

Physicochemical and urban land-use characteristics associated with resistance to precipitation in estuaries vary across scales

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Abstract. Estuaries are subject to frequent stressors, including elevated nutrient loading and extreme hydrologic events, which impact water quality and disrupt ecosystem stability and health. The capacity of an estuary to resist changes in function in response to precipitation events is a key component of maintaining estuarine health in our changing world. However, generalizable patterns in factors related to estuarine responses to extreme precipitation remain unknown. This knowledge gap complicates our ability to identify estuaries that are more likely to undergo major shifts in ecosystem services and predict the effects of urban and precipitation disturbances on estuarine water quality. We investigate which physicochemical and land-use characteristics are associated with ecological resistance to precipitation – defined as the magnitude of ecosystem change induced by an event – in five disparate estuaries distributed across the continental United States. Using long-term meteorological and water quality data from the National Estuarine Research Reserve System along with land use/land cover and population data, we examine relationships between the resistance index – a proxy for ecosystem stability calculated using dissolved oxygen – and physicochemical and urban land use characteristics on local-to-continental scales. Contrary to our initial hypothesis, we found that more urbanized estuaries were more resistant to precipitation events, possibly due to persistent disturbances on their baseline dissolved oxygen levels, and that water temperature, water column depth, turbidity, nitrogen, and chlorophyll-*a* were related to resistance on a continental scale. However, these continental-scale patterns were modulated by estuarine salinity and varied across individual estuaries, where additional relationships between resistance and salinity, phosphate concentrations, N:P, and tree cover emerged. These findings suggest that the relationships between urbanization and estuarine stability are complex and context-dependent, and thus management strategies need to consider both broad generalizations and local conditions. Considering emerging stressors from new environmental scenarios and from urbanization, these results are important for representing the impacts of disturbances in large-scale models and for informing management decisions regarding estuarine water quality.

1. Introduction

Estuaries are highly dynamic environments that often connect freshwater and saltwater systems, cycle organic matter and nutrients from land to oceans, and provide essential ecosystem services (Bianchi, 2007; He and Silliman, 2019). The function of estuarine ecosystems as unique sites for carbon and nutrient cycling, and habitats for macro- and micro-flora and fauna, relies on stability of a predictable range of dynamic processes like temperature fluctuations, hydrology and nutrient mixing. However, anthropogenic activities and extreme precipitation events threaten estuarine ecosystem stability and function (Kemp et al., 2009; Zhang et al., 2010). Predicted increases in urban population size and in the frequency and intensity of precipitation events highlight the urgency to understand the response of estuaries to urbanization and new precipitation patterns (Kyzar et al., 2021; Li et al., 2019; Martínez et al., 2007; Pickett et al., 2011). Yet, the factors that impact urban estuaries' response to precipitation are not fully understood.

When combined with intense precipitation, watershed urbanization and associated changes in land use/land cover (LULC) often result in increased stream hydrological flashiness (Gannon et al., 2022; Grimm et al., 2008; Reisinger et al., 2017). Hydrological flashiness induces high flow rates that cause changes in channel morphology (Booth and Jackson, 1997; Gregory, 2011; Leopold, 1968; Vietz et al., 2016), habitat destruction (Walsh et al., 2005), and disruption of microbial metabolic processes (Reisinger et al., 2017; Uehlinger, 2000). Flashiness can also drastically affect primary production – a regulatory component of dissolved oxygen (DO) dynamics in aquatic environments – through increases in flow velocity, transport of phytoplankton, sediment migration, and light limitation through increased turbidity (Bernot et al., 2010; Fisher et al., 1982; McSweeney et al., 2017; Reisinger et al., 2017; Uehlinger, 2000).

While DO is dynamic and depends on myriad biological, chemical, and physical processes, it is essential for many estuarine functions and has been widely used as an indicator for overall ecosystem health (Abdul-Aziz et al., 2007; Abdul-Aziz and Gebreslase, 2023; Chapra, 2008; Cox, 2003; Kannel et al., 2007; Zhi et al., 2021). Dissolved oxygen is critical to maintaining the life cycles of macro- and micro-fauna, and supports the biogeochemical cycling of carbon and nutrients by serving as a terminal electron acceptor (Bernhardt et al., 2018; Chapra, 2008; Zarnetske et al., 2012). Urban areas in particular have been linked to altered aquatic DO concentrations and other water quality parameters (Bernhardt et al., 2008; Chang, 2005; Freeman et al., 2019; Vietz et al.,

2016). Given the tight link between aquatic DO, land use, and diverse ecosystem functions, DO is an important indicator and holistic measure of estuarine health.

In addition to impacts on DO, extreme precipitation events can cause excess nitrogen (N) delivery and salinity change, particularly in waterways adjacent to urban-type LULC (Walsh et al., 2005). Nitrogen is central to mediating metabolic processes across a wide range of ecosystems and land covers (Mulholland et al., 2008; Schindler, 1977; Smith, 1984; Vitousek and Howarth, 1991). It is particularly important for primary production in aquatic ecosystems because many phytoplankton species are N-limited (Evans and Seemann, 1989; Howarth, 1988; Vitousek and Howarth, 1991). Further, extreme precipitation events can cause drastic changes in estuarine salinity through influx of freshwater (i.e., runoff) and/or saltwater (i.e., storm surges), which can induce stratification and hypoxia, and impact ecosystem metabolic functions (Rabalais et al., 2010; Wetz and Yoskowitz, 2013; Zhang et al., 2010). However, depending on the temporal and spatial scales of evaluation, the reported trends of estuarine responses to precipitation and salinity changes can be conflicting.

Identifying common responses to major precipitation events across disparate estuaries can help project long-term estuarine health under increasing urbanization and more extreme precipitation events. However, such commonalities have been difficult to discern and decipher, in part, due to the complex and interconnected dynamics within individual estuaries and their varied responses to precipitation. While Ombadi & Varadharajan (2022) report contrasting effects of urbanization on salinity during flood events when regional dynamics are considered, a continental-scale study by Kaushal et al. (2018) suggests that anthropogenic activity is associated with increasing salinity in waterways. However, Kaushal et al. (2018) recognize that regional, weather-related, LULC, and geologic variabilities also influence salinization patterns. Similarly, continental-scale evaluations showed that small watersheds appear consistently less flashy than large watersheds; and that there is a substantial amount of variability in these relationships at regional scale (Baker et al., 2004; Gannon et al., 2022; Hopkins et al., 2015; Poff et al., 2006). Such variation in relationships across scales may be particularly prevalent in ecosystems influenced by anthropogenic activities (Hopkins et al., 2015; Poff et al., 2006), which demonstrates the importance of considering multiple spatial scales in understanding estuarine responses to changes in precipitation patterns and watershed land use.

We aim to uncover generalizable responses to large precipitation events across five estuaries that span a gradient of urbanization and physicochemical characteristics. Using DO as an ecosystem health indicator (Abdul-Aziz and Gebreslase, 2023), we evaluate estuarine resistance to precipitation – defined here as the ability of estuaries to maintain DO stability (Isbell et al., 2015; Lake, 2013; McCluney et al., 2014; Pimm, 1984; Utz et al., 2016; Van Meerbeek et al., 2021). In the context of physicochemical and land-use factors, we evaluate estuarine resistance at: 1) the continental-scale (i.e., across all estuaries); 2) across estuaries grouped by salinity; and 3) within each estuary. We hypothesize that urbanization decreases estuarine resistance to precipitation, and that relationships between resistance and physicochemical and land-use factors will vary with spatial scale and ambient salinity. This study helps advance understanding of how ongoing changes in the Earth system influence estuarine ecosystem health.

2. Methods

2.1 Estuaries and monitoring locations.

We used long-term water quality monitoring data from five estuaries in the National Estuarine Research Reserve System (NERRS, 2023) to understand factors associated with their resistance to precipitation events. Lake Superior (LKS), WI; Chesapeake Bay Maryland (CBM), MD (Jug Bay only); Guana Tolomato Matanzas (GTM), FL; Weeks Bay (WKB), AL; and San Francisco Bay (SFB), CA span ecoregions, land uses, and salinity (0.1–35 ppt) (Fig. 1, Table 1). Across all estuaries, there were a total of 19 monitoring locations.

Briefly, LKS NERR is a freshwater estuary situated at the confluence of the St. Louis River and Lake Superior, that includes four continuous water quality monitoring locations influenced by lake seiche: Blatnik Bridge, Oliver Bridge, Pokegama Bay, and Barker’s Island. CBM NERR at Jug Bay, is a semi-diurnal minimally tidal estuary. Jug Bay is located in the upper tidal reaches of the Patuxent River, it is dominated by tidal freshwater marsh, and it comprises Railroad Bridge, Iron Pot Landing, and Mataponi Creek monitoring locations. WKB NERR is a partially-mixed (i.e., lagoonal/ semi-isolated), diurnal micro-tidal estuary located on the eastern shore of the Mobile Bay. WKB comprises Fish River, Weeks Bay, Middle Bay, and Magnolia River monitoring locations. SFB NERR estuary is a mixed, semi-diurnal tidal system comprising four monitoring locations across two embayments. The locations include China Camp and Gallinas Creek in the San Pablo Bay embayment within the China Camp State Park, and First Mallard and

Second Mallard in the Suisun Bay embayment within the Rush Ranch Open Space Preserve. GTM NERR is a mixed, semi-diurnal tidal estuary associated with the Tolomato and Guana River estuarine systems to the north and the Matanzas River estuary to the south. Monitoring locations include Pine Island, Fort Matanzas, San Sebastian, and Pellicer Creek, with the latter site noted to experience stratification (i.e., salt-wedge).

Upstream reservoirs are reported for several monitoring locations across the set of estuaries (Table S1).

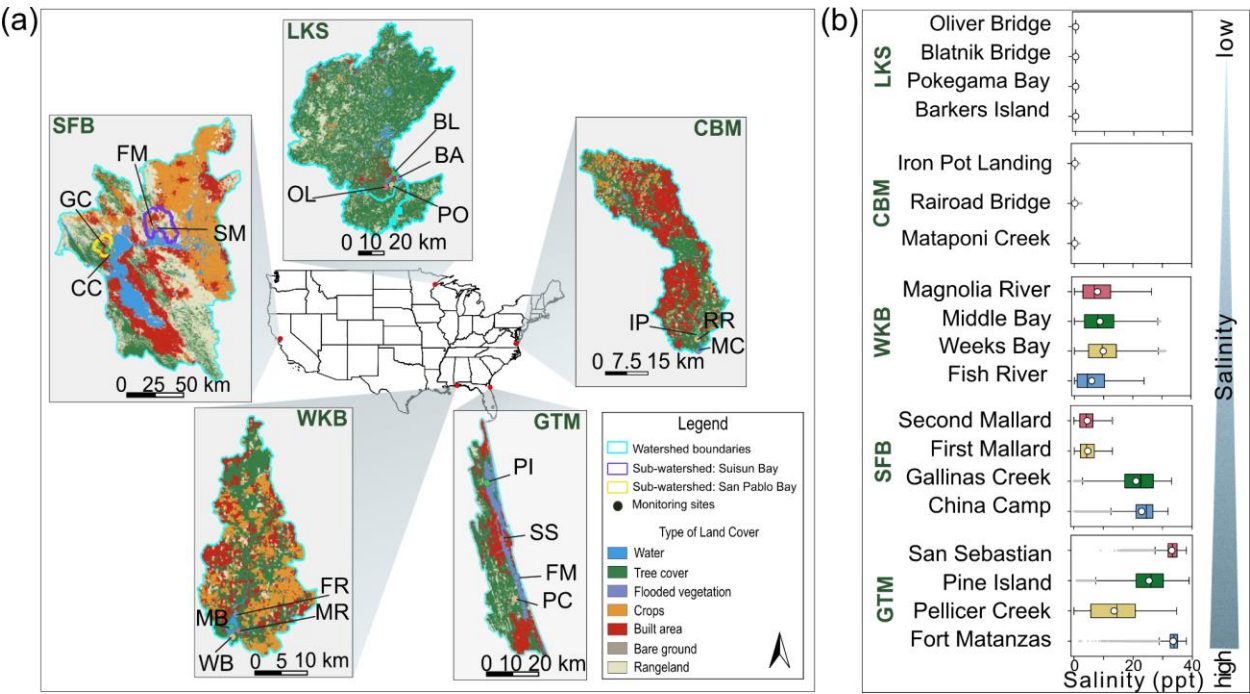


Figure 1. Selected National Estuarine Research Reserve (NERR) stations. a) Map of monitoring locations and land use/land cover within associated watersheds at Lake Superior (LKS) NERR, Chesapeake Bay, Maryland (CBM– Jug Bay) NERR, Guana Tolomato Matanzas (GTM) NERR, Weeks Bay (WKB) NERR, and San Francisco Bay (SFB) NERR. b) Salinity from 2012 to 2022 for each monitoring location at each NERR (all $n > 150,000$). Boxes indicate interquartile range. Means are shown in white circles. Black lines inside the boxes indicate medians.

Table 1. Land use/land cover (LULC) and population density in five National Estuarine Research Reserve System estuaries within 10-km to water quality and nutrient monitoring locations.

LULC class	Estuary and LULC class within 10-km to monitoring locations (%)					
	<i>Lake Superior (LKS)</i>	<i>Chesapeake Bay (CBM–Jug Bay)</i>	<i>Guana Tolomato-Matanzas (GTM)</i>	<i>Weeks Bay (WKB)</i>	<i>San Francisco Bay (SFB)</i>	
					<i>(San Pablo Bay)</i>	<i>(Suisun Bay)</i>
Water	10.43	2.30	8.58	4.85	5.04	3.59
Tree cover	58.78	53.72	45.11	38.52	26.91	1.16
Flooded vegetation	0.88	1.37	13.04	0.05	1.40	21.33
Crops	0.19	4.63	0.19	30.36	18.05	19.67
Built area	18.84	27.19	23.99	19.33	31.65	21.87
Bare ground	0.06	0.25	0.26	0.10	0.11	1.17
Rangeland	9.82	10.53	8.82	6.76	16.47	31.2
Surface area and population estimates.						
Area (km ²)	489	218	646	213	105	206
Estimated population within the area (ppl)	71,111	49,291	148,557	17,404	48,161	100,582
Population density (ppl km ⁻²)	145	225	230	81	458	488

LULC definitions per ESRI (Karra et al., 2021)

Water – areas where water was present throughout the year. Excludes man-made structures like docks.

Tree cover – vegetation with closed/dense canopy ≥ 15 meters.

Flooded vegetation – areas with intermixing of water and vegetation flooded seasonally or predominantly throughout the year.

Crops – human planted vegetation (cereals, grasses, and corps) that are not at tree height.

Built area – human made structures like roads, railroad networks, parking spaces, industrial and residential buildings.

Bare ground – areas dominated by rocks, soil, sand (i.e. desert) with sparse to no vegetation throughout the year

Rangeland – homogeneous grasses, mixes of vegetation below tree-height with rocks and soil, clearings in the forests.

2.2 Land use and land cover.

To assess the relationship between resistance and urbanization, we used LULC data at 10-meter resolution from ESRI (2017–2020) (Karra et al., 2021) and population density data at 100-meter resolution from World Population Hub (Bondarenko et al., 2020). The NERR watersheds shapefiles were obtained from NERRS, (2023). We analyzed LULC and population density using QGIS 3.30.3 (QGIS Geographic Information System, 2024) equipped with a semi-automatic classification plug-in, which generates percentages of each class within a given area based on the classification raster. The resulting breakdown of LULC and population density within watersheds is shown in Table S2.

Further, because the resistance index was calculated for precipitation events across short time-scales (days), large watershed areas could overrepresent hydrologic contributions to water quality at monitoring locations (watershed area range: 114 – 11,704 km², Table S2). To better match the temporal scale of precipitation responses, we restricted watershed characterization to a 10-km proximity zone surrounding each monitoring location. This approach captures watershed areas most likely to influence water quality over short time-scales, and improves comparability among estuaries (Table 1). LULC and population density were quantified within these proximity

zones. . For the SFB watershed, separate 10-km proximity zones were delineated for San Pablo and Suisun embayments to account for different hydrologic dynamics.

All data were collected using NERRS standard operating procedures. Briefly, water column DO (mg L^{-1} , calculated from % air saturation, temperature, and salinity at the time of measurement; thereby accounting for the impacts of temperature and salinity fluctuations on DO concentration), temperature, turbidity, salinity, and depth were measured at 15-min intervals using synchronized YSI-EXO2 multiparameter sondes. Meteorological conditions, including precipitation, were also measured at 15-min intervals using NERRS standard weather station instrumentation. PO_4^{3-} , NO_3^- , NH_4^+ , NO_2^- and chlorophyll-*a* (Chl-*a*) were measured monthly from grab samples and analyzed in the lab (NERRS, 2023). Samples for nutrients were analyzed following the U.S. Environmental Protection Agency (EPA) methods (O'Dell, 1996b, a; U.S. EPA., 1993a). For Chl-*a* analysis, the samples were collected as whole water, then filtered onto 0.45 μm glass-filters and processed following APHA, (2001) and U.S. EPA., (1993b) methods. While Chl-*a* is also measured as Chlorophyll fluorescence using optical sensor with YSI sondes, NERRS recommends using laboratory-based measurements due to the confounding effects of various fluorescent species, organism type, light, temperature and more on sensor-based Chl-*a* measurements (sensor and lab-based measurements often do not correspond to each other). We omitted all data flagged as 'suspect' or 'out of range'.

Additionally, we calculated water column depth as the sum of measured water depth plus the distance between the sonde and sediment bed (Table S1). We also calculated the sum of NO_3^- , NO_2^- , and NH_4^+ to assess dissolved inorganic nitrogen (DIN) concentrations and the ratio of DIN to PO_4^{3-} (hereafter, N:P).

2.3 Determination of major precipitation events.

Because hydrologic dynamics and related estuarine functions can vary dramatically with annual weather conditions, we first selected one 'wet' and one 'dry' year for each estuary using precipitation records from nearby airports. The purpose of selecting years with disparate rainfall patterns was to encompass the maximum range of variability in expected estuarine resistance. Following Murrell et al. (2018), we calculated the long-term interquartile range (IQR, 1990-2020) of total monthly precipitation for each estuary and then selected relatively wet/dry years based on the number of months plotting above/below IQR and total annual precipitation. Possible wet and

dry years were further filtered based on the completeness of NERRS data available for each estuary (Fig. S1). Long-term precipitation records included: Duluth International, Washington Reagan International, Jacksonville International, Birmingham, and San Francisco International airports available from the National Center for Environmental Information.

Further, we selected major precipitation events within each wet and dry year by plotting daily precipitation using data from NERR meteorological stations (Fig. S2). Specifically, because the definition of ‘major’ precipitation events is hard to quantify and it varies across estuaries, we first considered hurricanes, tropical storms, Nor’easters, atmospheric rivers, and other major storm events noted within NERR metadata sheets when selecting precipitation events. For example, at GTM we focused on tropical storms Colin, Julia, and Hermine, hurricanes Matthew and Irma, and two Nor’easters (Table S3). Data availability was a second consideration – we removed possible events for which there was a substantial amount of missing data. Lastly, because metabolic and hydrologic processes vary across seasons, we chose to focus on warm season events, with the exception of San Francisco Bay where most precipitation occurs in the cool season but seasonal temperature fluctuations are generally lower than in other systems (Fig. S2 and S3, Table S3).

2.4 Calculation of resistance.

To investigate physicochemical and urban land-use characteristics associated with estuarine responses to precipitation events, we calculated the resistance index described in Orwin & Wardle (2004). The resistance index is a normalized parameter (−1 to +1) describing the magnitude of shift in a response variable from an initial condition. It has been used across a wide variety of ecosystems and response variables, including aquatic ecosystems (Thayne et al., 2022, 2023; Tsai et al., 2011). The resistance index is calculated as:

$$Resistance = 1 - \frac{2|D_0|}{(C_0 + |D_0|)} \text{ (eq. 1)}$$

where, C_0 = concentration of the response variable pre-disturbance, and D_0 = difference between the concentration of the response variable pre- and post-disturbance (P_0) (i.e., $D_0 = C_0 - P_0$). An index value of +1 indicates the highest possible resistance. Index values between 0 and 1 show that the magnitude of response variable shift is less than the magnitude of the baseline (i.e., $|D_0| \leq C_0$). A resistance index of 0 indicates that the shift in the response variable is equivalent to the magnitude of the baseline (i.e., $|D_0| = C_0$), whereas index values between < 0 and -1 reflect that

change in the response variable is greater than the magnitude of the baseline (i.e., $|D_0| > C_0$). Overall, index values closer to 1 indicate more resistant systems (Orwin and Wardle, 2004).

While the resistance index is indicative of the ability of a system to maintain its pre-disturbance functions and ecological state (i.e., ecosystem stability), we emphasize that it is a normalized value that does not in itself convey information about overall estuarine health. Similarly, the resistance index alone cannot infer if a shift has a positive or a negative effect on ecosystem health. The resistance index enables the comparison of the amount of change induced by disturbance across vastly different estuaries.

In parallel, it is also useful to consider the absolute value of the response variable (in this case DO) pre- and post-disturbance, which conveys information on the ambient state of an estuary and the directionality of its response to the disturbance. We therefore present C_0 and P_0 (Fig. 2a–e, Table S4, Fig. S4) to define differences in DO within and across estuaries. We also present the resistance index, which pairs C_0 and P_0 values for the same event (Fig. 2f–j), to understand the ecosystem stability within and across estuaries after precipitation.

Because DO is critical to myriad functions that regulate the health of aquatic ecosystems (Abdul-Aziz et al., 2007; Abdul-Aziz & Gebreslase, 2023; Caffrey, 2004; Mulholland et al., 2001; Murrell et al., 2018; Odum, 1956) and because estuarine metabolism is virtually impossible to model due to bi-directional flow (Loken et al., 2021); we used temperature and salinity adjusted DO (mg L^{-1}) to calculate the resistance index. Additionally, the resistance index is highly sensitive to the researcher-defined baseline and post-disturbance time periods that are used to calculate C_0 and D_0 . Therefore, we calculated C_0 as average DO during a manually-curated timespan preceding each precipitation event (~24 hours to 6 days, without precipitation) where DO dynamics looked unremarkable (Fig. S5, Table S3). The timespan for estimating P_0 was also manually selected in the context of each event, defined here as the maximum displacement from C_0 during and after the event (Fig. S5, Table S3). We also verified that the resistance index calculated using DO (mg L^{-1}) versus DO (% sat.) showed no notable differences (Fig. S6).

2.5 Statistical analysis.

To compare resistance, nutrient concentrations, and concentrations of DO pre- and post-precipitation within and among estuaries, we used ANOVA with post-hoc Tukey HSD or Kruskal-Wallis test, as appropriate based on the Shapiro-Wilk normality test.

To test specific physicochemical and land-use factor associations with estuarine resistance, we used linear regressions at continental and local scales independently (i.e., all estuaries combined vs. within each individual estuary) and within salinity-based groups (i.e., using a threshold of average annual salinity less than or above 10 ppt calculated for wet and dry years separately). Specifically, we tested the relationships between resistance and turbidity, salinity, water temperature, water column depth, DIN, PO_4^{3-} , N:P, Chl-*a*, percent trees, percent crops, percent built area, and population density. For linear regressions that involved land-use factors, we used LULC and population density from the 10-km proximity zone (Table 1). Because of discrepancies in the temporal intervals at which measurements were collected, we conducted separate regressions involving turbidity, salinity, water temperature, water column depth (collected at 15-minute intervals) vs. DIN, PO_4^{3-} , N:P, and Chl-*a* (collected at monthly intervals) predictors. The low-salinity group included all LKS and CBM locations, First Mallard and Second Mallard locations at SFB during wet and dry years, as well as Fish River, Middle Bay, Magnolia Bay, and Weeks Bay (Weeks Bay during dry year only) at WKB estuary. The high-salinity group included all monitoring locations at GTM, China Camp and Gallinas Creek at SFB estuary for wet and dry years, and Weeks Bay at WKB during the wet year.

We used annual mean values for continental-scale, local-scale, and salinity-based groups regressions involving nutrients and Chl-*a* because monthly sampling intervals of lab-based measurements did not always correspond with selected precipitation events. This resulted in two data points per monitoring location for nutrients and Chl-*a* regressions. While not directly associated with any particular precipitation event, relationships of nutrients and Chl-*a* with resistance values analyzed on an annual basis carry valuable information about how the ambient conditions of the system can impact an estuary's response to precipitation. This knowledge is essential for deriving and testing hypotheses that describe why a certain estuary may respond to a storm event in a particular way.

For regressions involving sensor-based measurements, we attempted to provide as much resolution as possible into specific events. We therefore used values averaged over the event-specific baseline periods (Table S3) for sensor measurements across scales and salinity-based groupings. Values averaged over event-specific baseline periods were matched with corresponding event-specific resistance index resulting in one data point per precipitation event per each monitoring location. We did not include LULC and population density as predictors of resistance

at individual estuaries because of the close proximity of some monitoring locations to one another (< 800 m).

To ensure the robustness of linear regression analysis, we assessed the dataset for multicollinearity (Fig. S7) and conducted principal component analysis (PCA) with Varimax rotation on loadings (Fig. S8). For PCA, we used sensor-based, 15-minute interval measurements (temperature, salinity, turbidity, water column depth) and omitted the nutrient data collected at monthly intervals, to keep most data points in our analysis while remaining consistent with continental-scale linear regressions. Most variables were largely uncorrelated and the PCA results were in agreement with continental-scale linear regressions.

Statistical analyses and visualizations were performed in Python 3.10.11 using scipy, statsmodels, seaborn, scikit-learn, and matplotlib libraries.

3. Results

3.1 Land use/land cover and nutrient concentrations across estuaries.

The five study estuaries spanned strong gradients in LULC and nutrient concentrations. Both embayments of the SFB had higher percentages of urban-type land (e.g., built area) and population density than any other estuary, followed by CBM (Table 1). Agricultural land was more prevalent at WKB (30.36%) compared to other estuaries. GTM and LKS had more mixed LULC, with high proportions of tree cover. SFB and CBM had high DIN concentrations compared to LKS and GTM ($p < 0.01$). Mean DIN values at all SFB and CBM monitoring locations were $> 0.50 \text{ mg-N L}^{-1}$ versus $< 0.16 \text{ mg-N L}^{-1}$ at LKS and GTM (Fig. S9). Phosphate concentrations were the highest at SFB (mean across all monitoring locations = $0.135 \text{ mg-P L}^{-1}$, $\text{SD} = 0.08$; means at all other estuaries $< 0.03 \text{ mg-P L}^{-1}$, $p < 0.001$, Fig. S9). Overall, mean N:P across LKS, CBM, GTM, WKB, and SFB estuaries were 28.04 ($\text{SD} = 28.08$), 26.56 ($\text{SD} = 10.75$), 2.84 ($\text{SD} = 0.91$), 83.82 ($\text{SD} = 55.27$), and 5.10 ($\text{SD} = 1.4$), respectively (Fig. S10), with GTM and SFB indicating N limiting conditions based on a 16N:1P Redfield ratio (Redfield, 1934).

3.2 Changes in dissolved oxygen and resistance to precipitation across estuaries and monitoring locations.

DO concentrations pre- and post-precipitation differed across all estuaries, when evaluating the overall directionality (pre- (C_0): $F = 45.6$, $p < 0.0001$; and post- (P_0): $F = 17.1$, $p <$

0.0001, Fig. 2a–e). Across all precipitation events (i.e., non-specific to an event), SFB had the highest pre-precipitation DO concentration of all estuaries. Generally, at SFB and CBM (more urban) and at WKB (more agricultural), DO declined following precipitation ($p < 0.01$). At LKS, precipitation events significantly increased DO concentration ($F = 5.1$, $p = 0.03$). There was no significant difference between the overall pre- and post-precipitation DO concentrations at GTM ($F = 0.5$, $p = 0.48$).

Resistance also differed across estuaries ($F = 21.6$, $df1 = 4$, $df2 = 172$, $p < 0.0001$, Fig. 2). SFB monitoring locations had the highest mean resistance (mean = 0.68), while monitoring locations at LKS were the least resistant (mean = 0.47). Within individual estuaries, resistance varied across monitoring locations at LKS ($F = 13.9$, $df1 = 3$, $df2 = 24$, $p < 0.0001$), CBM ($F = 7.9$, $df1 = 2$, $df2 = 27$, $p < 0.01$), and SFB ($F = 10.2$, $df1 = 3$, $df2 = 27$, $p < 0.001$) but was not significantly different between monitoring locations within GTM ($F = 3.5$, $df = 3$, $p = 0.32$) and WKB ($F = 0.9$, $df = 3$, $p = 0.81$) (Fig. 2f–j). Resistance was most variable across monitoring locations at LKS (−0.26 to 0.89) and least variable across locations at WKB (0 to 0.73).

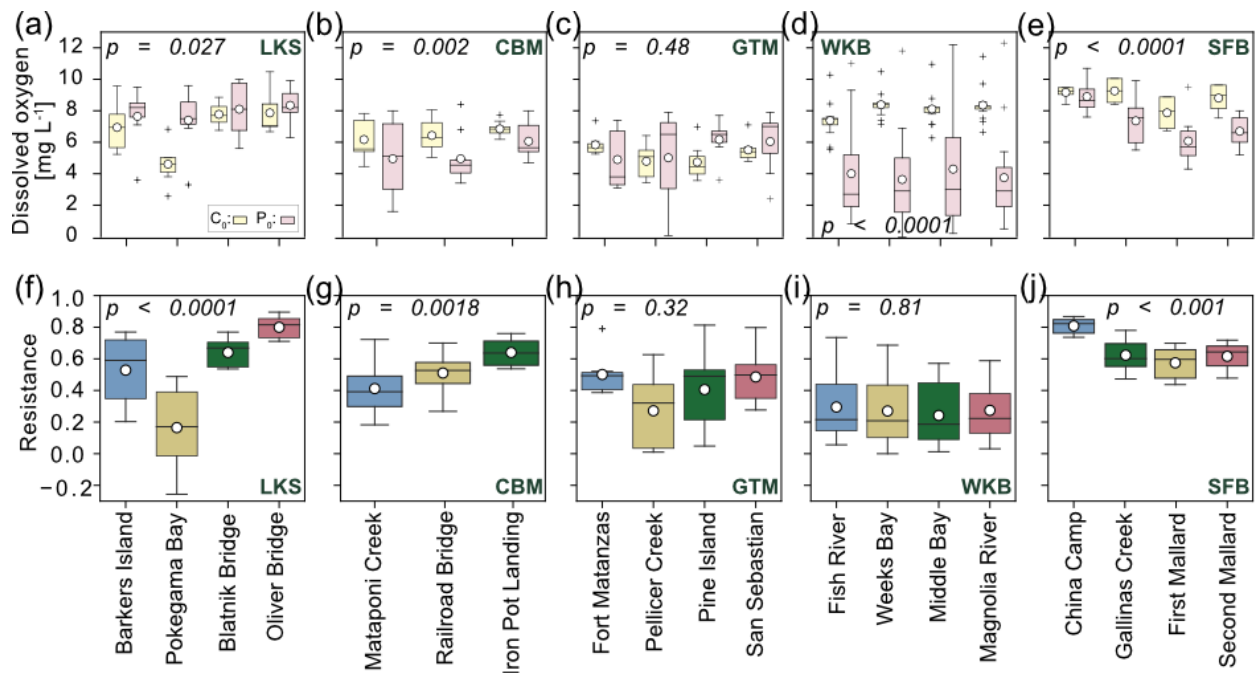


Figure 2: Variation in pre- and post-disturbance distribution of dissolved oxygen and resistance within individual estuaries. Boxes show the quartiles of the dataset and the whiskers show the rest of the distribution. Means are shown in white circles, and medians are shown in black solid lines. a-e) Distribution of dissolved oxygen concentrations prior to (C_0 , yellow boxes) and post (P_0 , pink

boxes) precipitation. f-j) Resistance index across monitoring locations at: Lake Superior (LKS) NERR (all $n = 7$), Chesapeake Bay, Maryland (CBM– Jug Bay) NERR (all $n = 10$), Guana Tolomato-Matanzas (GTM) NERR (all $n = 7$), Weeks Bay (WKB) NERR (all $n = 15$), San Francisco Bay (SFB) NERR ($n = 8$ except Second Mallard (SM) $n = 7$).

3.3 Continental, salinity-based, and local relationships between resistance and physicochemical factors, land use/land cover, and population density.

When data from the five estuaries were considered together (i.e., continental-scale), we found significant positive relationships between resistance and water column depth ($p < 0.0001$; $R^2 = 0.10$), turbidity ($p = 0.0015$; $R^2 = 0.06$), log(DIN) ($p = 0.031$; $R^2 = 0.12$), percent built area ($p = 0.02$; $R^2 = 0.14$), and population density ($p = 0.001$; $R^2 = 0.27$), as well as significant negative relationships with water temperature ($p < 0.0001$; $R^2 = 0.39$) and Chl-*a* ($p < 0.0001$; $R^2 = 0.39$) (Fig. 3, Table S5).

When grouped by salinity, estuarine resistance was more strongly related to physicochemical and land-use factors (Fig. 3, Table S5). Within ‘low-salinity’ estuaries, resistance was positively related to depth of the water column ($p < 0.0001$; $R^2 = 0.10$), which is consistent with continental-scale results; and negatively related to salinity ($p < 0.0001$; $R^2 = 0.21$), a relationship not found on continental-scale. Within ‘high-salinity’ estuaries, mean resistance showed positive relationships with annual mean log(DIN) ($p = 0.004$; $R^2 = 0.55$) and percent built area ($p < 0.001$, $R^2 = 0.73$), consistent with continental-scale results. Also, resistance in high-salinity estuaries was positively related to turbidity ($p < 0.0001$, $R^2 = 0.29$). Observations present in ‘high-salinity’ estuaries but absent from continental-scale evaluations included negative relationship between mean resistance and tree cover ($p = 0.011$; $R^2 = 0.46$), as well as negative relationship with N:P ($p < 0.0001$; $R^2 = 0.81$). Additionally, mean resistance in both low- and high-salinity groups was positively related to population density ($p = 0.051$; $R^2 = 0.16$, and $p < 0.001$; $R^2 = 0.73$, respectively), and negatively related to water temperature ($p = 0.001$; $R^2 = 0.38$, and $p = 0.002$; $R^2 = 0.60$, respectively) and Chl-*a* ($p < 0.001$, $R^2 = 0.46$, and $p = 0.044$; $R^2 = 0.32$, respectively). The temperature and Chl-*a* relationships were consistent with continental-scale results. Generally, ‘low-salinity’ estuaries had fewer significant relationships between resistance and physicochemical factors, LULC, and population density compared to ‘high-salinity’ estuaries.

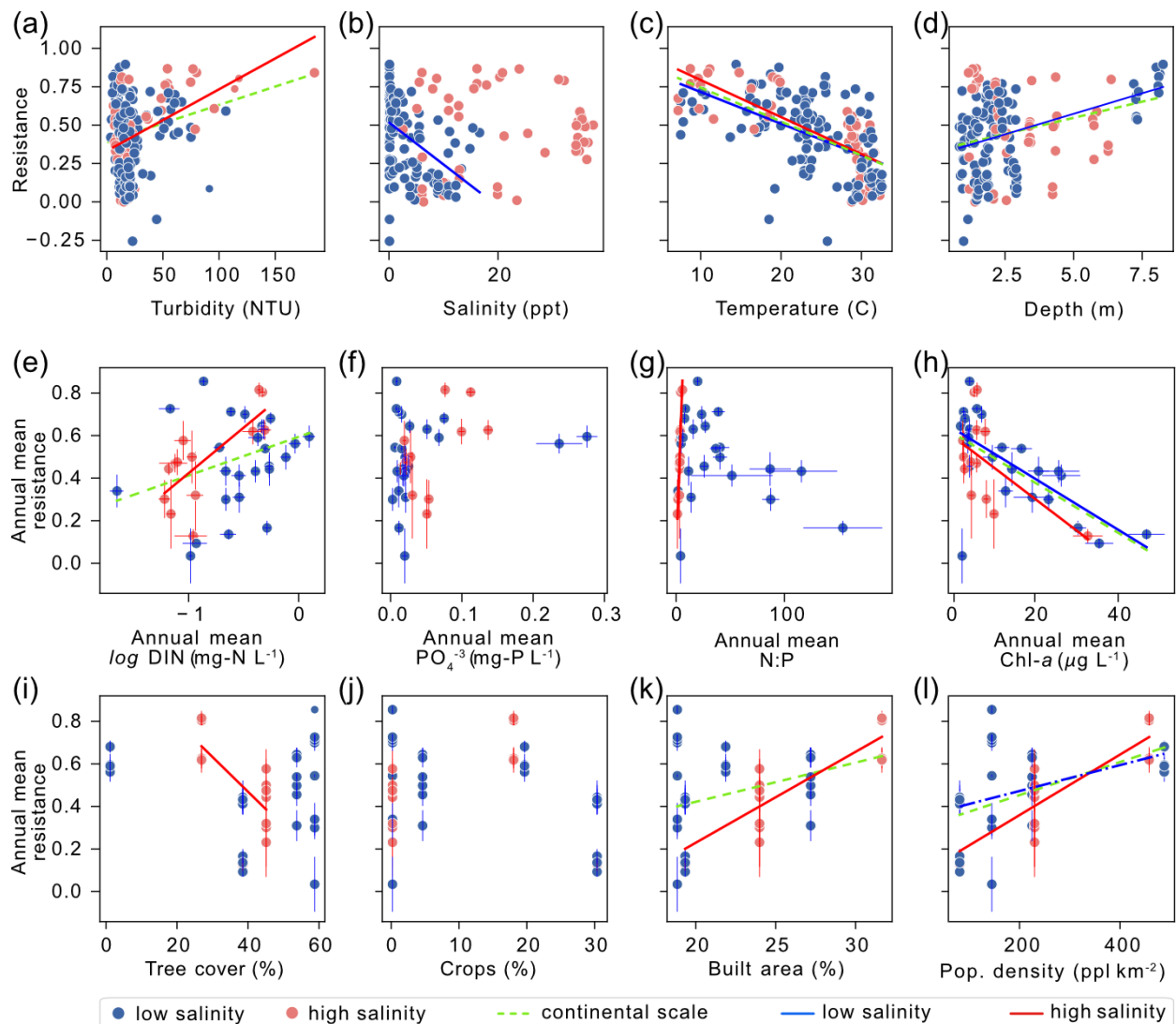


Figure 3. Relationships of continental-scale and salinity-based resistance with physicochemical factors, land use/land cover, and population density. Continental-scale regressions considered all monitoring locations across all estuaries. Significant relationships ($p < 0.05$) are shown in dotted green, solid red, and solid blue lines for continental-scale, high-salinity and low-salinity estuaries, respectively. Standard errors of the mean are shown in vertical and horizontal lines. The LULC parameters and population density were used from within the 10-km proximity zone adjoined to the monitoring locations. Please refer to Table S5 for resulting statistics and to Fig. S7 for information on covariance among predictor variables.

At local scales (i.e., within each estuary), resistance was related to some physicochemical factors not observed in continental-scale or salinity-based groups. The number, strength, and

direction of relationships between local-scale resistance and physicochemical factors varied substantially across estuaries (Fig. 4, Fig. S11, Fig. S12). GTM had the highest number of significant relationships between resistance and physicochemical factors. Resistance was negatively related to water temperature ($p < 0.001$; $R^2 = 0.52$), PO_4^{3-} ($p = 0.013$; $R^2 = 0.67$) and Chl-*a* concentrations ($p = 0.009$; $R^2 = 0.71$), as well as positively related to salinity and N:P ($p = 0.02$, $R^2 = 0.19$ and $p = 0.009$, $R^2 = 0.71$, respectively). In contrast, at CBM, resistance was related only to water column depth (positive, $p = 0.013$; $R^2 = 0.21$). At LKS, resistance was positively related to water column depth ($p < 0.0001$; $R^2 = 0.50$), and negatively related to turbidity ($p = 0.004$; $R^2 = 0.28$). At WKB, relationships between resistance and water temperature, salinity, and Chl-*a* concentrations were all negative ($p < 0.0001$, $R^2 = 0.30$; $p = 0.013$, $R^2 = 0.10$; and $p = 0.015$, $R^2 = 0.66$, respectively). At SFB, resistance was negatively related to water column depth ($p < 0.0001$; $R^2 = 0.48$), and positively related to salinity ($p = 0.00016$; $R^2 = 0.42$). There was no overarching relationship between resistance and total precipitation amount of each event, except for a significant but weak negative relationship at GTM estuary ($p = 0.026$, $R^2 = 0.18$, Fig. S13).

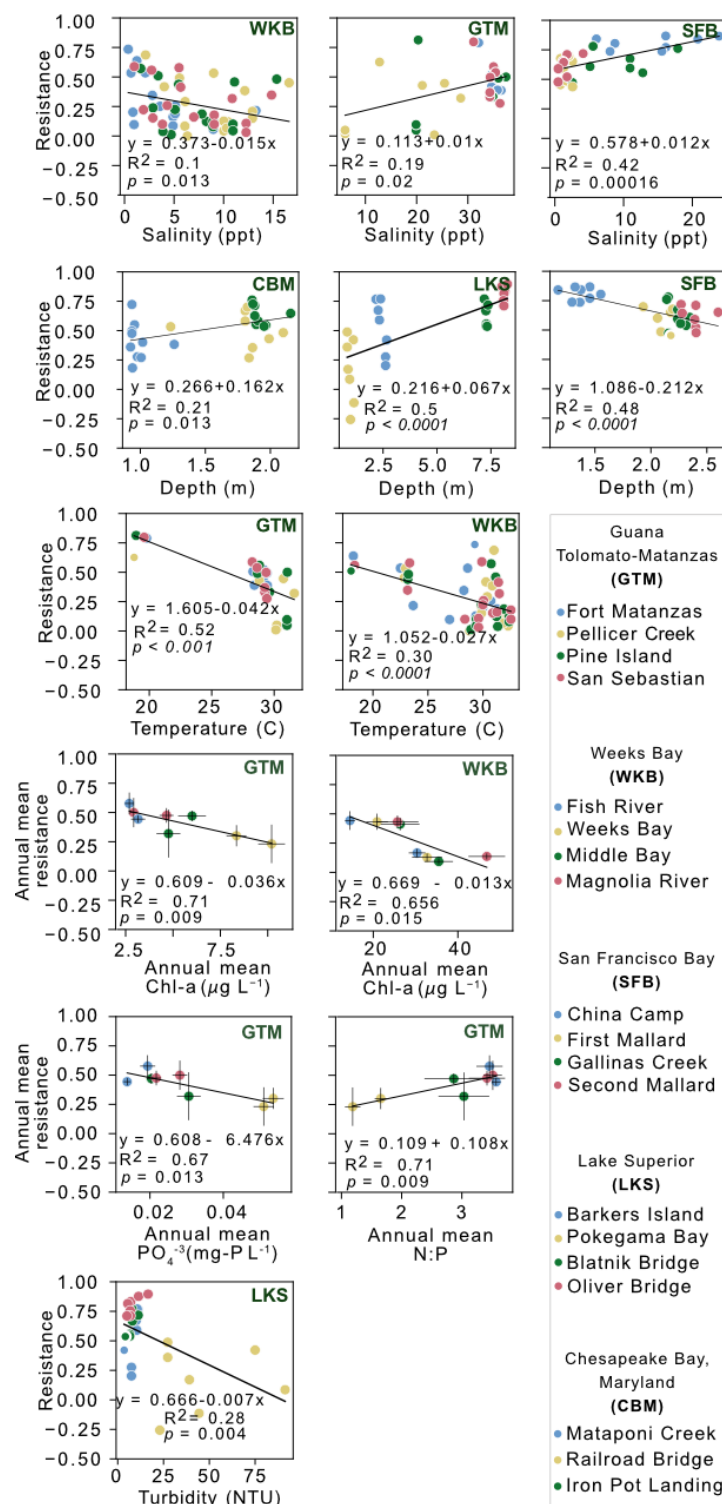


Figure 4. Relationships between resistance and physicochemical factors for each estuary. National Estuarine Reserve System (NERR) estuary abbreviations: Lake Superior (LKS) NERR, Chesapeake Bay, Maryland (CBM– Jug Bay) NERR, Guana Tolomato Matanzas (GTM) NERR, Weeks Bay (WKB) NERR, and San Francisco Bay (SFB) NERR. Significant correlations ($p <$

0.05) are shown with black lines. Standard errors of the mean are shown in vertical and horizontal black lines for relationships using annual means for chlorophyll-*a* (Chl-*a*), PO₄³⁻, and N:P. For additional results see Fig. S11, Fig. S12, and Fig. S14.

In summary, some relationships between estuarine resistance and physicochemical factors and urban land use appeared to be more universal while others varied across scales (Fig. 5). For instance, resistance was related to water temperature, water column depth, turbidity, and Chl-*a* across continental- and local-scales, and within salinity-based estuary groups. Dissolved inorganic N, percent built area, and population density were all related to resistance at the continental scale and in salinity-based groups. Both local and salinity-based evaluations revealed that N:P, salinity, and turbidity were related to resistance. Unique relationships included PO₄³⁻ at the local scale (GTM only) and tree cover in ‘high-salinity’ estuaries.

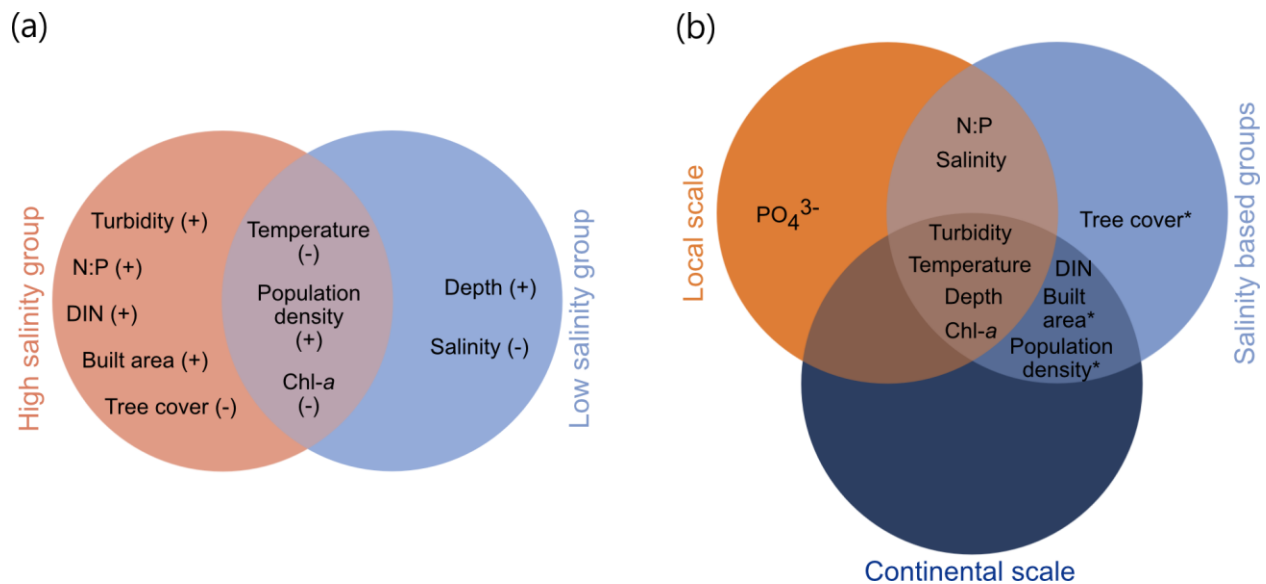


Figure 5. Cross-scale relationships between estuarine resistance and physicochemical and land-use factors. a) Venn diagram of resistance relationships with physicochemical and land-use factors in high- vs. low-salinity estuaries. Positive or negative relationships are indicated with ‘+’ and ‘-’ respectively. b) Venn diagram of relationships between resistance and physicochemical and land-use factors in continental, local, and salinity-based groups. Estuarine resistance with land use/land cover and population density marked with asterisks (*) were not evaluated at local-scale due to overlap in the spatial domains of some monitoring locations.

4. Discussion

Discerning estuarine response patterns following precipitation is important for predicting future impacts of urbanization and changing global weather patterns on estuarine ecosystems. Previous studies have shown that patterns identified at large scales may not be applicable across different ecoregions, geologies, and/or other ecosystem factors (Baker et al., 2004; Gannon et al., 2022; Hopkins et al., 2015; Kaushal et al., 2018; Ombadi and Varadharajan, 2022; Poff et al., 2006). Our results underscore the importance of cross-scale evaluations that can elucidate commonalities in estuarine response to precipitation, as well as variability in the factors associated with resistance across individual estuaries.

We show that while relationships between estuarine resistance, urbanization, and DIN may exist at the continental-scale, they do not consistently apply within individual estuaries. This is because an estuary's resistance depends on myriad specific factors in addition to many factors identified at larger scales. In contrast to our overarching hypothesis, we find that urbanized estuaries tend to have higher resistance than more pristine estuaries. These results suggest that the effects of watershed urbanization may impact estuarine stability by either providing some mechanism that allows estuaries responding to major precipitation events to dampen large shifts in DO; or by disturbing the baseline DO to an extent where even a major precipitation event would not produce a significant shift in DO, making the system appear highly resistant.

4.1 Dissolved oxygen dynamics differ between estuaries influenced by urban or agricultural land use and those with minimal urbanization.

There were vast differences in geometry, circulation, and hydrologic conditions among urbanized estuaries versus estuaries surrounded by agricultural land in this study. Yet, they exhibited similar patterns in DO response to precipitation. Precipitation generally reduced DO concentrations at WKB, SFB, and CBM (Fig. 2a–e). Simultaneously, the estuary with the least amount of urbanized LULC–LKS—experienced an overall increase in DO concentration following precipitation.

While a wide range of physical factors can impact DO in estuaries including channel geometry, river discharge, circulation, wind patterns, residence time, and upstream reservoirs (Kemp and Boynton, 1980; Raimonet and Cloern, 2017; Raymond et al., 2012; Raymond and Cole, 2001; Scully, 2010; Zheng et al., 2024), urban estuaries in particular are often impacted by

a combination of these processes. Urban estuaries often serve as basins for wastewater treatment outflows, which can supply continued freshwater discharge and nutrients during baseflow. Both urban and agriculturally influenced estuaries are prone to increased nutrient loading during and shortly after precipitation events (Chapin et al., 2004; Costanzo et al., 2003; Mallin et al., 2009), which impacts primary production and microbial metabolism and may lead to declines in DO concentrations (e.g., algal blooms). Similarly, power-generating dams and other large reservoirs often present in urban estuaries can strongly influence DO dynamics through their regulation of flow conditions, salinity, temperature, microbial processes, and solute residence time (Abbott et al., 2022; Ferencz et al., 2021b, a).

Therefore, closely examining the directionality of change in DO response to precipitation is important for informing the environmental significance of storm events for a specific estuary. An increase in DO following precipitation could reduce hypoxia through water column reaeration (Bianucci et al., 2018; Bohórquez-Bedoya et al., 2024), and have positive effects on nutrient cycling and growth conditions for macro- and micro- fauna (Harris et al., 2015). A decrease in DO following precipitation may reflect the opposite effects (e.g., hypoxia), leading to changes in microbial community structure, disruptions in nutrient cycles, and fish die-off (Diaz et al., 1992; Llansó, 1992). Further, the extent to which ecosystem health is linked to DO changes is intertwined with its ambient ecological state and the magnitude of the DO change. An increase in DO following a storm may be less consequential for estuaries with DO-replete ambient conditions as compared to estuaries with chronically depleted DO. This implies that a comprehensive assessment of storm impacts on estuarine ecosystem stability and health, should consider resistance index alongside historical DO conditions and the magnitude and directionality of DO change.

For example, resistance index values for the more pristine and tree-cover dominated Pokegama Bay at LKS was comparable to most monitoring locations at WKB, which were associated with agriculture. However, Pokegama Bay and WKB experienced different patterns in DO change following precipitation (Fig. 2). While the full nature of the causes and impacts of DO change in each estuary is beyond the scope of this study, the differences in absolute DO change suggests that different underlying processes influence stability in estuarine dynamics in response to precipitation across different estuaries. Our goal is to identify common predictors of estuarine resistance at the continental scale despite substantial variability in processes, while also identifying

salinity-dependent and estuary-specific predictors that help elucidate dynamics that may be particularly influential in individual systems.

4.2 Urbanization and inorganic nitrogen correspond with elevated resistance to precipitation at the continental-scale.

Higher resistance to precipitation in the most urbanized estuaries and overarching relationships between urban LULC and resistance across all estuaries reflect that estuaries within urban watersheds typically experience relatively small shifts in DO following precipitation (Fig. 2, Fig. 3). These results contradict our hypothesis that urban estuaries should show low resistance to precipitation because of greater physical and chemical disturbances like flashiness, streambed scouring, N loading, and turbidity that impact DO dynamics (Bernhardt et al., 2008; Groffman et al., 2004; Hession et al., 2003; Hopkinson & Vallino, 1995; Walsh et al., 2005).

It is possible that watershed urbanization could equip estuaries with adaptations that help dampen precipitation impacts on shifts in DO. For instance, an increase in flashiness in waterways would increase flow velocity and water column reaeration (Raymond et al., 2012; Raymond and Cole, 2001) and contribute to phytoplankton removal via transport or turbidity-driven light attenuation (Caffrey, 2004; Pennock and Sharp, 1986). This impact may be particularly important for estuaries whose baseline DO conditions are influenced by algal blooms, which induce large diel DO fluctuations by overproducing DO during the day and severely depleting it at night (Chapin et al., 2004; Ni et al., 2020). Managing phytoplankton overgrowth could help maintain DO closer to baseline conditions. Supporting this explanation, turbidity-driven limitations on phytoplankton were previously reported for the SFB estuary (Cloern, 1987). We also found that turbidity was positively related to built area, Chl-*a* was negatively related to population density, and PO_4^{3-} concentration was negatively related to tree-cover (Fig. S7, Fig. S15). Tree-populated riparian zones, often reduced or absent in urban environments, can help regulate phytoplankton overgrowth and DO dynamics by significantly reducing PO_4^{3-} concentrations (Keller and Fox, 2019).

Relatively high levels of ambient N and low ambient DO in urban estuaries may also facilitate comparatively small DO fluctuations in response to precipitation. Here, estuarine inorganic N concentrations were related to urbanization and also were a major predictor of resistance at the continental-scale (Fig. 3, Fig. S15). While moderate levels of N support biological

metabolisms (Howarth, 1988; Howarth and Marino, 2006; Vitousek and Howarth, 1991; Zhang et al., 2021), nutrient loading, including N, is a significant problem for urban aquatic environments (Beman et al., 2005; Bernot et al., 2010; Bettez et al., 2015; Black et al., 2011; Hopkinson and Vallino, 1995; Mulholland et al., 2008; Reisinger et al., 2016). For urban estuaries, where N loading is associated with chronically low DO availability, this could mean that precipitation-driven increases in DO under simultaneous high N input would be small, which would result in higher resistance. Nitrogen limitation, in contrast, is prevalent in coastal marine systems and can negatively impact overall estuarine health by constraining biological activity (Elser et al., 2007; Guildford and Hecky, 2000; Paerl, 2018; Paerl and Piehler, 2008). Relationships of resistance with N and urbanization were particularly evident in more coastal (high-salinity) estuaries where we found positive relationships between resistance, N:P, and built area (Fig. 3). Thus, altered N dynamics in urban estuaries may be key to understanding their comparatively low change in response to precipitation.

While the mechanisms behind positive relationship between resistance and N are not clear, changes in concentrations of NH_4^+ were previously suggested to regulate algal blooms in the San Francisco Bay estuary (Dugdale et al., 2007; Parker et al., 2012). Elevated NH_4^+ concentrations resulting from changes in water-treatment practices in San Francisco were shown to inhibit NO_3^- uptake and suppress algal blooms, thereby altering a major control on DO dynamics in SFB. Therefore, the prevalent N species and shifts in DIN composition following precipitation events should be considered in attempts to explain the underlying mechanisms behind the positive relationship between N and resistance.

Alternatively, we also consider if baseline DO concentrations in urban estuaries are already disturbed to such an extent that precipitation events cause minimal further disruption. Urbanization itself can promote large diel DO fluctuations, as suggested by Gold et al.(2020), and if a precipitation event induces fluctuations that are similar in magnitude, the post-precipitation DO concentrations will not deviate significantly from baseline conditions. Under such a scenario, urban estuaries will appear more resistant to precipitation events compared to more pristine systems.

4.3 Water column depth, temperature, turbidity, and Chl-*a* are related to resistance across all scales.

We found generalizable patterns (across all scales), in which resistance was positively related to water column depth and turbidity, and negatively related to water temperature and Chl-*a* (Figs. 3–5). As discussed above, phytoplankton dynamics may be an important factor in regulating DO in response to disturbance due to their tight linkage with N loading, which is consistent with results previously reported by others (Thayne et al., 2023). This is further underscored by the existence of a relationship between Chl-*a* and resistance across continental, salinity-based, and local analyses. In parallel, the relationship between resistance and turbidity was positive at the continental scale and in the high-salinity group (Fig. 3), while at LKS resistance and turbidity correlated negatively (Fig. 3, Fig. 4). The positive relationship between resistance and turbidity and simultaneous negative relationship with Chl-*a* can be attributed to turbidity-driven light attenuating conditions that restrict phytoplankton growth, which is further confirmed through negative correlation between turbidity and Chl-*a* (Fig. S7).

Water column depth can also influence estuarine resistance to precipitation through physical and hydrologic impacts on DO. Through dilution, deeper estuaries may better resist hydrologic changes following precipitation, for instance by attenuating the effects of oxygen-saturated rainwater, nutrient loading, turbulence, and water-atmosphere gas exchange. Similarly, deep estuaries have longer equilibration time with environmental conditions and less diel variability in parameters like temperature (Caissie, 2006; Macan, 1958). These characteristics also influence diel and seasonal DO dynamics, potentially resulting in more stable DO baseline and more moderate responses to precipitation. Deeper estuaries can also be associated with urbanization, which shows high resistance in this study, because shifts towards more urban and agricultural LULC generally deepen estuarine channels through increased runoff, sediment transport, hydrological flashiness, channel dredging, and/or other anthropogenic activities (O’Driscoll et al., 2010; Simon and Rinaldi, 2006; Walsh et al., 2005). Therefore, water column depth appears to be a significant factor in regulating the resistance to precipitation in estuaries.

Lastly, because temperature controls various chemical and biological processes that impact DO availability (e.g., microbial growth, oxygen solubility), many studies have focused on the impact of rising temperature on DO dynamics in estuaries (Apple et al., 2006; Caffrey, 2003; Caffrey et al., 2014). Highlighted results link changes in weather patterns to thermal pollution of aquatic systems following rain events (Zahn et al., 2021) and link elevated global temperatures to decreased primary production (Song et al., 2018). As such, estuaries with elevated ambient

temperatures may have a decreased capacity to resist disturbances to DO dynamics relative to estuaries with more moderate temperatures.

4.4 High variability in factors associated with resistance at local scales.

The substantial differences in physicochemical factors linked to resistance across individual estuaries highlight the need to consider both generalized and local estuarine patterns when predicting and managing the effects of major precipitation events and watershed urbanization on water quality (Fig. 5). Individual estuaries may require consideration of factors beyond water temperature, water column depth, turbidity, and Chl-*a*—identified here as generalizable at the continental scale— and be mindful of differences in the directionality of relationships between physicochemical factors and resistance among estuaries. For example, we found that resistance at CBM is related to one factor (water column depth), while at GTM, resistance is related to five factors (salinity, water temperature, Chl-*a*, PO_4^{3-} , and N:P), from which three factors were not identified at the continental scale. Similarly, although resistance and salinity are positively related at GTM and SFB, at WKB this relationship is negative. Likewise, while the relationship between resistance and water column depth is negative at SFB, at CBM and LKS this relationship is positive.

Variabilities in responses to stressors, at both the system and individual factor levels, is common. Understanding how individual physicochemical factors respond to precipitation broadly and within each system provides insight into the observed relationships between those factors and resistance. For example, Chl-*a* and dissolved inorganic phosphorus have been shown to respond more strongly to storms in some estuaries than in others, due to differences in light availability, grazing, and nutrient concentrations (Chen et al., 2015; Cloern, 2001; Cloern and Jassby, 2010; Dix et al., 2013, 2008; Liao et al., 2021; Zhang et al., 2022). Likewise, water temperature and salinity also have variable responses to precipitation (Buelo et al., 2023; Chen et al., 2015; Dix et al., 2008), leading to differences in biological processes and phytoplankton activity across estuaries (Apple et al., 2008). System dependencies on groundwater discharge—as a driver of nutrient inputs and a regulator of DO dynamics (Brookfield et al., 2021; Kornelsen and Coulibaly, 2014)—or on microbial community structure (Cheng et al., 2021) should also be appraised when evaluating factors associated with urban estuaries' resistance to precipitation.

4.5. Conceptual model for resistance to precipitation based on estuary characteristics.

Through observed relationships between resistance and physicochemical and land-use factors, we propose the following conceptual model for expected resistance based on estuarine characteristics (Fig. 6, Table S1). We suggest that the highest resistance could be expected in tidal/marine-dominated or urban-influenced estuaries that are well-mixed and have relatively short residence time, similar to SFB. In such estuaries, the disturbances to DO dynamics driven by precipitation events, riverine freshwater inputs and/or terrestrial-aquatic exchange across mixed land uses (e.g., changes in salinity, turbidity, and/or nutrients), are mediated by tidal mixing. In contrast, the lowest resistance index could be expected in enclosed, shallow, poorly-mixed (i.e., stratified) estuaries with long residence time, strong influence from river-systems, and/or surrounded by agricultural LULC, similar to WKB, Pellicer Creek (GTM), or Pokegama Bay (LKS). In such estuaries, the abrupt disruptions to ambient salinity, turbidity, and/or nutrient concentrations driven by precipitation events would be long-lasting and cause large fluctuations in DO, which in turn would result in low resistance.

Because estuaries are characterized by a continuum of interacting variables, we expect that many estuaries will show moderate resistance that varies with precipitation events of different sizes. In particular, we expect medium-high resistance indexes in well-mixed estuarine systems with engineered flow, short residence time, and periodic nutrient loading conditions (e.g., CBM, remaining LKS and GTM sites), while medium-low resistance could be more common in partly-mixed estuaries, with periodic nutrient loading and moderate residence time.

High Resistance • Tidal-dominated • Well-mixed • High salinity • Short residence time • Varying depth • Varying urbanization • Moderate nutrient loading	Medium-high Resistance • Engineered flow • Well-mixed • Low salinity • Short residence time • Deep • Urbanized • Periodic nutrient loading
Medium-low Resistance • River-dominated • Partly-mixed • Low salinity • Moderate residence time • Deep • Varying urbanization • Periodic nutrient loading	Low Resistance • Enclosed / Lagoon • Poorly-mixed • Abrupt change in salinity • Long residence time • Shallow • Agricultural • High nutrient loading

Fig. 6. Conceptual model for expected estuarine resistance to precipitation events based on estuarine characteristics.

Although numerous estuary-specific interactions can produce different responses to stressors, our results provide a framework for understanding generalizable responses to precipitation based on measurable characteristics, while also highlighting the importance of system variability in determining estuarine resistance. We recognize that additional scales of investigation (e.g., regional) and other ecological, hydrological and environmental factors could further refine predictions of estuarine resistance. Incorporating these elements in future studies could strengthen our ability to anticipate estuarine responses to disturbances.

5. Conclusions.

In light of increasing urbanization and changing rainfall patterns, cross-scale evaluations of estuarine responses to precipitation events are imperative for developing effective management strategies. We find that urban estuaries appear to be more resistant to changes in DO in response

to precipitation than more pristine estuaries; and that the depth of the water column, water temperature, turbidity, and Chl-*a* are generalizable/cross-scale predictors for estuarine resistance. However, across different scales, we find that system variability results in additional factors that are important to consider when managing the responses to major precipitation events of individual estuaries. Based on our results, we suggest that investigations targeting N, nutrient delivery mechanisms, and microbial activity could help understand what makes urban estuaries more resistant to large shifts in DO following precipitation. We also highlight that high-resolution water quality and nutrient data surrounding precipitation events, along with careful considerations of local variabilities and models for system responses to precipitation, are needed to help elucidate the underlying mechanisms for high resistance of urban estuaries. This study serves as a platform for improvement of guidelines and predictive capabilities addressing system response to future weather and urbanization scenarios.

Data availability: This study used publicly available datasets, which included: 1) Long-term estuarine water quality, nutrients and meteorological conditions and watershed boundaries for Lake Superior (LKS) NERR, Chesapeake Bay, Maryland (CBM) NERR, Guana Tolomato Matanzas (GTM) NERR, Weeks Bay (WKB) NERR, and San Francisco Bay (SFB) NERR stations from <https://cdmo.baruch.sc.edu>; 2) Long-term precipitation data from U.S. airports from <https://www.ncei.noaa.gov/cdo-web/datasets>; 3) Land use/land cover maps from <https://livingatlas.arcgis.com/landcover/>; 4) U.S. population data from <https://www.worldpop.org/>. Data and processing code are available at <https://figshare.com/s/49d2f3dca084d885638a>.

Author Contribution: E.B.G. and A.B.T. developed the study and interpreted the results. A.B.T. performed data analysis and drafted the manuscript. All authors contributed to manuscript editing.

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