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Deciphering the water quality impacts of COVID-19 human mobility shifts in estuaries surrounding New York City

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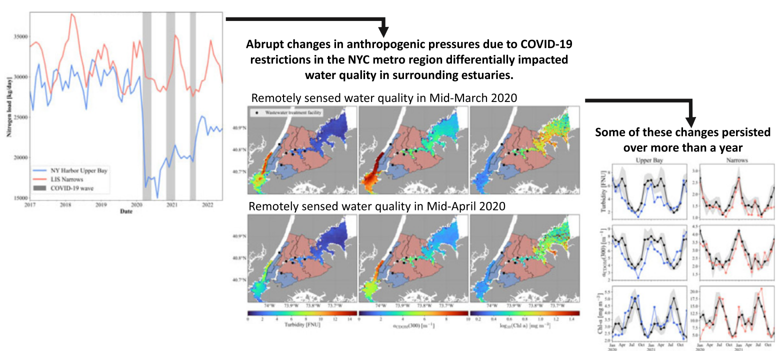
HIGHLIGHTS

- Mobility shifts during COVID-19 influenced wastewater discharge into NY Harbor.
- Nitrogen load reductions resulted in short- and long-term water quality improvement.
- Shifts in nutrient loads and water quality were markedly less pronounced across LIS.
- High resolution satellite imagery captured impacts of COVID-19 on water quality.
- Changes in wastewater outflow predominantly drove early shifts in NYC water quality.

GRAPHICAL ABSTRACT



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ABSTRACT

The COVID-19 pandemic altered human mobility, particularly in large metropolitan areas. In New York City (NYC), stay-at-home orders and social distancing led to significant decreases in commuting, tourism, and a surge of outward migration. Such changes could result in decreased anthropogenic pressure on local environments. Several studies have linked COVID-19 shutdowns with improvements in water quality. However, the bulk of these studies primarily focused on short-term impacts during shutdown periods, without assessing longer-term impacts as restrictions eased. Here, we examine both concurrent lockdown and societal reopening impacts on water quality, using pre-pandemic baseline conditions, in two highly urbanized estuaries surrounding NYC, the New-York Harbor estuary and Long Island Sound (LIS). We compiled datasets from 2017 to 2021 of mass-transit ridership, work-from-home trends, and municipal wastewater effluent to assess changes in human mobility and anthropogenic pressure during multiple waves of the pandemic in 2020 and 2021. These were linked to changes in water quality assessed using high spatiotemporal ocean color remote sensing, which provides near-daily observations across the estuary study regions. To distinguish anthropogenic impacts from natural environmental variability, we examined meteorological/hydrological conditions, primarily precipitation and wind. Our results show that nitrogen loading into the New York Harbor declined significantly in the spring of 2020 and remained below pre-pandemic values through 2021. In contrast, nitrogen loading into LIS remained closer to the pre-pandemic average. In response, water clarity in New-York Harbor significantly improved, with less of a change in LIS. We further show that changes in nitrogen loading had higher impact on water quality than meteorological conditions. Our study demonstrates the value of remote sensing observations in assessing water quality changes when field-based monitoring is hindered and highlights the complex nature of urban estuaries and their heterogeneous response to changes in extreme events and human behavior.

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1. Introduction

The emergence of COVID-19 in late 2019 and the ensuing global pandemic has had far-reaching impacts on society, some of which continue to be felt more than three years later (Delardas et al., 2022; Gershberg, 2022; Ozili and Arun, 2020). Measures imposed to limit COVID-19 transmission during the different stages of the pandemic (i.e., lockdowns and travel restrictions, social distancing, and remote work) changed the degree of anthropogenic pressure across the biosphere (Armstrong et al., 2022; Lenart-Boroń et al., 2022; Rutz et al., 2020). Estuarine and coastal ecosystems are disproportionally impacted by anthropogenically-enhanced stressors, such as climate change, in addition to natural and extreme weather events compared to other aquatic and marine environments (Panton et al., 2020; Verdonschot et al., 2013; von Glasow et al., 2013). Anthropogenic stressors are even more acute in estuaries near major metropolises that are highly urbanized and densely populated, such as New York City (NYC) (Tzortziou et al., 2022; von Glasow et al., 2013).

Effects of COVID-19 related lockdowns on estuarine water quality have been examined in several coastal ecosystems globally. Some studies showed improved water quality in the months immediately following lockdown measures. For example, water clarity reportedly increased due to decreased boat traffic and harbor operations in Venice, Italy (Braga et al., 2020), Belize (Callejas et al., 2021) and Chennai, India (Vijay Prakash et al., 2021), while decreased industrial pollution and tourism in Vembanad Lake (Yunus et al., 2020) and the Ashtamudi Wetland system in India (Aswathy et al., 2021) also led to clearer waters. Turbidity levels in the Hudson River showed a significant decrease near the NYC's North River water pollution control plant immediately following the COVID-19 lockdown (Schulberg and Subramaniam, 2021). Water quality parameters such as dissolved oxygen (DO) and ammonia nitrogen concentrations across multiple rivers in China also showed significant changes, with greater improvements in rivers draining more economically developed landscapes (Liu et al., 2022). Most of these studies, however, used observations limited to the first wave of the pandemic (i.e., spring of 2020 in the United States) and did not assess longer-term ecological impacts. Liu et al. (2022) showed that after restrictions in China were lifted in June 2020, water quality quickly returned to pre-pandemic values, stressing the ephemeral nature of these improvements. Conversely, other studies found little to no measurable impact of COVID-19 on water quality parameters. For example, in the Meriç-Ergene River Basin in Turkey, turbidity, total suspended solids (TSS) and DO levels were not significantly impacted during COVID-19 restrictions as domestic wastewater and agricultural runoff persisted throughout the basin (Tokatlı and Varol, 2021). Wetz et al. (2022) argued that changes in water quality observed across multiple coastal systems in the U.S., which may have been attributed to decreased human activities, were greatly outweighed by natural variability. Furthermore, Braga et al. (2022) noted that lower chlorophyll-*a* (Chl-*a*) in the North Adriatic Sea in April 2020 could mostly be explained by seasonal variability in meteorological and hydrological conditions, although the possibility of a second-order anthropogenic effect could not be ruled out. These studies highlight the difficulty of disentangling natural and anthropogenic factors when assessing the impacts of COVID-19 on coastal water quality and the need to consider the extent to which these changes were temporally localized or persisted beyond initial lockdown periods.

Compared to other cities in the U.S., NYC's population, public health, and economy were among the hardest hit by COVID-19 (Cordes and Castro, 2020; McKinley, 2020), with the most dramatic changes between April and June 2020 (Kaplan et al., 2022; Pagsuyoin et al., 2022). On March 20th 2020, New York State signed a statewide stay-at-home executive order to curtail COVID-19 transmission (New York State, 2020). In May 2020, Kevin Quealy reported in *The New York Times* that roughly 420,000 people (~5% of the population) moved away from the city due to COVID-19, primarily from the wealthiest neighborhoods in Manhattan and Brooklyn, where the resident population dropped by ~40% (Quealy, 2020; Tzortziou et al., 2022). In addition, the number of people commuting into the city for work (typically ~1,000,000 people daily), office occupancy, and total tourist visits plummeted in 2020 (NYC and Company, 2022; NYS Comptroller, 2021). These changes in population distributions and mobility have been shown to affect the local environment. Tzortziou et al. (2022) examined the impact of multiple waves and response phases of the pandemic on NYC's air quality and reported significant declines in atmospheric nitrogen dioxide (NO_2) closely following changes in the stringency of lockdown measures, and particularly changes in traffic.

In this study, we investigated the extent to which abrupt changes in human mobility and shifts in anthropogenic pressures during the height of the pandemic in 2020 and throughout 2021 in the NYC metro region affected water quality in surrounding estuaries (Fig. 1). To this end, we examined annual trends and anomalies in population and mobility, nitrogen loadings from wastewater treatment facilities (WWTFs), and meteorological data from NYC and Connecticut (CT) to assess their connections with water quality parameters retrieved using multiple high spatial resolution (30–300 m) ocean color remote sensing platforms. The use of remote sensing is emphasized here as it provides synoptic coverage of the study region at near-daily to weekly time scales and is particularly useful for periods when in situ monitoring was not feasible due to COVID-19 restrictions. Our study focuses on changes in water turbidity, colored dissolved organic matter absorption at 300 nm, $a_{CDOM}(300)$, and Chl-*a*, as these parameters can be linked not just to the flux of material from terrestrial sources, but also directly to anthropogenic activity (Paerl et al., 2006; Spencer et al., 2007) and can be reliably retrieved using satellite algorithms (Cao et al., 2018; Nechad et al., 2009; Sherman et al., 2023). In addition, we contrast the urbanized NYC region with two areas in CT, New Haven, and Mystic, where anthropogenic pressure is lower (i.e., smaller, and less dense populations with reduced industrial presence), and water quality is typically better (i.e., smaller, or nonexistent hypoxic extents and higher water clarity). These two CT municipalities were selected as mobility changes during COVID-19 were expected to be different, with New Haven representing a larger city (3rd largest city in CT), while the small town of Mystic is a major tourist destination that saw strong variability in the number of visitors throughout the pandemic.

2. Methods

2.1. Study region

NYC is surrounded by two estuaries of significant socioeconomic and ecological importance, the New York–New Jersey Harbor estuary (NY Harbor) and Long Island Sound (LIS) (Fig. 1). Over the last century, human activities in the NYC metro region have led to degraded water quality, including an increased frequency of harmful algal blooms, recurrent summertime hypoxia, and decreased water clarity (Anderson and Taylor, 2001; Greenfield et al., 2005; O'Neil et al., 2016; Taillie et al., 2020). Heavy nitrogen loading from NYC's municipal WWTFs has been identified as a primary cause of impaired local water quality (Latimer et al., 2014). Anthropogenic perturbations pose a significant threat to the NYC regional estuarine ecosystems and the ecological services they provide (Latimer et al., 2014; Taillie et al., 2020). To track the health of these ecosystems and the impacts of conservation and restoration efforts, several municipal and state agencies and local community groups conduct regular water quality monitoring programs, including the CT Department of Energy & Environmental Protection (CT DEEP), the NYC Department of Environmental Protection (NYC-DEP), and the Save the Sound Unified Water Study (UWS) program (Save the Sound, 2022). These programs, some with decades-long records, assess several key ecological indicators including water clarity, (e.g., turbidity, TSS), phytoplankton biomass (Chl-*a*), dissolved organic carbon (DOC), and DO.

In LIS, water quality exhibits a strong east-west gradient. In eastern LIS (ELIS), water quality is typically good, with overall clearer waters, lower concentrations of Chl-*a*, DOC and $a_{CDOM}(300)$, and increased DO (Turner et al., 2022; Sherman et al., 2023). Water quality in western LIS (WLIS), by contrast, is highly impacted by urban pollution, particularly in the Narrows region that is within close proximity to NYC (Fig. 1). Here, waters are

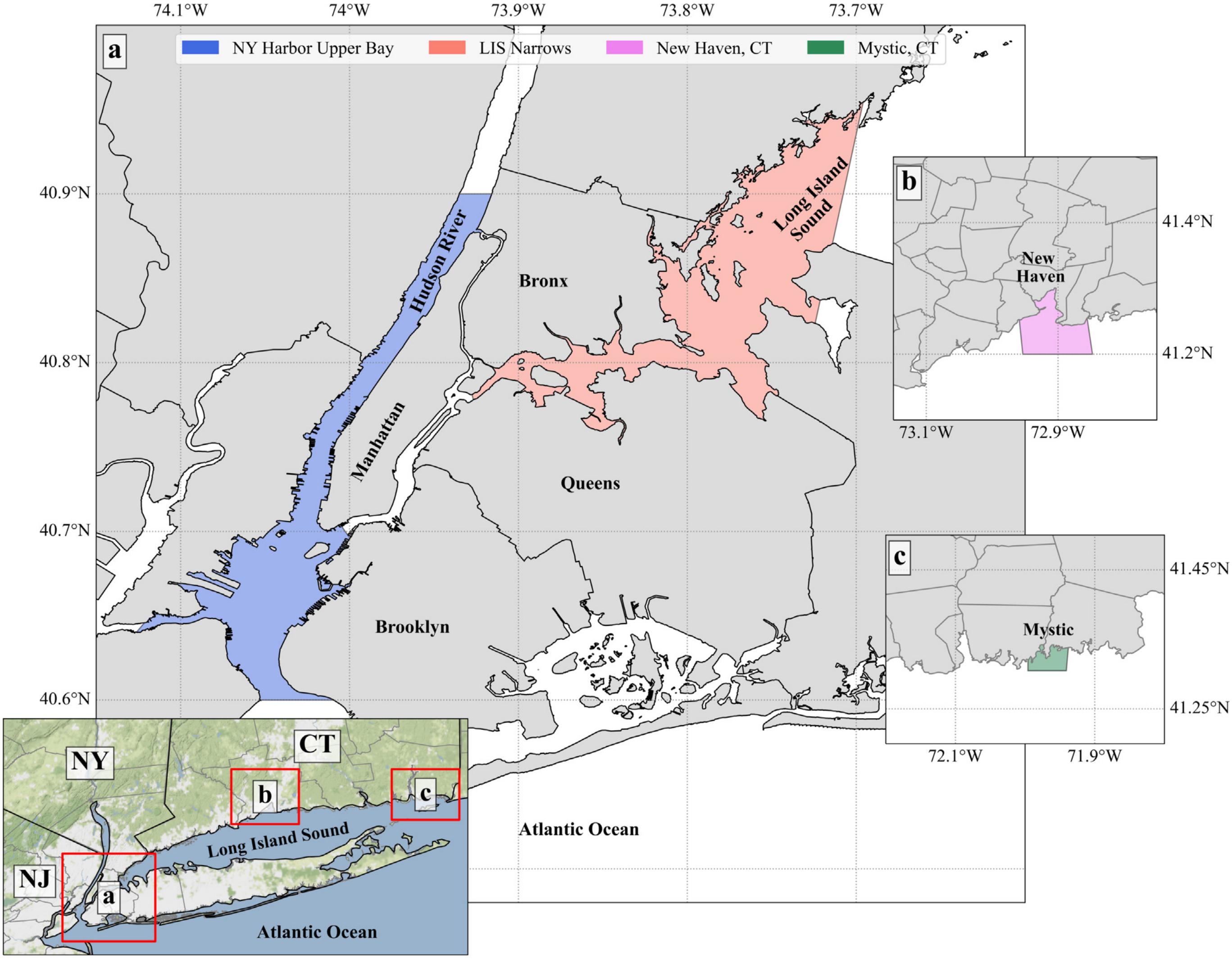

Fig. 1. Study regions. Map (a) shows the two estuaries surrounding NYC, NY Harbor Upper Bay, and LIS Narrows. Insets (b) and (c) show the regions of New Haven and Mystic, Connecticut, in central and eastern LIS, respectively. Shaded regions denote the area over which remotely sensed parameters in this study were averaged. Inset map in the bottom left shows the surrounding states of the study region, New York (NY), New Jersey (NJ) and Connecticut (CT) for reference.

less clear, Chl-*a*, particularly the relative contribution of smaller cells (Roldan Ayala, 2022), DOC and a_{CDOM}(300) are elevated, and seasonal hypoxia develops in the summer (CT DEEP, 2019; Humphries et al., 2023; Latimer et al., 2014; O'Shea and Brosnan, 2000; Sherman et al., 2023; Vlahos and Whitney, 2017). This gradient is attributed to approximately 3-fold higher nitrogen loadings from NYC (Latimer et al., 2014; Tetra Tech, Inc., 2018) and longer residence time in WLIS compared to the less populated ELIS, which has greater tidal exchange with the Atlantic Ocean. Water quality in NY Harbor also displays strong geospatial gradients, driven mainly by the extent of tidal exchange with the Atlantic Ocean. The best water quality is usually observed in the Lower Bay, at the mouth of the estuary, while in the Hudson River, Upper Bay and East River, water quality is more impacted by nitrogen loading and high turbidity. Such conditions limit light availability in the water column and in turn result in low Chl-*a* concentrations, despite the influx of nutrients, and limit benthic communities (O'Shea and Brosnan, 2000; Taillie et al., 2020). Between the two estuaries, there is a net flux of water into NY Harbor from LIS, primarily through the East River (O'Shea and Brosnan, 2000).

In 2001, the states of NY and CT, along with the U.S. Environmental Protection Agency (EPA), established maximum daily nitrogen loads with the goal of reducing total nitrogen loading by 58.5% through investments in WWTF upgrades and infrastructure (Long Island Sound Study, 2001). Even with these reductions, water quality surrounding NYC is still impaired, particularly in NY Harbor's Upper Bay and LIS Narrows. For example, it was shown that decreased nitrogen loadings have reduced the spatial extent of hypoxia in LIS, however no temporal trend in Chl-*a* concentrations has been observed since nitrogen reductions started (Whitney and Vlahos, 2021). Moreover, Whitney and Vlahos (2021) showed that hypoxia reductions are largely offset by climate-driven ocean warming due to reduced oxygen solubility, and that further loading reductions are needed to account for climate change impacts. Similarly, in NY Harbor, decreased nitrogen inputs are directly linked to improvements in some water quality indicators (e.g., increased DO), however, water clarity is still declining (Taillie et al., 2020).

2.2. Human mobility around NYC

To assess changes in human mobility in response to the COVID-19 restrictions, we assessed ridership changes for the primary railways used by commuters into NYC, the Long Island Railroad (LIRR) and Metro-North, as well as general ridership on the NYC Metropolitan Transportation Authority (MTA) subway system. Ridership information from March 2020 through December 2022, including total daily ridership for each mode of transportation and a percent comparison with same-day ridership in 2019 (i.e., pre-pandemic), for LIRR, Metro-North, and MTA subways were downloaded from NY State's OPEN data portal (https://data.ny.gov/). We also collected data from the U.S. Census Bureau's annual American Community Survey (ACS), which provides population and employment statistics in communities with >65,000 residents. From the annual ACS reports, we assessed the percentage of workers above the age of 16 that worked from home in the 31 counties considered within the NYC-metro area between 2017 and 2021. The Census Bureau did not release a standard annual ACS report for 2020 due to COVID-19. ACS data were queried using the Census Bureau's Application Programming Interface (API) in Python.

2.3. Wastewater discharge

Discharge monitoring reports for WWTFs in NYC were obtained from the EPA's Enforcement and Compliance History Online website (https://echo.epa.gov/trends/loading-tool/get-data/monitoring-data-download). Data from CT facilities were provided directly by CT DEEP. These data included monthly average flow rates (converted from gallon/day to m^3/day) and total nitrogen concentrations (mg/L), calculated as the sum of ammonia, nitrate, nitrite, and organic nitrogen. Total nitrogen loadings (kg/day) were then computed from the total nitrogen concentrations in combination with the flow rates. The WWTFs included in this study are NYC's Newtown Creek (NC) and North River (NR), which serve west and lower Manhattan and Brooklyn, and discharge into NY Harbor's Upper Bay, as well as Hunts Point (HP), Tallman Island (TI), Wards Island (WI), and Bowery Bay (BB) facilities, which serve Queens, the Bronx and the upper east side of Manhattan, and discharge into the upper East River and LIS Narrows. To compare the impacts of nitrogen loading on the two estuaries, we combined the loadings from facilities that discharge into each estuary. Data from CT included the New Haven WWTF (the largest facility in coastal CT), and the facility serving the village of Mystic (Fig. 1).

2.4. Regional meteorology

Meteorological data were downloaded from NOAA's Applied Climate Information System (ACIS) using an API query in Python. Data included gridded (5 × 5 km) daily precipitation, interpolated from the National Weather Service Cooperative observer network by the Northeast Regional Climate Center (DeGaetano and Belcher, 2007). This data source was chosen to align with annual summer hypoxia reports for LIS compiled by CT DEEP. Daily precipitation data were first summed to monthly cumulative precipitation for each calendar month and then spatially parsed and averaged for the four study regions: NY Harbor Upper Bay, LIS Narrows, New Haven CT, and Mystic CT (Fig. 1).

2.5. Remote sensing of water quality

Level-1 (L1) imagery from the Ocean and Land Color Instrument (OLCI) onboard the European Space Agency (ESA) Sentinel-3A/B satellites covering the period 2017–2021 were downloaded from NASA's Ocean Biology Processing Group (https://oceancolor.gsfc.nasa.gov/). OLCI provides near-daily coverage at 300 m spatial resolution. To derive remote sensing reflectance (R_{rs}) from the L1 data, we applied the POLYMER atmospheric correction algorithm (Steinmetz et al., 2011), which has been shown to perform better than other commonly used methods for OLCI in LIS (Sherman et al., 2023). R_{rs} was then binned into daily, 8-day, and monthly composites. In addition, clear scenes covering the NY Harbor and LIS Narrows between February 2020 and August 2021 from the Landsat-8 Operational Land Imager (OLI) and Sentinel-2 Multispectral Imager (MSI) sensors were downloaded from the USGS Earth Explorer data portal (https://earthexplorer.usgs.gov/). OLI and MSI were used to supplement the OLCI data with higher spatial resolution data (here processed to 30 m and 60 m, respectively). For consistency with the OLCI processing, POLYMER was also used to obtain R_{rs} for OLI and MSI, as early testing indicates good agreement with in situ R_{rs} collected across LIS (not shown).

Turbidity was estimated from R_{rs} following Nechad et al. (2009), using the equation

$$\text{Turbidity} = \frac{A_T * \rho_w}{1 - \rho_w / C} + B_T \quad (1)$$

where, ρ_W is the water leaving reflectance ($R_{rs} * \pi$) and A_T, B_T and C are empirical calibration parameters for the spectral band used in the derivation (655 nm for OLI and 665 nm for OLCI and MSI). Chl-*a* concentrations were derived from OLCI using a multi-spectral multi-linear regression algorithm developed for LIS (Sherman et al., 2023). $a_{\text{CDOM}}(300)$ was estimated from OLCI based on Cao et al. (2018) after reparameterization for LIS (Cao and Tzortziou, 2022). Both approaches to derive Chl-*a* and $a_{\text{CDOM}}(300)$ from OLCI have been optimized for LIS and have shown a high degree of accuracy in retrievals when validated with in situ measurements (mean absolute errors of 0.54 mg m^{-3} and 1.2 m^{-1} for Chl-*a* and $a_{\text{CDOM}}(300)$, respectively) and improved performance in the region compared to other commonly used algorithms (Cao and Tzortziou, 2022; Sherman et al., 2023). Turbidity algorithm performance showed a mean absolute error of 0.55 FNU and root mean squared error was 0.84 FNU when validated using in situ data collected in NY Harbor by NYC-DEP and in LIS by CT DEEP (see Sherman et al. (2023) for more details on matchup analyses and validation metrics).

2.6. Trends in 2020 and 2021

To evaluate changes in human mobility, environmental conditions, WWTF nitrogen loading, and water quality parameters during 2020 and 2021, we first calculated a monthly mean and standard deviation baseline for each dataset over the three-year pre-pandemic period from January 2017 to December 2019. We then compared the monthly mean values of 2020 and 2021 to the pre-pandemic baseline. To do so, we calculated the % change for each month In 2020 and 2021 following

$$\%\text{change} = \frac{X_{2020\ or\ 2021} - X_{2017\ -\ 2019}}{X_{2017\ -\ 2019}} * 100 \quad (2)$$

where $X_{2020\ \text{or}\ 2021}$ denotes the monthly average of a given parameter (e.g., precipitation) for a given month in either 2020 or 2021 and $X_{2017–2019}$ is the 2017–2019 baseline average value for that same month. Weekly % changes were also calculated for OLCI-derived parameters following the same approach. Note that for transit data provided by MTA, baseline data included only 2019.

2.7. Correlation analysis

Spearman's rank correlation coefficient (ρ) matrices were used to examine the associations between water quality parameters, nitrogen loading, and precipitation for each of the four study regions over 2017–2021. Spearman's coefficient was used as not all parameters assessed followed a normal distribution. Statistical significance of the correlation was determined using *p*-values at three significance levels (<0.001, <0.01 and <0.05). All statistical analyses were done in Python using common modules, primarily, NumPy, Scipy, Xarray and Pandas.

3. Results

3.1. Human mobility

Following the stay-at-home order on March 22, 2020, NYC experienced a massive decrease in the number of daily commuters entering the city. Commuter line and subway ridership fell to <5% of pre-pandemic levels, while bridge and tunnel use dropped by ~40% in April 2020 (Fig. 2). As restrictions relaxed, ridership in summer 2020 increased to ~20% pre-pandemic values, however rising COVID-19 cases in the fall and winter led to another, smaller decline. Pitiranggon et al. (2022) reported a similar

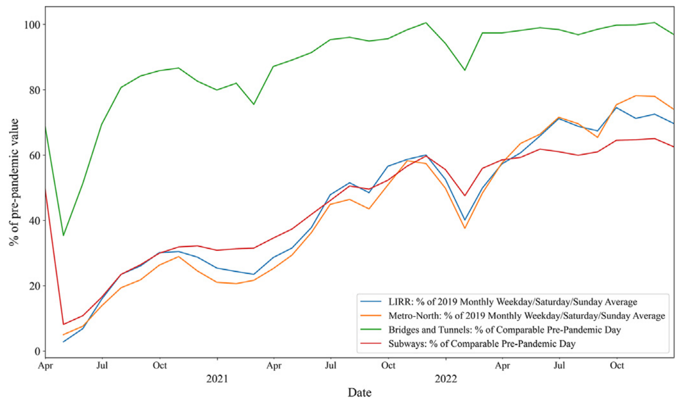

Fig. 2. 2020–2022 public transportation ridership percent changes from 2019 in main commuter lines, tunnels, and bridges into NYC, as well as overall NYC MTA subway ridership.

trend in weekly traffic volumes over the same period. Ridership has been steadily increasing since the beginning of 2021 and by January 2023 it has rebounded to ~60% of pre-pandemic values (Fig. 2). The sharp drop in January 2022 was attributed to the spread of the Omicron variant (CDC, 2020). Interestingly, the use of tunnels and bridges in and out of NYC recovered to 80% pre-pandemic values in May 2020, and by January 2022 had nearly fully rebounded (Fig. 2). The rapid recovery of bridge and tunnel use is attributed to the preferential use of private automobiles over more crowded public transportation throughout the course of the pandemic.

Changes in commuting patterns can also be attributed to surging prevalence of remote work. Data from the U.S. Census Bureau ACS survey showed that in 2021, the percentage of workers working from home rose significantly throughout the entire NYC metro region, with spikes of nearly 500% compared to the 2017–2019 baseline in counties immediately surrounding NYC, and counties further away from the city still exceeding 100% increases (Fig. 3).

3.2. WWTF discharge analysis

In NYC, two trends in WWTF nitrogen loading emerged following COVID-19 restrictions. Nitrogen loadings from the NR and NC facilities, which have sewersheds spanning west and lower Manhattan and Brooklyn and outflow into NY Harbor (Fig. 4a), abruptly and significantly decreased

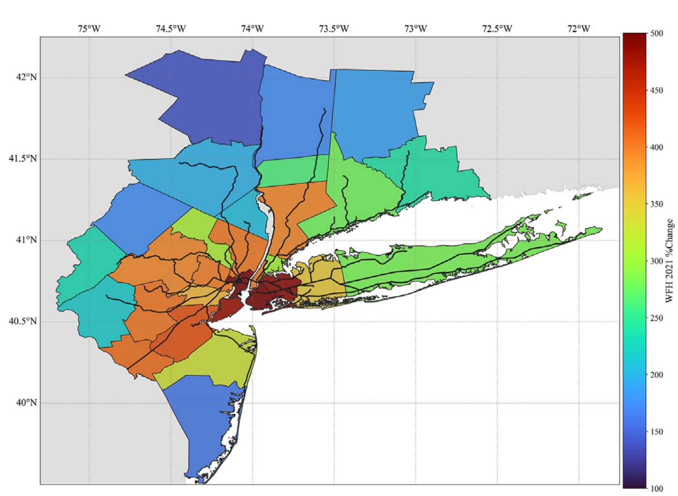

Fig. 3. Percent change of working population working from home in 2021 from the average of 2017–2019. Data from the yearly U.S. Census American Community Survey (ACS 1).

in April 2020 from ~10,000 kg/day to ~4000 kg/day in NR and from ~20,000 kg/day to ~12,000 kg/day in NC (Fig. 4a). When considered together, this is a nearly 50% decrease in nitrogen loading from wastewater treatment in the Upper Bay region compared to the baseline (Figs. 4a, 5). In contrast, in facilities that serve the upper east side of Manhattan, Queens, the Bronx, and western Long Island (WI, HP, TI, and BB) that discharge into the East River and LIS Narrows, changes in wastewater nitrogen loading in April 2020 were far less pronounced. Moreover, the overall annual trend was close to the range of regular seasonal and interannual variability (Figs. 4a, 5). Combined, these four WWTFs only experienced about a 10% decrease compared to the pre-pandemic baseline (Fig. 5). From May 2020 to December 2021, nitrogen loading into NY Harbor steadily increased, but still remained well below pre-pandemic values (Fig. 5). In contrast, nitrogen loadings into the East River and the Narrows returned to the baseline average by summer 2020 but then slightly fell again with recurrent COVID-19 waves during fall 2020 and winter 2021 (Fig. 5).

In the New Haven and Mystic, CT facilities, the 2017–2021 time series shows that by mid-2019 (well before COVID-19 restrictions) nitrogen loadings were lower. (Fig. 4b, c). When assessing changes in 2020, nitrogen loadings were below the baseline in January then increased into the spring. However, these below average values were still almost entirely within the baseline variability (Fig. 5). In New Haven, nitrogen loading started to decline in May 2020, and values were ~50% below the baseline, however this again mimicked the baseline seasonal trend and variability (Fig. 5). By summer 2020, nitrogen loading leveled off and remained closer to the baseline for the rest of the year. In Mystic, 2020 started ~60% below the baseline, but nitrogen loading started to increase in May and by late summer greatly exceeded the baseline. Following this summer peak, values returned close the baseline and within pre-pandemic variability (Fig. 5).

3.3. Precipitation

Overall, 2020 and 2021 were relatively drier years compared to the baseline. Across the four study regions, monthly cumulative precipitation in early 2020 was below or close to the baseline. From April to June 2020, concurrent with the first lockdown period, monthly precipitation was between 20% and 40% below the baseline and exceeded one standard deviation (Fig. 5). Storm events between July and October 2020 produced above average rain in NYC's Upper Bay and Narrows regions as well as further west into New Jersey. However, only in July the precipitation exceeded the baseline variability. Precipitation in New Haven and Mystic remained below average during the same months. Spring and the first half of summer 2021 were also drier than the baseline average, however several tropical storms and heavy rainfalls led to significant positive anomalies ranging from +100% to +200% (Fig. 5).

3.4. First COVID-19 wave impact

We first assessed whether any immediate changes in water turbidity were observable during the abrupt shift in anthropogenic pressure following the initial stay-at-home order. Select high resolution daily remote sensing observations of turbidity from MSI and OLI between March 16th and April 17th indicated an overall decline in turbidity in the Upper Bay (from an average of 6.2 to 4.2 FNU). In the Narrows, only minor variability was observed, and turbidity remained around 1 FNU (Fig. 6), in agreement with Schulberg and Subramaniam (2021). To supplement the limited temporal coverage offered by the MSI and OLI sensors, we evaluated 8-day composite imagery from OLCI over the same period. Good agreement in magnitude and spatial patterns of turbidity are observed between sensors, capturing the turbidity decrease in the Upper Bay and overall lower values with limited variability in the Narrows (Fig. 6). Comparison of the 8-day OLCI composites to 8-day composites from the same weekly period in 2017–2019 in the Upper Bay revealed a strong shift from average turbidity anomalies, decreasing from +58% in mid-March (pre-pandemic) to −40% in mid-April (after restrictions were imposed) (Fig. 6). In the Narrows, weekly turbidity anomalies were highly variable (Fig. 6).

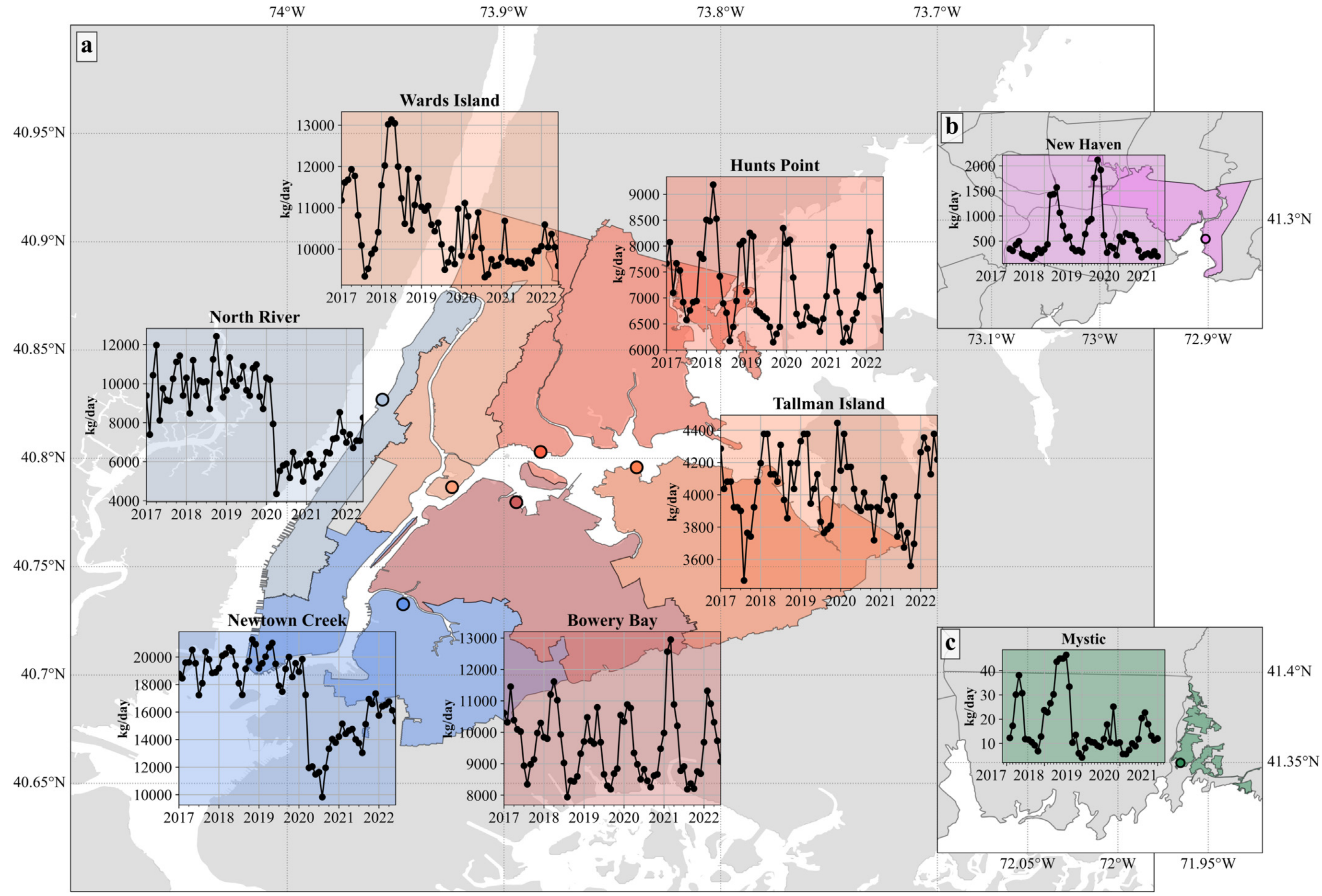

Fig. 4. Nitrogen loadings (kg/day) from WWTFs serving NYC between January 2017 to June 2022 (a), New Haven and Mystic, CT, between January 2017 to December 2020 (b) and (c), respectively. Circle markers denote the location of each facility. Shaded regions indicate the sewersheds of each respective facility. Similar color palettes in NYC facilities highlight WWTFs that discharge into the same estuary (blues into NY Harbor Upper Bay and reds into LIS Narrows). Note that the y-axis range is different between plots.

Likewise, weekly average $a_{CDOM}(300)$ in the Upper Bay decreased from 8.3 in 14–21 March to 6.4 m^{-1} in 15–22 April, a shift from $+25\%$ to -20%, respectively, relative to the corresponding weekly baseline. In the Narrows, $a_{CDOM}(300)$ values varied less and no trend was observed.

Chl-*a* in the Upper Bay was below the weekly baseline pre-pandemic (January–March 2020) and remained lower than baseline in April. However, the magnitude of the departure from the weekly baseline was higher in March compared to April (-30% to -15%, respectively) while an abrupt increase in Chl-*a* was observed later in spring and early summer (see next section). On the other hand, in the Narrows, Chl-*a* decreased from 9.37 to 5.2 mg m^{-3}. This decrease corresponds to a transition from Chl-*a* concentrations on average 25% higher than the baseline in March, during the occurrence of the annual spring bloom (CT DEEP, 2020), to an average of ~20% below the baseline throughout April. Furthermore, this decrease was also observed further east from the LIS Narrows region focused on in this study, with most of WLIS containing Chl-*a* values 20% or more below the baseline average for April (Fig. 7).

3.5. Extended impacts on water quality through 2021

3.5.1. Water clarity, turbidity and CDOM absorption

Extending observations of water quality over a longer period, shows that, compared to the 2017–2019 baseline, turbidity in 2020 and 2021 followed a similar seasonal cycle (Fig. 8). Yet, in the Upper Bay, mean monthly turbidity was 40% below pre-pandemic levels in April 2020 and remained lower than baseline and its variability until most of 2020 and early 2021 (Figs. 8, 9). In June 2021, turbidity increased close to pre-pandemic levels and exceeded the baseline ($+35\%$) in August (Figs. 8, 9). In the LIS Narrows, turbidity after the closures was only ~10% below the baseline and remained lower but close to baseline variability through summer 2021 (Figs. 8, 9). In New Haven and Mystic, there was less of a consistent trend in response to shutdown measures. Turbidity in New Haven was higher than normal from February to June 2020, then was very close to the baseline through the rest of 2020 until December 2021, when values dropped to 50% below baseline (Figs. 8, 9). A noteworthy exception was August 2020, when turbidity increased to ~50% above the baseline average and exceeded 1.5 standard deviations (Figs. 8, 9). In Mystic, turbidity values were below average pre-pandemic in early 2020, and then remained close to the baseline into the fall until November and December 2020, when turbidity greatly increased ($+25\%$ and $+62\%$, respectively; Figs. 8, 9). In 2021, turbidity in Mystic was close to the baseline, with a few months exceeding the range of variability, but with no discernible trend (Figs. 8, 9).

Trends in $a_{CDOM}(300)$ tracked with the observed trends in turbidity across the four study regions (ρ values ranging from 0.93 to 0.84, $p < 0.001$) (Figs. 8, 9). In the Upper Bay, monthly mean $a_{CDOM}(300)$ was considerably below baseline levels from April 2020 through June 2021. In the Narrows, $a_{CDOM}(300)$ values were below the baseline, although to a lesser degree, for the majority of 2020 and 2021, with the largest departures observed in September 2020 as well as April and September 2021 (Figs. 8, 9). No clear changes were observed in a_{CDOM} post-pandemic in New Haven or Mystic (Figs. 8, 9).

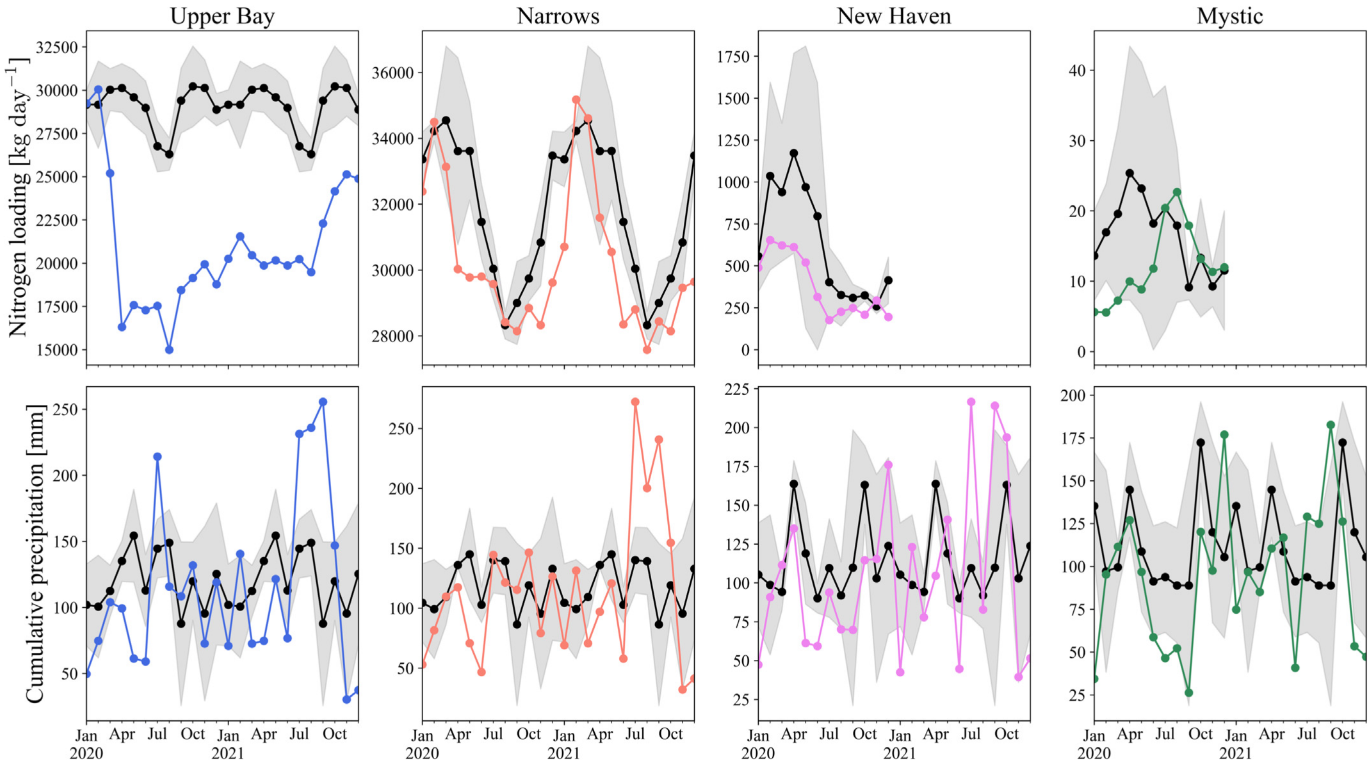

Fig. 5. 2020 to 2021 (colored lines) mean monthly nitrogen loading (kg/day) (top) and cumulative monthly precipitation (mm) (bottom) in the four regions of interest. Black lines and gray shading denote the average and standard deviation of the 2017–2019 baseline period.

3.5.2. Chl-a

Chl-*a* in the Upper Bay was negatively and significantly correlated with turbidity ($\rho = -0.78, p < 0.001$). While Chl-*a* levels were below the baseline across the region in winter 2020, concentrations abruptly increased to 36% above baseline post-pandemic in May 2020 and remained relatively high until July 2020 (Figs. 8, 9). Apart from September and November, where Chl-*a* levels were greater than baseline variability, the reminder of 2020 saw values similar to the baseline (Figs. 8, 9). In 2021, Chl-*a* remained close to baseline levels, with the exception of higher concentrations during the winter months (January–February), and particularly low (approx. 35% lower) concentrations in August (Figs. 8, 9). A different pattern was observed in the LIS Narrows, New Haven and Mystic, where Chl-*a* did not show a significant change from baseline immediately after the onset of the pandemic (i.e., late spring-early summer 2020). Chl-*a* in Mystic exceeded the baseline standard deviation in August 2020 while New Haven saw a large, although ephemeral, increase in Chl-*a* (+153%) in January 2021 (Figs. 8, 9). In summer 2021, increases in Chl-*a* surpassing the baseline standard deviation were observed in all three regions (Figs. 8, 9). Furthermore, in these three regions, the negative correlation with turbidity was weaker than that observed in the Upper Bay ($\rho = -0.34, p < 0.05$ in the Narrows and $\rho = -0.48, p < 0.001$ in New Haven and $\rho = -0.62$, $p < 0.001$ in Mystic).

4. Discussion

The COVID-19 pandemic had far-reaching impacts on the world. Social and economic norms were altered and, even with a “return to normal”, major cities are experiencing ongoing effects (e.g., decreased use of public transportation and commuting, increased work from home trends). As shown, these changes altered anthropogenic pressures on the environment. In NYC, COVID-19 led to dramatic changes in population dynamics during the height of lockdown periods in April 2020 and subsequent waves of the pandemic, which persisted even into the start of 2023 (Figs. 2, 3). While the “great exodus” from large cities projected early in the pandemic turned out to be far more muted by 2021 and 2022 (Hong and Haag, 2022), work-from-home trends have drastically reduced the amount of people in the city during the day (Fig. 3). The most impacted area was NYC's Manhattan urban core, particularly lower Manhattan, which includes the city's Financial District. This is evident in significant decreases in nitrogen loading from the wastewater facilities that serve this region (NR and NC) (Figs. 4, 5), in contrast with smaller declines observed in the city's other boroughs. In agreement, the Urban Green Council, a nonprofit organization, reported a decrease in residential and commercial water and electricity use across Manhattan, whereas use in other boroughs increased (Archambault, 2022). These results are consistent with observed changes in air quality, showing significant (up to 36%) post-shutdown NO_2 reductions in the NYC urban core, particularly in Manhattan, during the first three months after the initial COVID-19 lockdowns, and smaller declines in the surrounding areas of NJ, upstate NY, CT, and Long Island Sound (Tzortziou et al., 2022, their Fig. 2 and Table 1). Such changes in atmospheric composition could also be a driver of changing estuarine water quality, with prior studies indicating that atmospheric deposition accounts for 25% or more of the annual nitrogen load to systems such as Long Island Sound (Decina et al., 2020, Decina et al., 2017).

These disparate changes in anthropogenic pressure between Manhattan and other boroughs provide evidence for the contrasting water quality trends observed across different regions in this study. NY Harbor's Upper Bay, which receives wastewater from the facilities serving west and lower Manhattan, saw changes in water quality from April 2020 through the summer of 2021, most notably, an increase in water clarity (i.e., lower turbidity and $a_{CDOM}(300)$) (Figs. 8, 9), which has been historically low in this region. High turbidity in the waters of the Upper Bay strongly limits phytoplankton and benthic communities despite high nutrient flux (O'Shea and Brosnan, 2000; Taillie et al., 2020). Along with decreased turbidity in the Upper Bay, we observed increases in Chl-*a* in the late spring and summer 2020, suggesting an alleviation in light limitation (Figs. 8, 9). On the other hand, in the LIS Narrows, where turbidity is typically lower than in the Upper Bay, but still contributes to degraded water quality, little to no change in Chl-*a* was observed relative to pre-pandemic levels (Figs. 8, 9), with similar results for nitrogen loading (Figs. 4, 5).

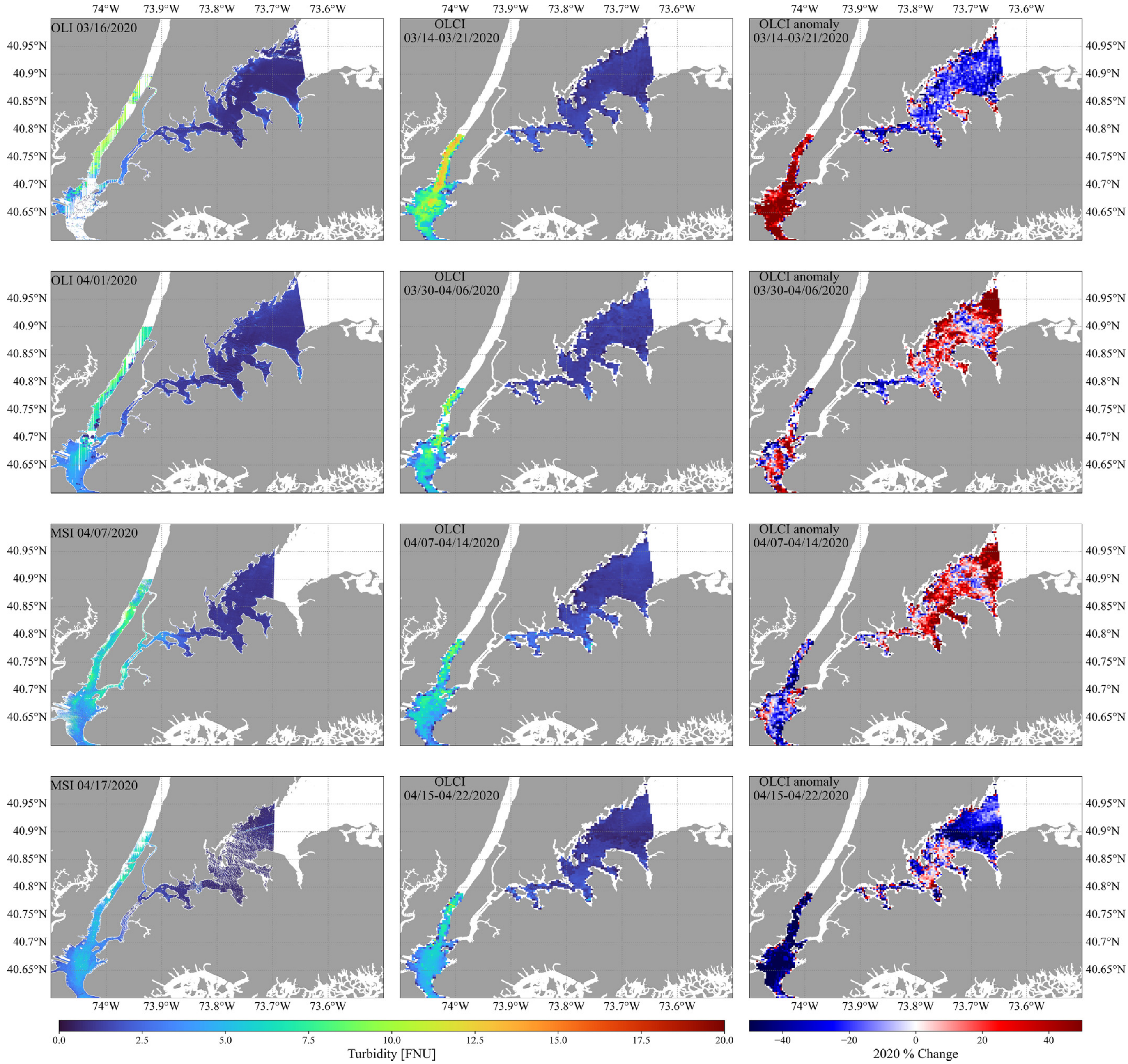

Fig. 6. Water turbidity in the Upper Bay and LIS Narrows between March 16 and April 17, 2020. Panels in the left column show daily OLI/MSI scenes, middle column panels show OLCI 8-day composites, and right column panels show the corresponding OLCI 8-day percent change from the 2017–2019 pre-pandemic baseline period.

In addition to changes in atmospheric and effluent nutrient and pollution inputs, changes in meteorological and hydrological conditions significantly impact nutrient and pollution loading into coastal waters (Sinha et al., 2017). While precipitation was on average 50% lower than the 2017–2019 average at the start of 2020 (Fig. 4), monthly turbidity was close to the pre-pandemic average and only decreased when nitrogen loading decreased in April (Figs. 8, 10). Furthermore, above-average increases in precipitation in the NYC region (e.g., July and September 2020) did not always have directly observable impacts on water quality (Figs. 5, 8), although the most extreme storm events did have significant and direct impacts. An increase in precipitation by 200% from the baseline due to tropical storm Henri in August 2021 and Hurricane Ida in September 2021 caused severe flooding leading to anomalously high turbidity in the Upper Bay (+35%), though it was short lived (Fig. 9). Aside from such transient events, our correlation analysis of the full 2017–2021 time series showed that precipitation was generally weakly associated with turbidity and $a_{CDOM}(300)$ in the Upper Bay ($\rho = -0.17$ and -0.08 respectively, $p > 0.05$). In comparison, nitrogen loading had a moderately strong and statistically significant positive correlation with turbidity and $a_{CDOM}(300)$ (ρ

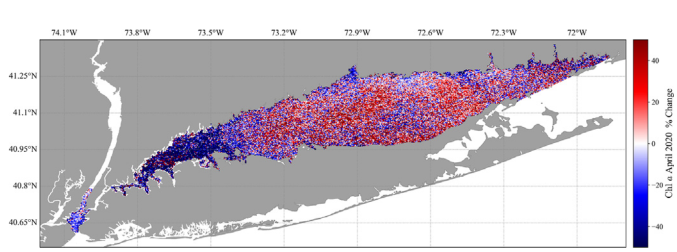

Fig. 7. Percent change in Chl-*a* from OLCI in April 2020 compared to the 2017–2019 baseline.

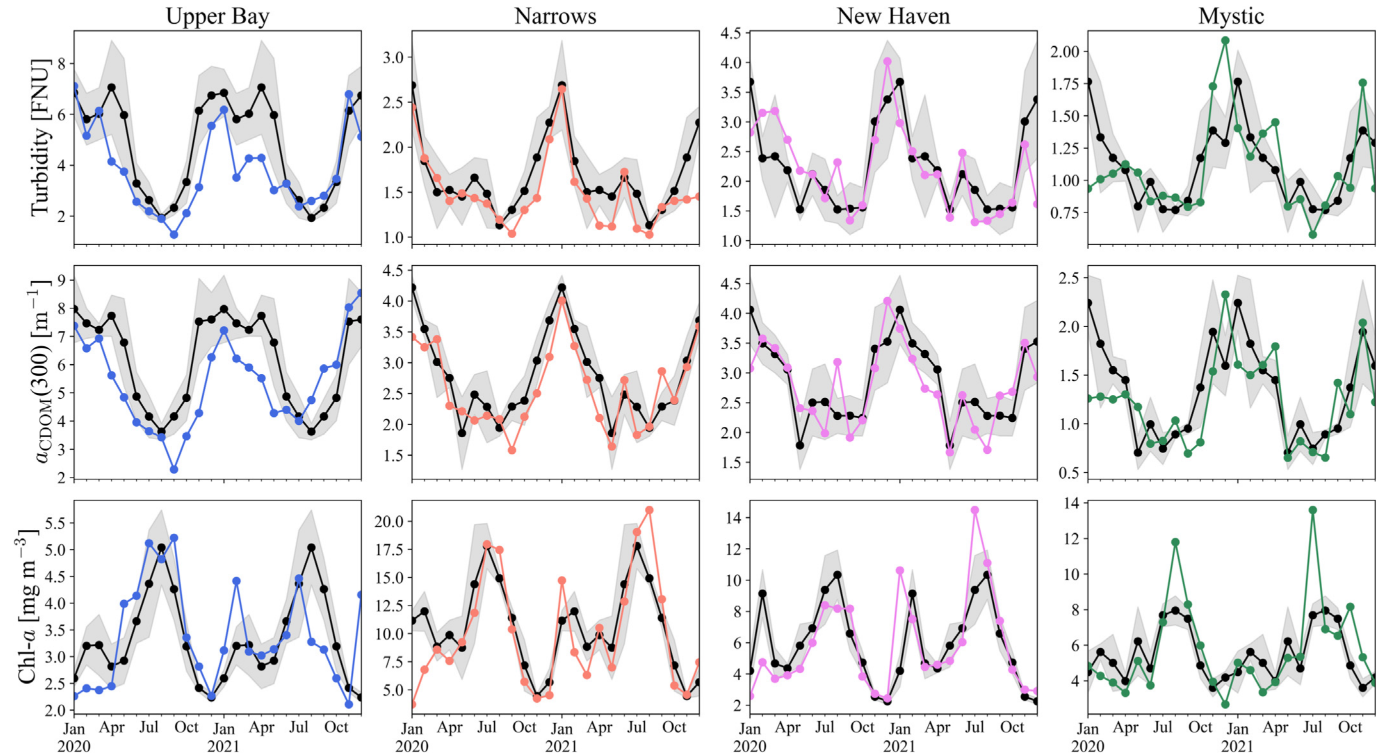

Fig. 8. 2020 to 2021 (colored lines) mean monthly turbidity (FNU) (top), $a_{CDOM}(300)$ (m^{-1}) (middle), and Chl-a (mg m^{-3}) (bottom) from OLCI in the four study regions. Black lines and gray shading denote the average and standard deviation of the 2017–2019 baseline period.

= 0.37 and 0.38, respectively, $p < 0.01$). This further supports our hypothesis that anthropogenic pressure was the primary factor driving these trends in water clarity.

We assessed similar impacts in CT, along the central and eastern sections of LIS. Here, estuarine water quality is historically better due to lower urban pressure and increased tidal exchange with the Atlantic

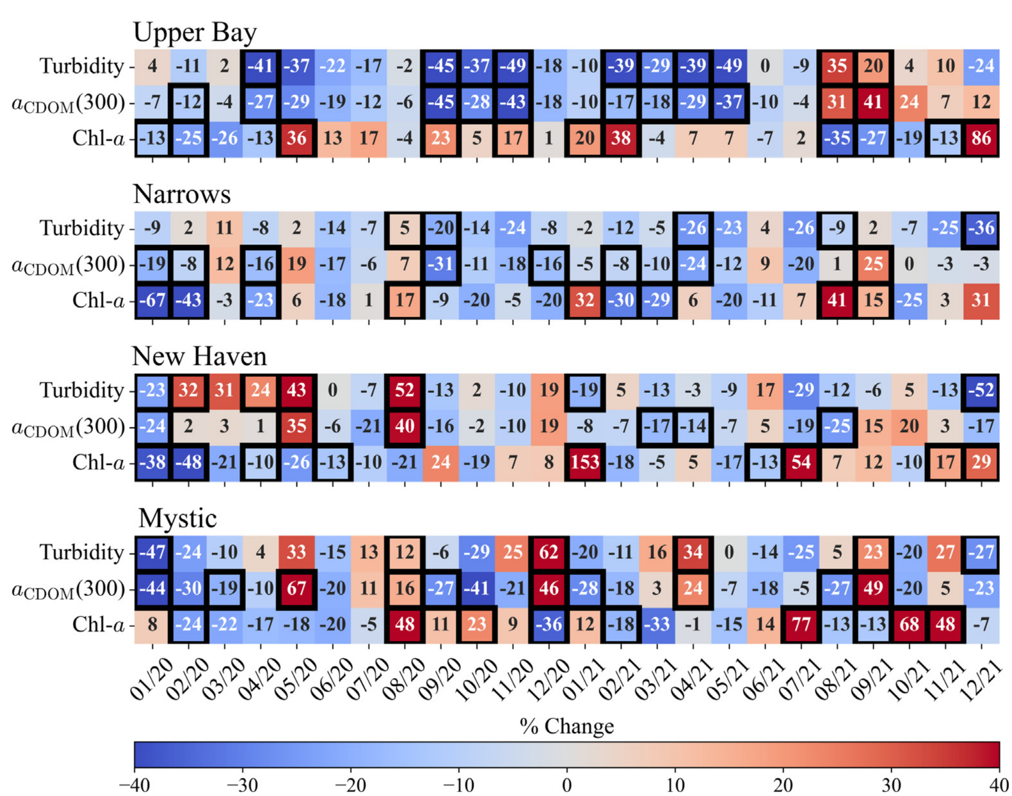

Fig. 9. Turbidity, $a_{CDOM}(300)$ and Chl-a monthly percent change in 2020 and 2021 from the 2017–2019 baseline in the four study regions. Bold cells denote a percent change in which a monthly 2020/21 value is >1.5 standard deviations from the baseline average.

Ocean (Latimer et al., 2014). In both of our study locations, New Haven and Mystic, CT, we observed less consistent trends in water quality in response to pandemic-related changes (Figs. 8,9), and non-existent or very weak, non-significant correlations between nitrogen loading and water clarity ($\rho = 0.08$ and 0.08 respectively, $p > 0.05$). In contrast to the NYC metro region, the WWTFs in these smaller cities process far less wastewater and output a fraction of total nitrogen (Fig. 4). While precipitation was only weakly correlated with water quality in the Upper Bay, we observed a more direct connection between precipitation and water quality parameters in these less urbanized locales ($\rho = 0.21$, $p > 0.05$ in Mystic). For example, in December 2020, when average precipitation was ~70% above normal in Mystic (Fig. 5), water turbidity and $a_{\mathrm{CDOM}}(300)$ increased and reached 62% and 46% above the baseline, respectively (Figs. 8, 9). A notable departure from this was observed in summer 2020, when the restrictions of the first COVID-19 shutdown were eased and more people were looking to travel locally. During this period, Mystic saw a large increase in tourism, with activity far exceeding pre-pandemic summertime levels (Hallenbeck, 2021). This agrees with the observed increase in nitrogen loading in August and September 2020 (Fig. 5) and may have contributed to the decrease in water clarity observed between August and October (Figs. 8, 9).

A particular aspect regarding water quality specifically in the Narrows and WLIS is the summer-time hypoxia dynamics (duration and spatial extent) and the conditions leading to it. Historically, the spatiotemporal extent of the seasonal hypoxia has been linked to the preceding spring phytoplankton bloom that typically occurs between February and April (CT DEEP, 2020; Lee and Lwiza, 2008). In 2020, high levels of Chl-*a* occurred in early March (pre-pandemic), with a maximum concentration of 23.1 mg m^{-3} measured in WLIS on March 5 (CT DEEP, 2020). However, the stringent lockdown measures imposed in response to COVID-19 interrupted routine in situ monitoring efforts following this early March sampling. Satellite OLCI retrievals, on the other hand, were continuous during the lockdown period, allowing to assess the spring bloom progression. Indeed, OLCI revealed that Chl-*a* in the western Sound was 20% lower than the pre-pandemic baseline in April following the observed bloom in early March (Fig. 7). In fact, satellite imagery showed that, unlike in the NY Upper Bay where turbidity decreased and Chl-*a* increased post-pandemic, in WLIS water clarity was far less impacted by COVID-19 shutdowns and Chl-*a* was lower than the 2017–2019 average over the entire spring bloom period (Feb-Apr). The overall weaker spring Chl-*a* bloom may have contributed to the shorter than average duration (43 days compared to a climatological mean of 53 days) and smaller spatial extent (43.5 km^2 with DO <2 mg/L compared to a climatological mean of 122 km^2) of hypoxia in WLIS during summer 2020 (CT DEEP, 2020). Summer wind conditions also relate to seasonal and interannual DO variations. Bratton et al. (2015) showed that elevated DO due to wind-driven vertical mixing is favorable with wind direction from 30° to 110° and wind speed ≥ 4 m s^{-1}. However, 15-min sea-level wind data in the eastern LIS Narrows in summer 2020 (NOAA National Data Buoy Center Station 44,022-Execution Rocks), showed that winds were predominantly from 170° to 210° and only <10% of the time did the wind direction and speed favor vertical mixing according to these criteria (Fig. S1). Storm events, and particularly the passage of Hurricane Isaias over LIS on August 4–5, 2020, could have contributed to storm-driven mixing and re-aeration in the water column, influencing LIS summer hypoxia dynamics (area and duration).

5. Conclusions

As stringent lockdowns were imposed across the globe during COVID-19, halting routine data collection efforts, remotely sensed imagery provided a favorable means of monitoring the environmental impacts of the pandemic with the potential to aid decision makers, and inform scientists and stakeholders across a variety of fields and disciplines. In this study, we evaluated the short and longer-term impacts that COVID-19 related changes in anthropogenic pressure had on water quality in highly urbanized estuaries surrounding NYC. Combining population, mobility, wastewater, and precipitation data with high spatiotemporal resolution remote sensing observations, we noted short-term improvements in water quality in response to stay-at-home orders in spring 2020 in NY Harbor's Upper Bay, where nitrogen loadings were significantly decreased from pre-pandemic levels. The continued pervasiveness of remote work and multiple waves of the pandemic into 2021 maintained lower atmospheric pollution and nitrogen loadings from WWTFs, resulting in improved water clarity in the Upper Bay. In the LIS Narrows and ELIS along the CT coastline, changes in anthropogenic pressure had less of an observable impact and trends in water quality were less clear. Our results support the possibility that significant decreases in wastewater discharge and nutrient loadings – as those observed in NY's Upper Bay immediately after initial shutdown measures – could lead to direct and immediate improvements in water quality.

COVID-19 shutdowns provided a natural experiment that offered a glimpse at direct and rapid impacts of curtailing anthropogenic pressures on the environment. This also reveals the opportunity for water quality managers to further push towards improvements that increase efficiency in wastewater treatment plants and decrease loadings (including from deposition of atmospheric pollution) which can directly impact marine productivity, biogeochemical cycling, and water quality. Lastly, this study highlights the multifaceted nature of coupled urban-human-estuarine systems and their heterogeneous responses to disturbances as extreme as a global pandemic.

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CRediT authorship contribution statement

Jonathan Sherman: Investigation, Methodology, Data curation, Formal analysis, Software, Visualization, Writing – original draft, Writing – review & editing. **Maria Tzortziou:** Conceptualization, Investigation, Methodology, Formal analysis, Project administration, Resources, Supervision, Funding acquisition, Writing – review & editing. **Kyle J. Turner:** Investigation, Data curation, Software, Writing – review & editing. **Dianne I. Greenfield:** Conceptualization, Investigation, Funding acquisition, Project administration, Writing – review & editing. **Alana Menendez:** Data curation, Investigation, Writing – review & editing.

Data availability

Data will be made available on request.

Declaration of competing interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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