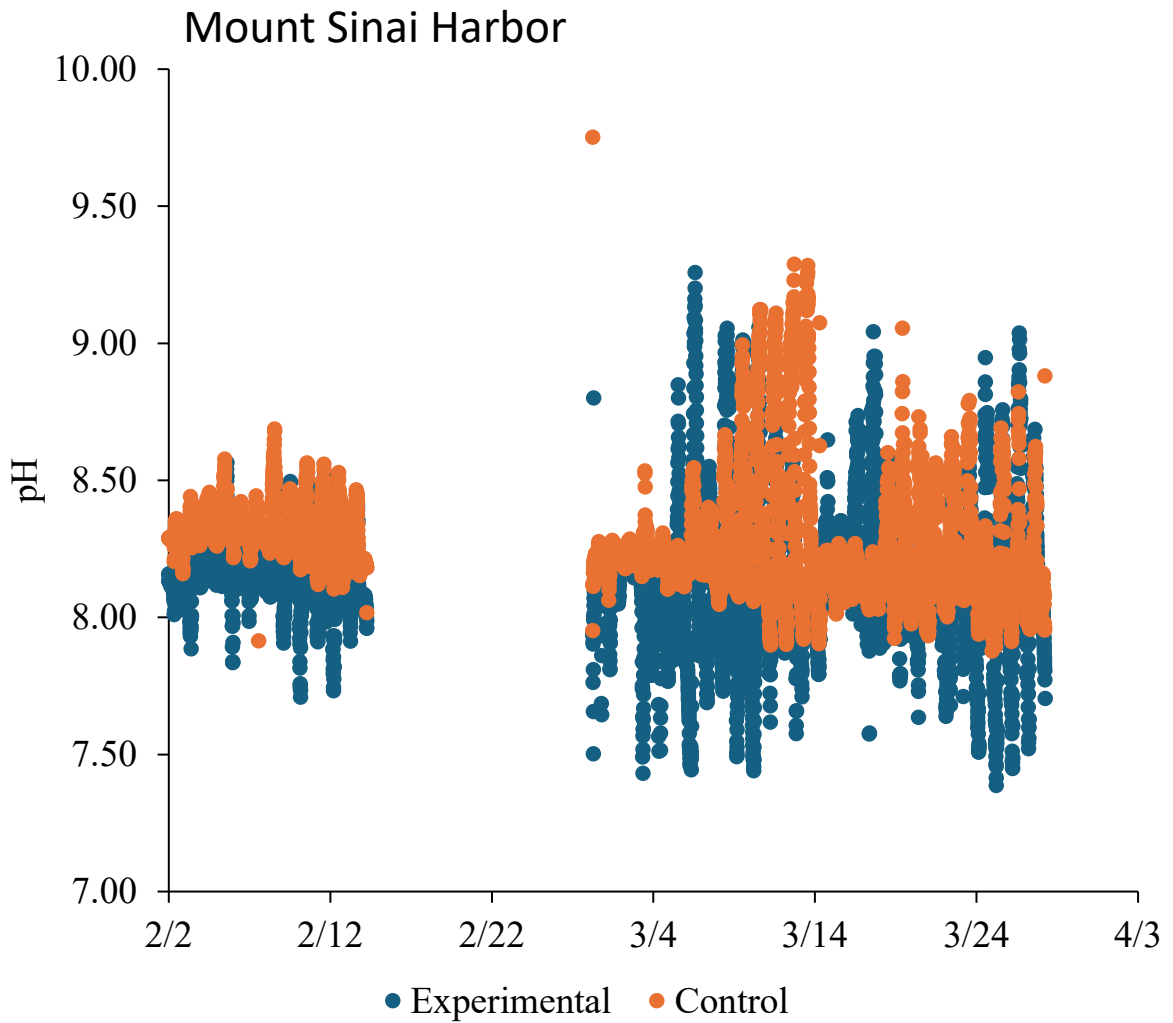


Figure 40. Kelp 2025: Mount Sinai Harbor experimental vs. control continuous pH readings at 10-minute intervals.

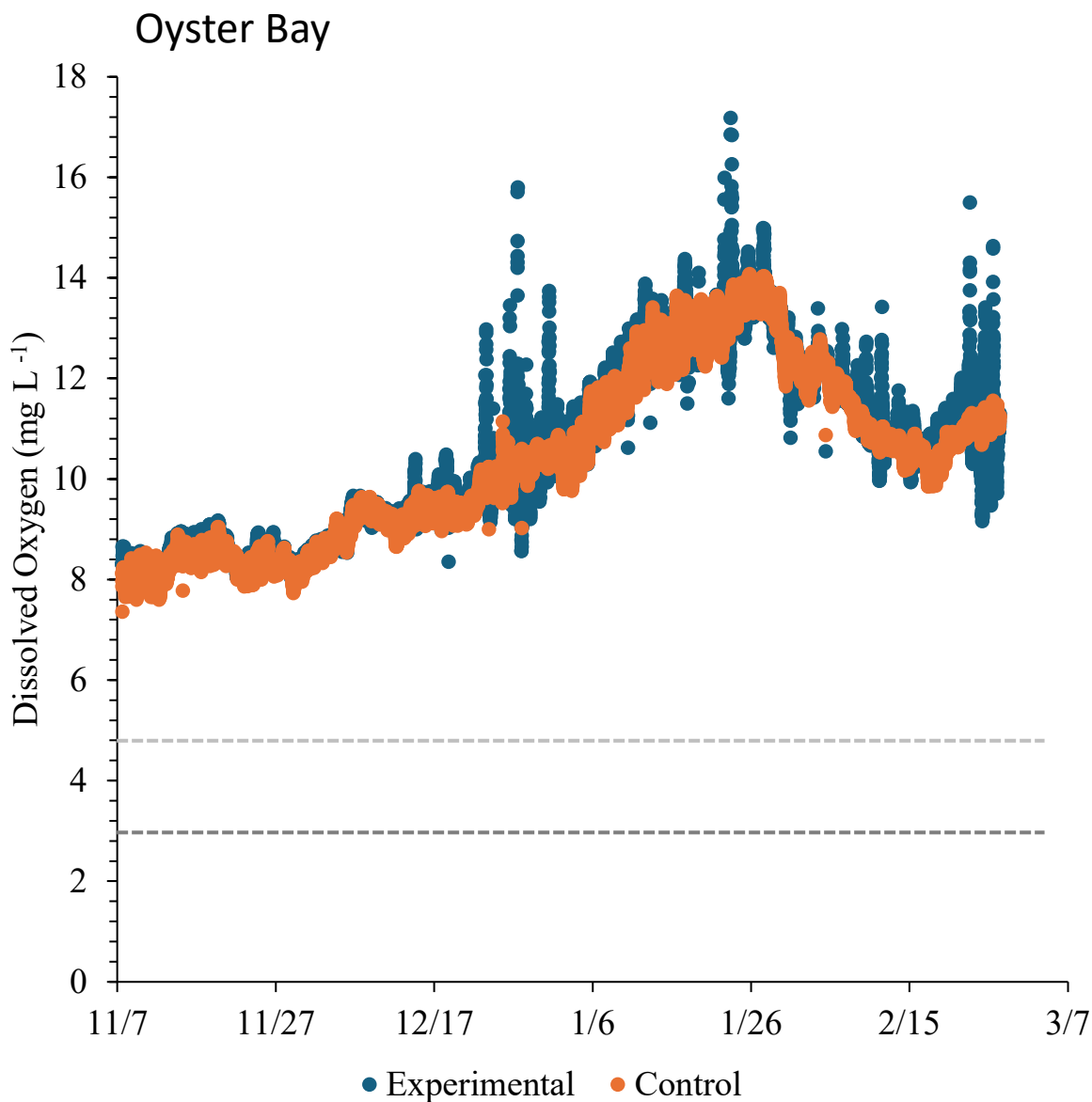


Figure 41. Kelp 2025: Oyster Bay experimental vs. control continuous dissolved oxygen (mg L⁻¹) readings at 10-minute intervals. Dotted lines mark chronic (dark gray) and acute (light gray) dissolved oxygen levels recognized by the NYSDEC at 4.8 mg/L and 3.0 mg/L, respectively.

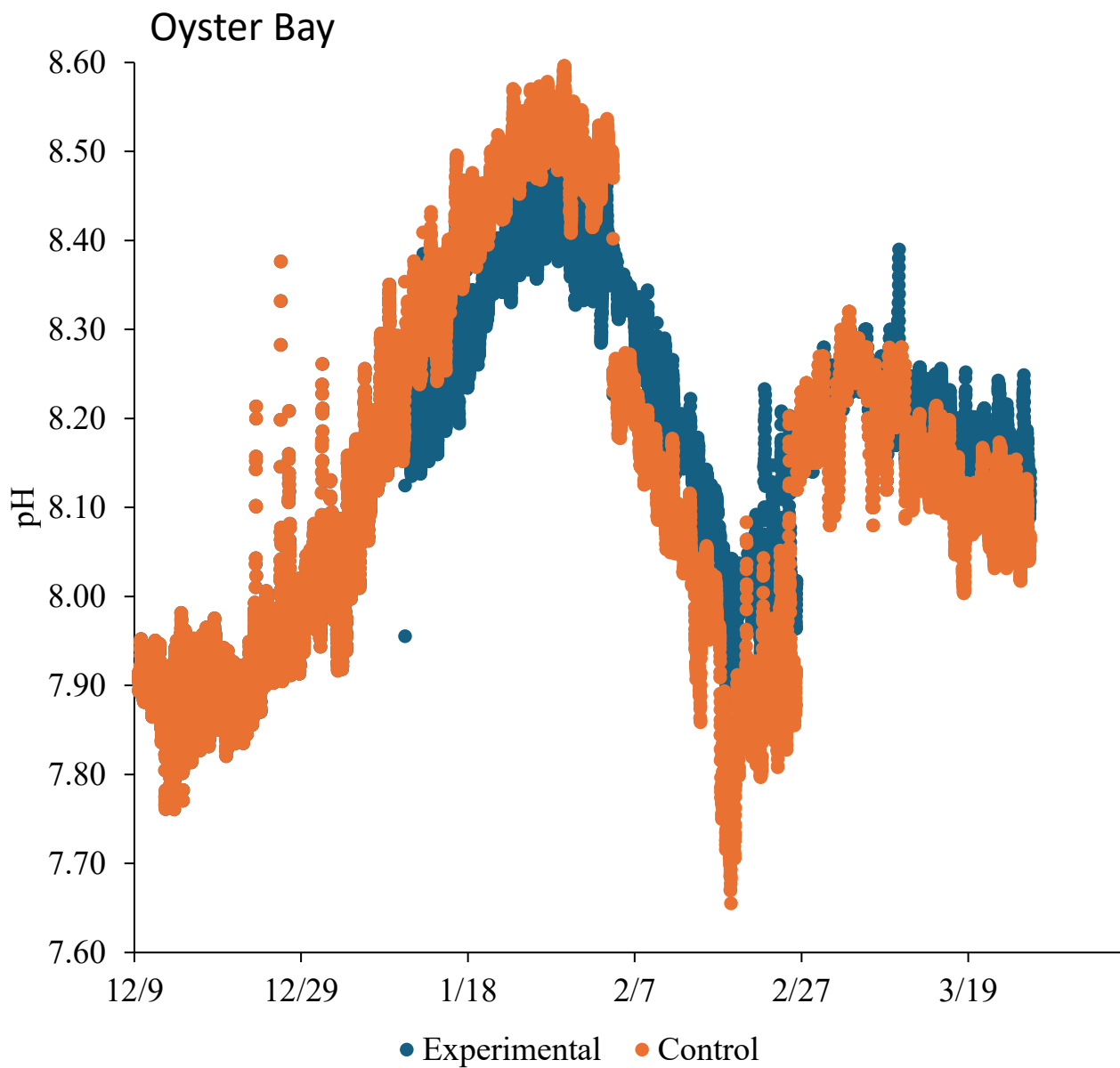


Figure 42. Kelp 2025: Oyster Bay experimental vs. control continuous pH readings at 10-minute intervals.

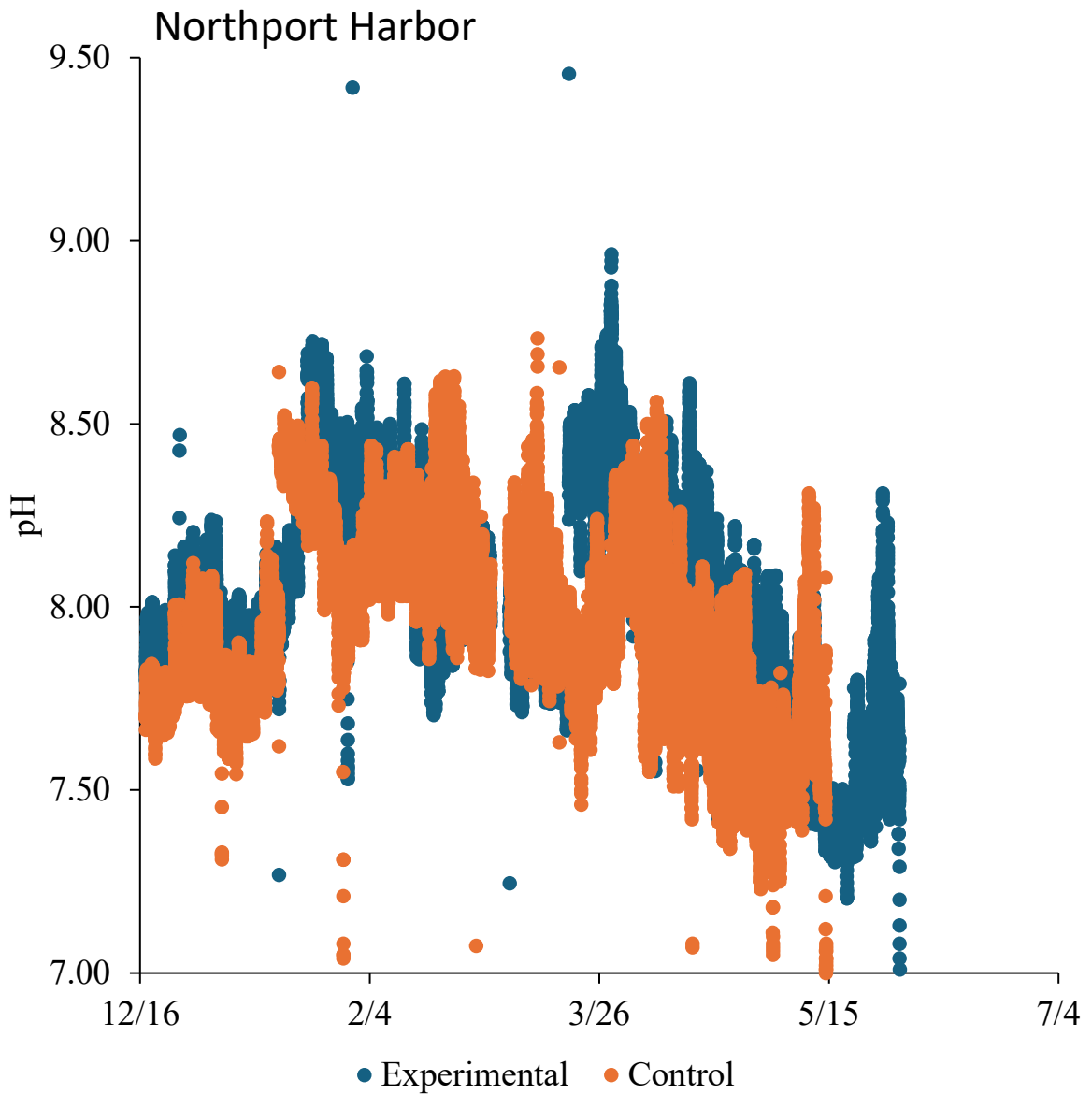


Figure 43. Kelp 2025: Northport Harbor experimental vs. control continuous pH readings at 10-minute intervals.

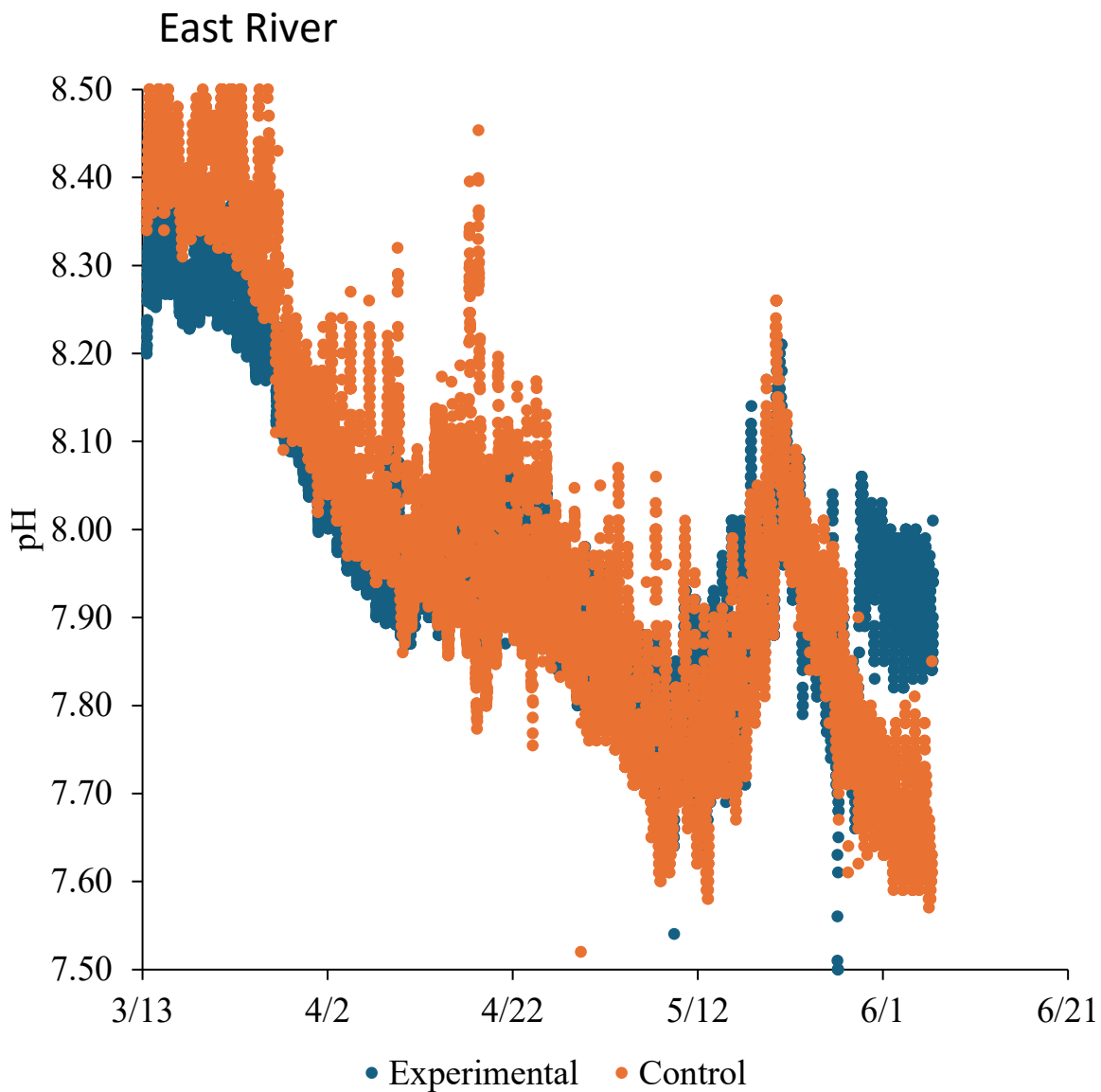


Figure 44. Kelp 2025: East River experimental vs. control continuous pH readings at 10-minute intervals.

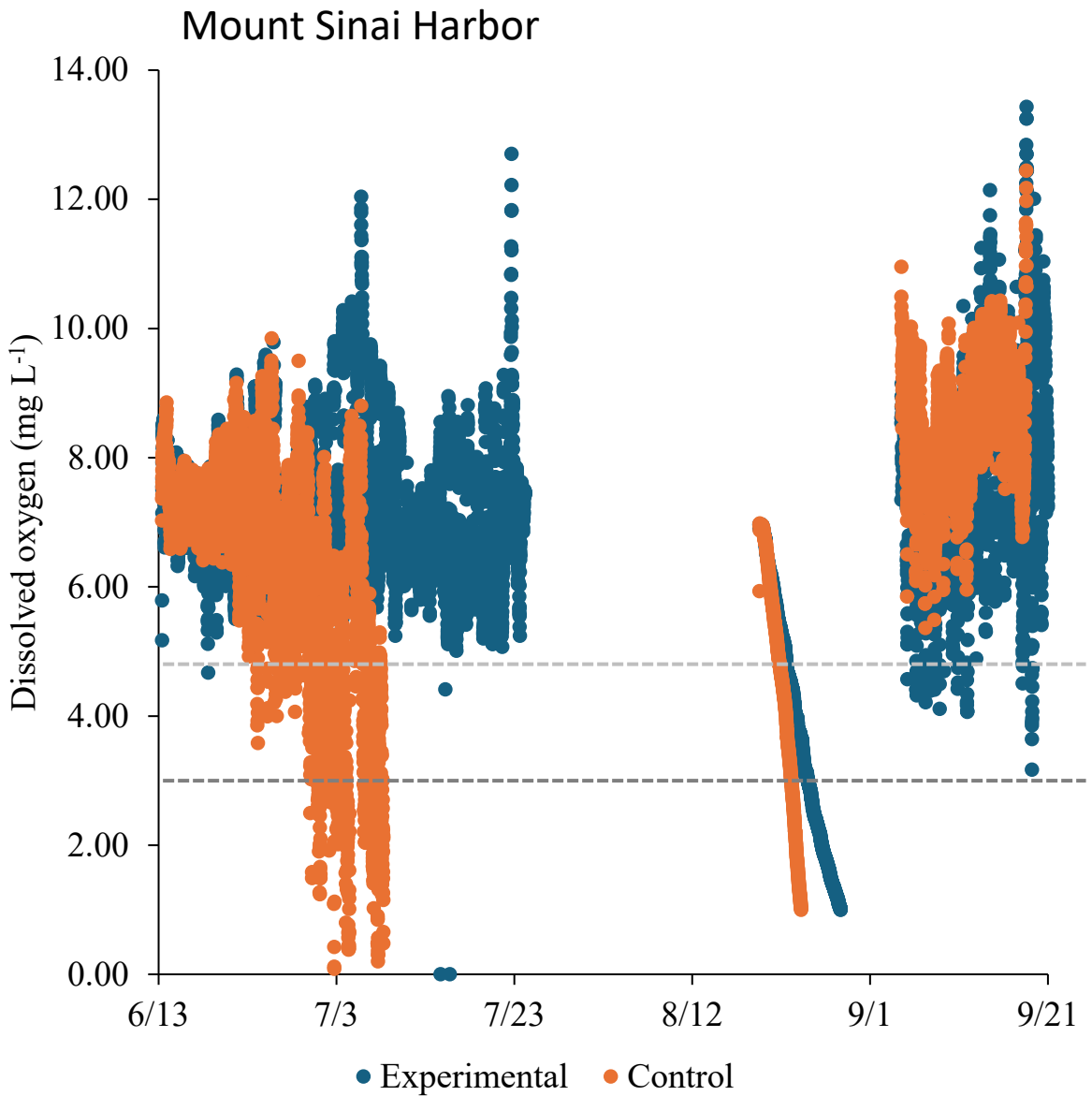


Figure 45. Summer 2025: Mount Sinai Harbor experimental vs. control continuous dissolved oxygen (mg L⁻¹) readings at 10-minute intervals. Dotted lines mark chronic (dark gray) and acute (light gray) dissolved oxygen levels recognized by the NYSDEC at 4.8 mg/L and 3.0 mg/L, respectively.

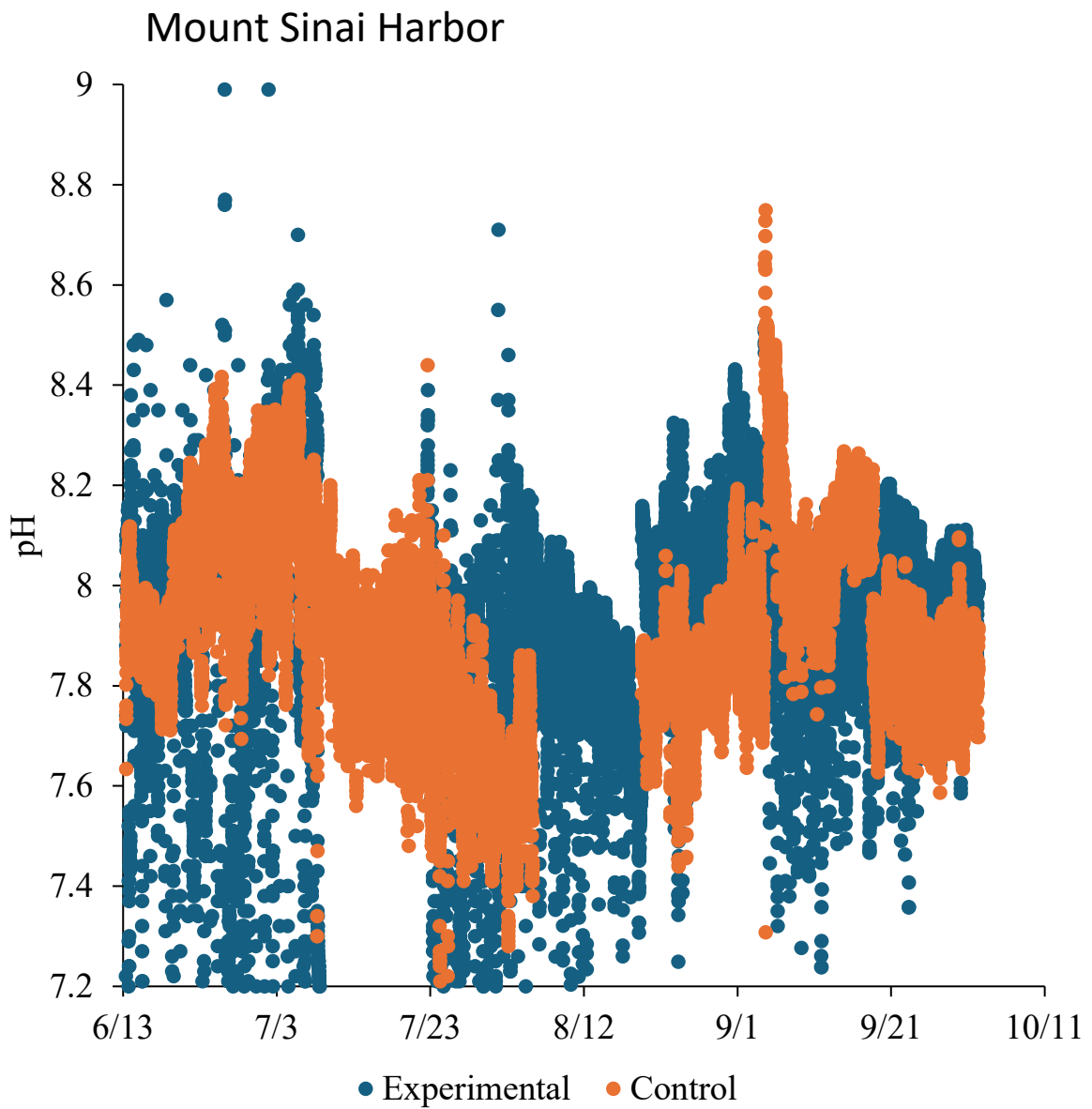


Figure 46. Summer 2025: Mount Sinai Harbor experimental vs. control continuous pH readings at 10-minute intervals. Gap in data reflect sensor servicing.

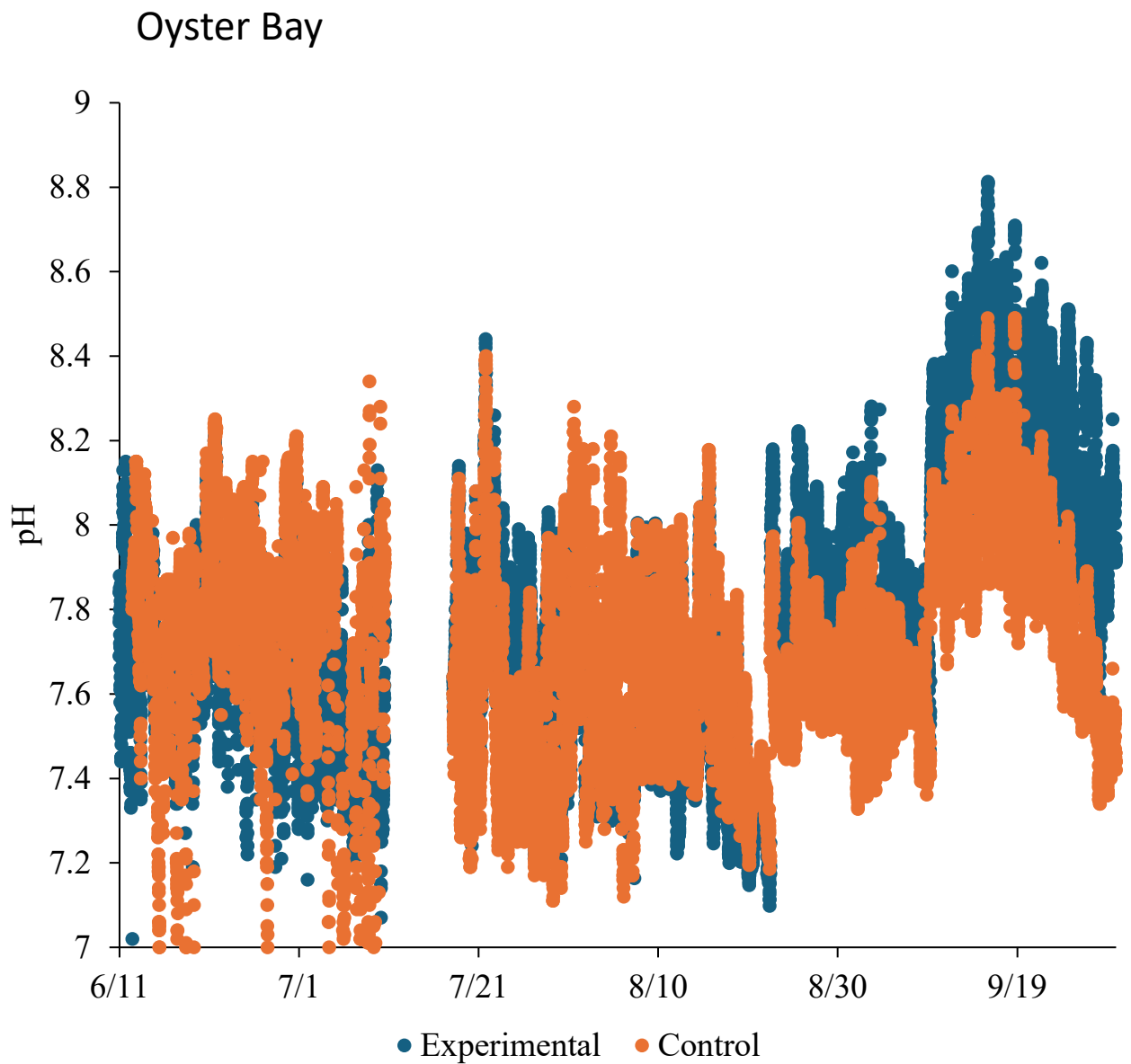


Figure 47. Summer 2025: Oyster Bay experimental vs. control continuous pH readings at 10-minute intervals.

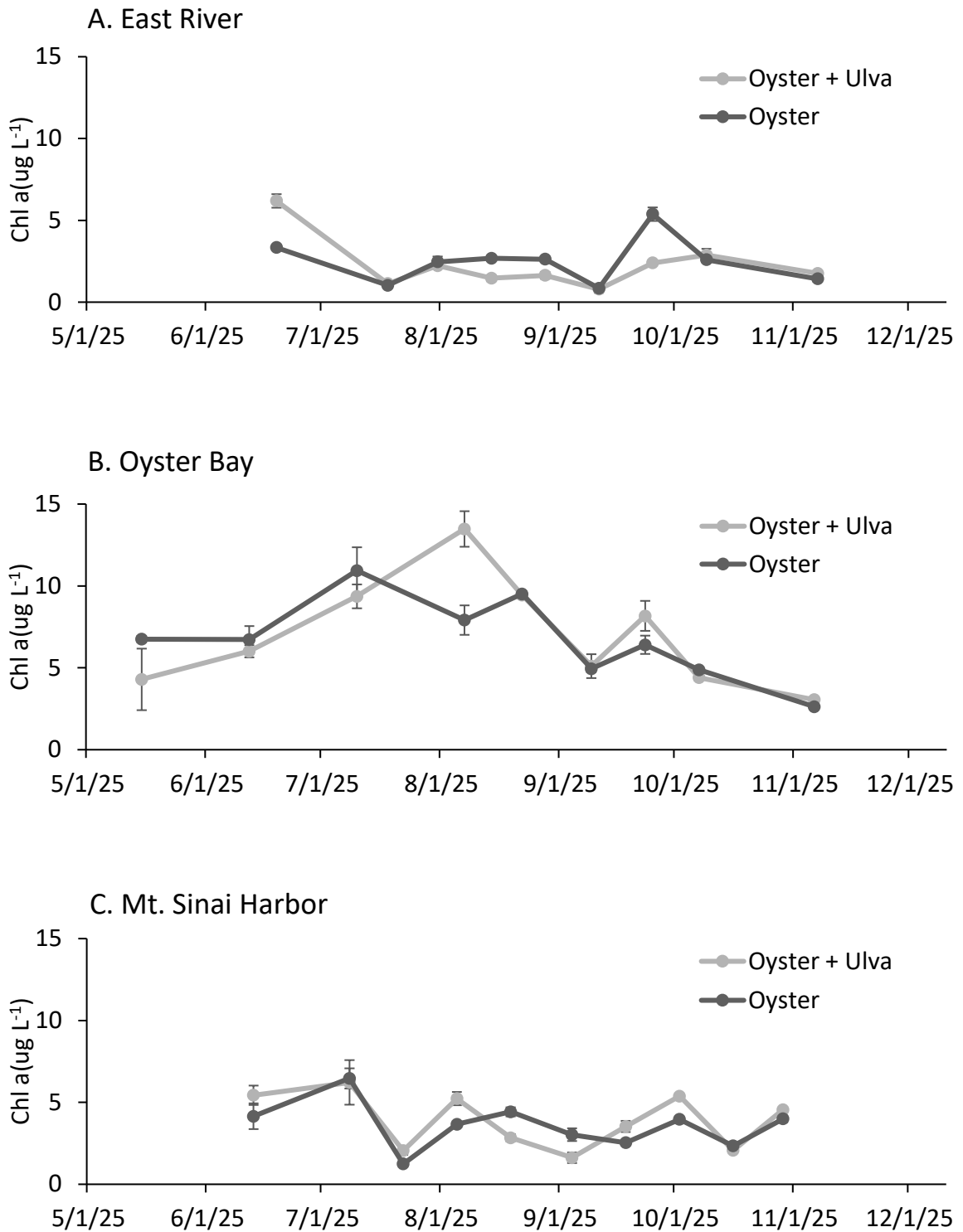
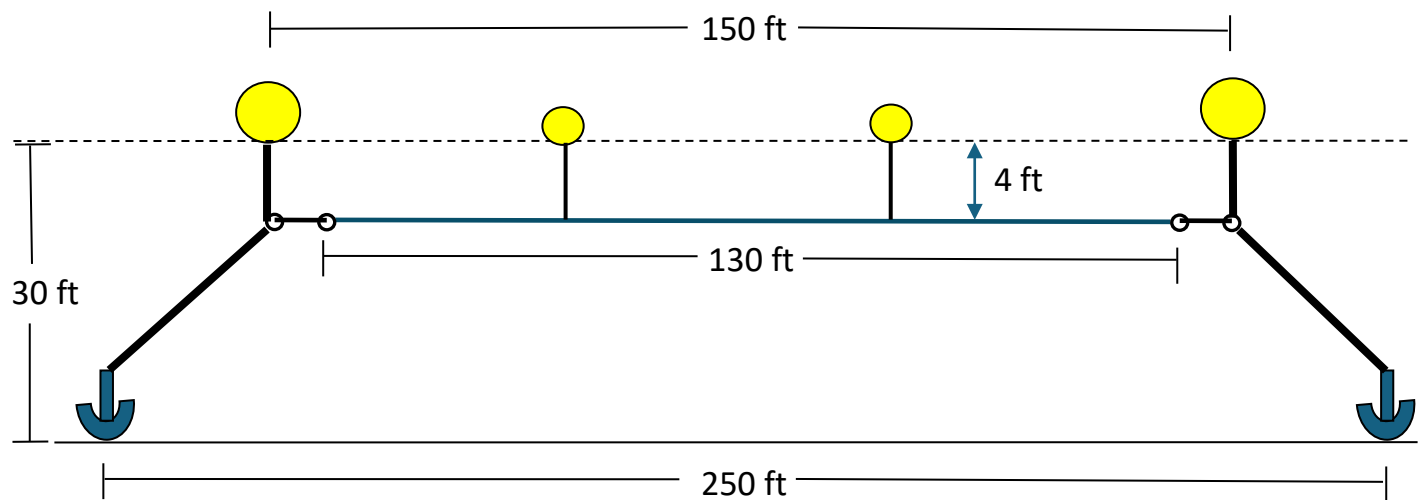






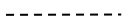


Fig 48. Comparison of chl a concentrations between areas co-cultivated with oysters and Ulva, and areas with oysters only, at three study sites including (A) East River, (B) Oyster Bay, and (C) Mt. Sinai Harbor in 2025.

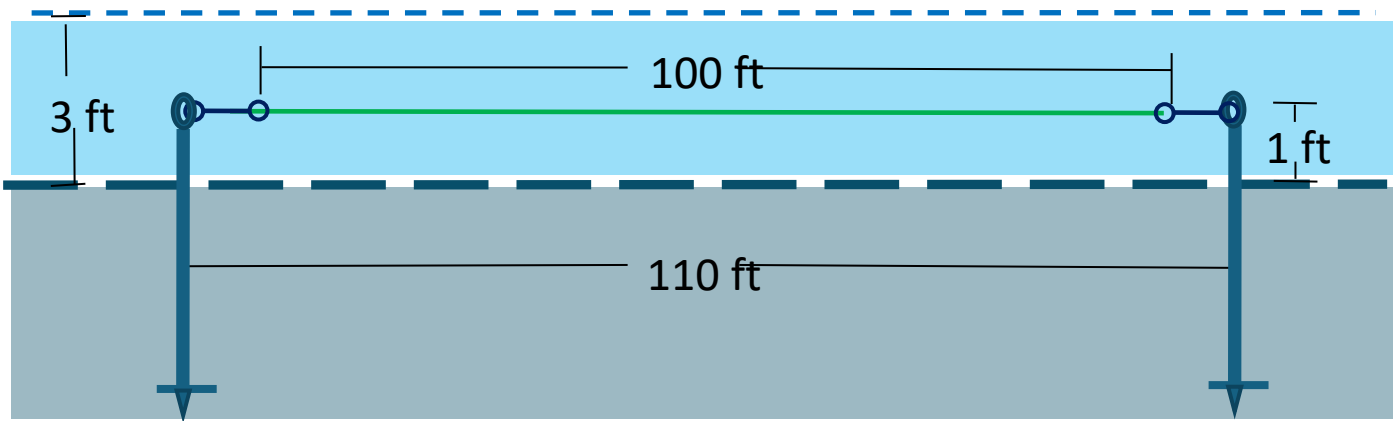
Appendix figures








Legend

-  300-lb mushroom anchor
-  16" white buoy
-  10 ft pigtail
-  12" yellow buoy
-  1/2" rope (130 ft kelp line)
-  Mooring line (3/4" rope + 1/2" chain)
-  Water surface

Appendix Fig 1. Side-view diagram of suspended kelp line installations in deep waters (~30 ft deep) at the East River site (not drawn to scale).



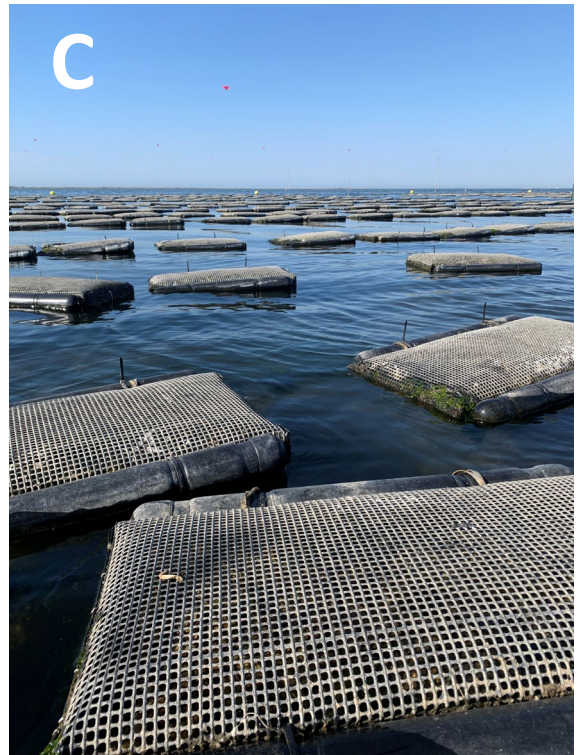
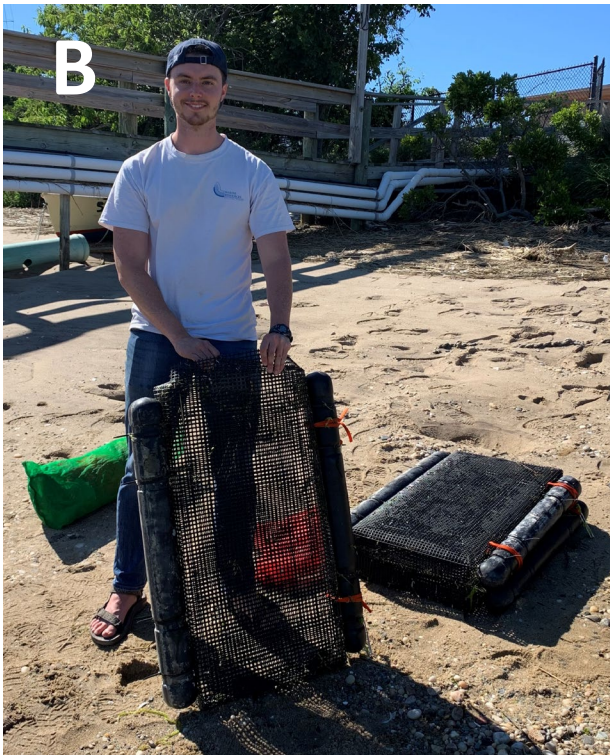
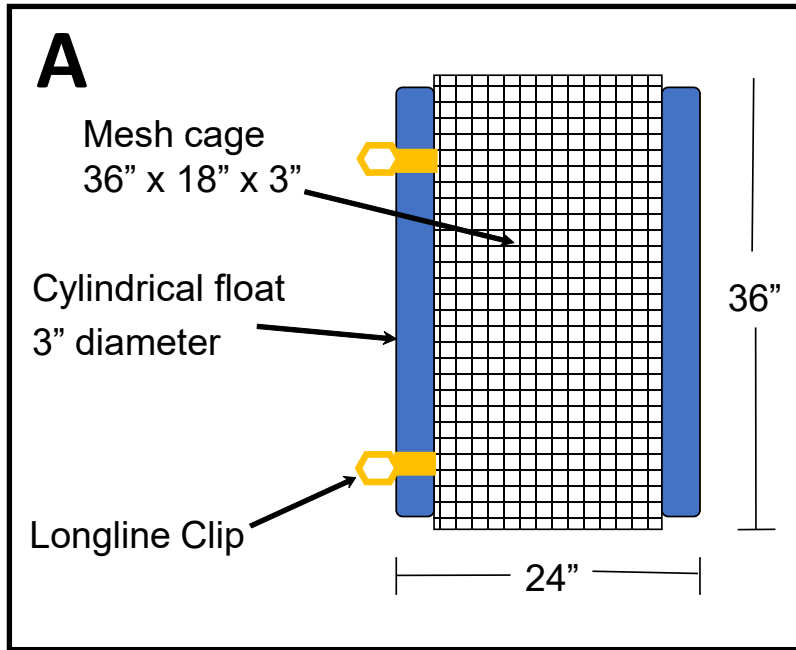
Legend

-  4' Screw anchor
-  5 ft pigtail
-  ½" rope (100 ft kelp line)
-  Water surface
-  Bay bottom

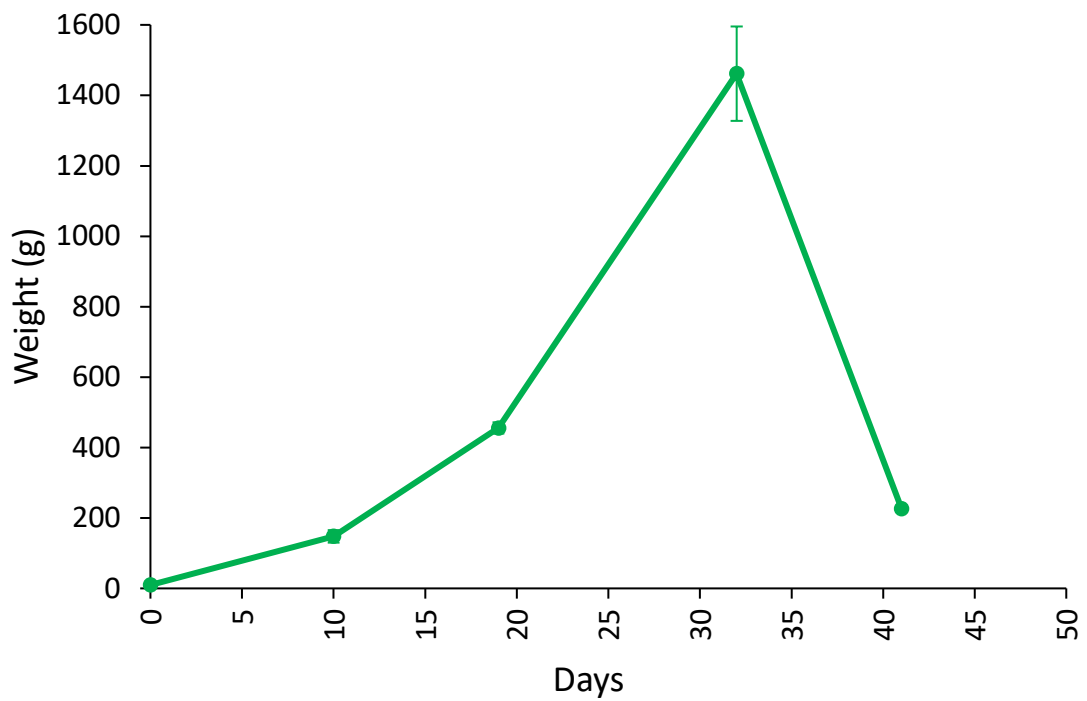
Appendix Fig 2. Side-view diagram of staked kelp line installations in shallow waters (< 4 ft MLW) at the Northport Harbor and Mt. Sinai Harbor sites (not drawn to scale).



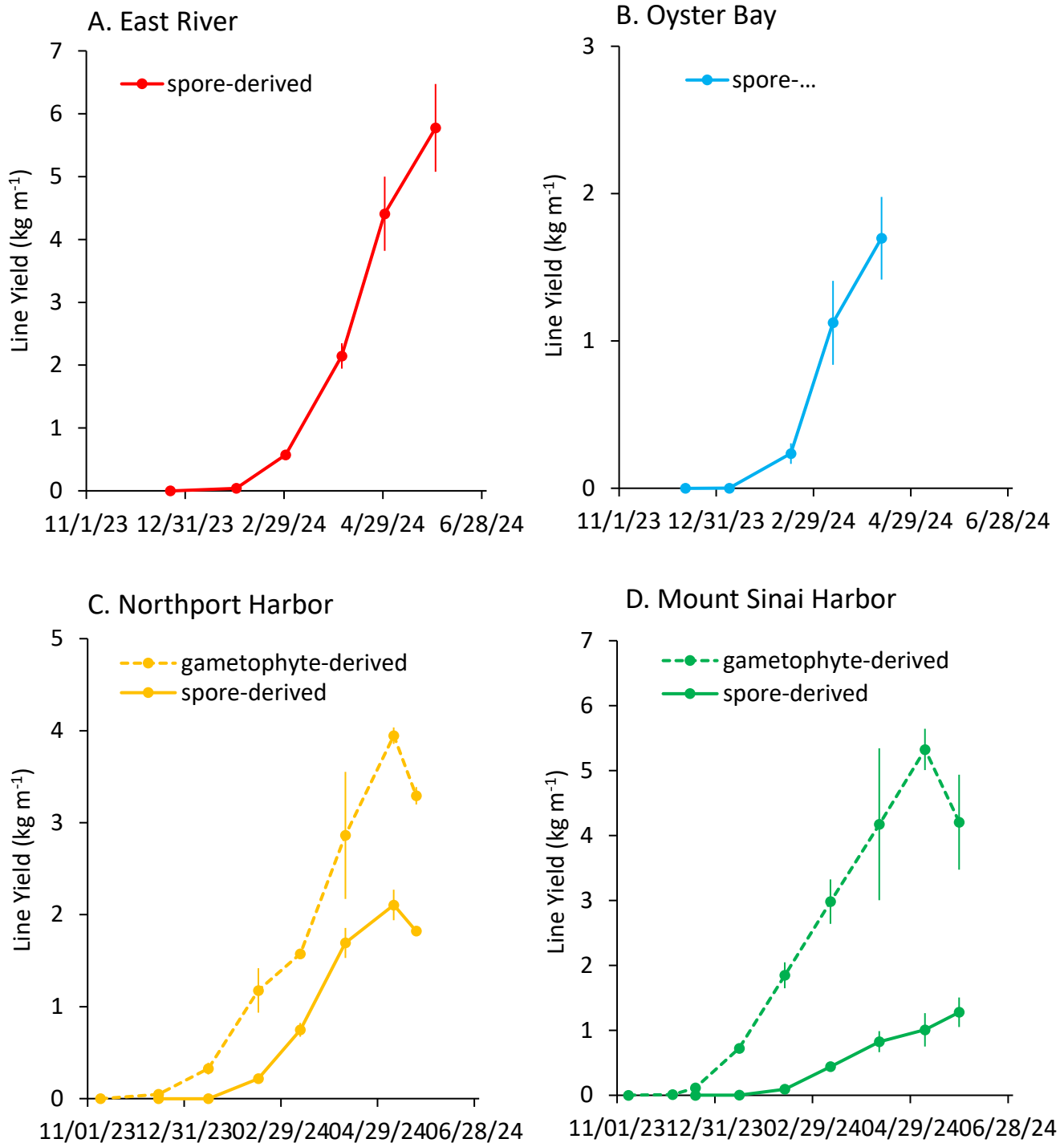
Appendix Fig 3. Photos outlining steps in the process of making spore-derived kelp seedstock, including (A) collecting reproductive tissue (i.e., sorus tissue) from local wild populations of sugar kelp, (B) preparing the sorus tissue for spore release, (C) releasing spores from the sorus tissue following overnight desiccation, and (D) adding the spores to aquaria containing with spools of string.



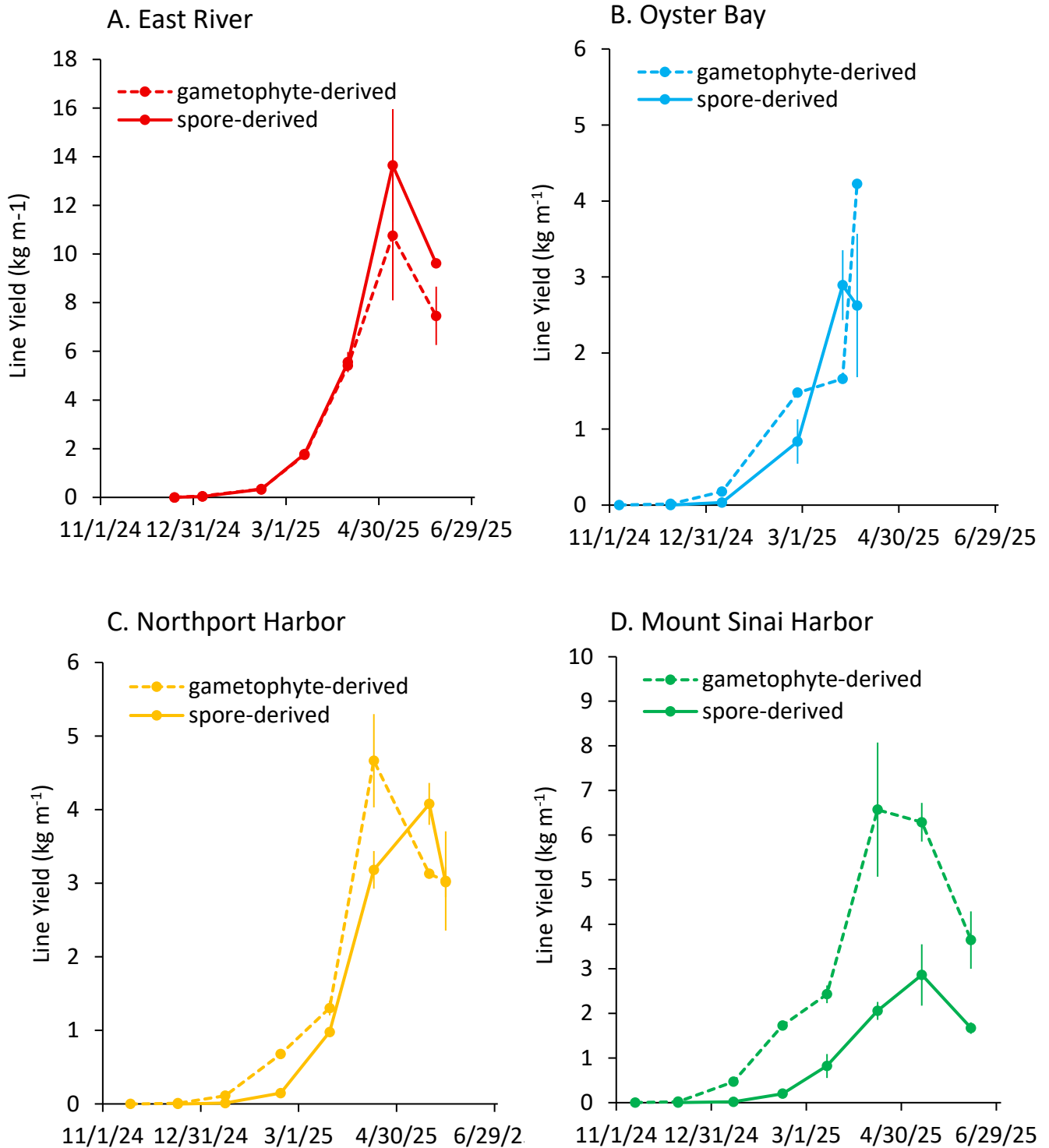
Appendix Fig 4. (A) Diagram and **(B, C)** photos of floating grow-out bag used to cultivate *Ulva* sp. The bag is commonly used to cultivate oysters, particularly in shallow waters.



Appendix Fig 5. Growth curve of *Ulva* cultivated in floating oyster bags in Northport Harbor in 2021. The data was collected in previous experiments conducted by the Gbler Lab.



Appendix Fig 7. Comparison of kelp growth between gametophyte-derived (dotted lines) and spore-derived (solid lines) kelp lines at each of four study sites, including (A) East River, (B) Oyster Bay, (C) Northport Harbor, and (D) Mount Sinai Harbor during the 2024 growing season. Gametophyte-derived were only deployed at only two of the sites, Northport Harbor and Mount Sinai Harbor. In both locations, higher line yields were achieved with gametophyte-derived seedstock. The large difference in growth between gametophyte-derived and spore-derived kelp lines has important implications for nutrient bioextraction potential. Gametophyte-derived kelp data from ongoing parallel study conducted by the Gobler Lab and funded by the Long Island Sound Partnership (unpublished).



Appendix Fig 8. Comparison of kelp growth between gametophyte-derived (dotted lines) and spore-derived (solid lines) kelp lines at each of four study sites, including (A) East River, (B) Oyster Bay, (C) Northport Harbor, and (D) Mount Sinai Harbor during the 2025 growing season. . Gametophyte-derived kelp data from ongoing parallel study conducted by the Gobler Lab and funded by the Long Island Sound Partnership (unpublished).