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Identifying wastewater pollution hotspots in coastal Hawaiian waters with high freshwater inputs

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ABSTRACT

Identification of wastewater pollution hotspots and sources is a management priority because wastewater threatens human and ecosystem health in coastal waters. We assessed wastewater pollution from onsite sewage disposal systems (OSDS) and an offshore wastewater treatment plant (WWTP) outfall in the coastal waters of Hilo, Hawai'i, an area with high freshwater inputs from groundwater and rivers. Twenty shoreline stations were sampled monthly from July 2020 to October 2021 for fecal indicator bacteria, nutrients, and stable nitrogen isotopes of macroalgae and nitrate. Wastewater hotspots were identified using a wastewater pollution score and stable isotope mixing models. Wastewater was detected at all stations, although most wastewater indicator concentrations were low. High groundwater inputs likely diluted wastewater indicators making their detection challenging. Stable isotope mixing models support this conclusion as groundwater was the dominant nitrate source at all stations (44–87%), while wastewater (OSDS + WWTP) comprised 8–49%. Addition of OSDS density to the wastewater pollution score improved identification of hotspots against the high freshwater input background. This finding is supported from concurrent dye tracer tests that confirmed the presence of wastewater and rapid transport from OSDS to shoreline springs at hotspot locations. Our results illustrate the importance of using a variety of approaches to understand the wastewater pollution threat, especially in areas with high freshwater inputs.

1. Introduction

Wastewater pollution is pervasive in coastal waters globally. Wastewater includes septage leached into groundwater from onsite sewage disposal systems (OSDS: cesspools, septic tanks, aerobic treatment units (ATU)), untreated sewage entering into a wastewater treatment plant (WWTP), and treated effluent released from WWTPs' outfalls and injection wells (Wear et al., 2024). Wastewater from all of these sources reaches the ocean through groundwater, rivers, and direct release; thus, knowing which transport pathway helps guide management actions. In rural areas, OSDS are used to dispose of domestic wastewater because populations are geographically dispersed, topography and climate can be extreme, and communities lack financial resources to build and maintain wastewater collection and treatment infrastructure (USEPA, 2016). Because OSDS do not treat wastewater, their septage contains pathogens, which are rarer in treated wastewater. In urban areas, domestic and industrial wastewater are transported via sewer systems to WWTP, where wastewater is treated to different degrees depending on the plant, and discharged via outfalls and injection wells.

Regardless of source, wastewater in nearshore waters is problematic because it has high concentrations of nutrients, pathogens, endocrine disruptors, heavy metals, and other toxins (Wear and Vega Thurber, 2015). Hence, various approaches and indicators are used to detect its presence in coastal waters. Dye tracer tests document hydraulic connectivity between OSDS and WWTP to coastal waters, as well as detect

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failing sewer infrastructure (HDOH, 1984; Glenn et al., 2013; Abaya et al., 2018; Wiegner et al., 2021; Waiki et al., 2025). However, measurements of wastewater indicators are more common and include the use of fecal indicator bacteria (FIB, *Enterococcus* spp., *Clostridium perfringens*), microbial source tracking (MST, i.e., human-associated *Bacteroides* markers), nitrogen (N) stable isotopes in macroalgae and NO_3^-, nutrient concentrations, optical brighteners, artificial sweeteners, and certain contaminants of emerging concern including pharmaceuticals. Using stable isotopes of NO_3^- in mixing models can provide further information on relative contribution of different pollution sources, which is invaluable for management decisions. A challenge to using these indicators is that individually, they can provide conflicting information on the intensity and location of wastewater pollution. However, when these indicators are used together in a scoring tool, they can provide more accurate information on pollution hotspots (Cronin et al., 2006; Richards et al., 2017; Abaya et al., 2018; Waiki et al., 2025).

The State of Hawaiʻi is challenged with wastewater pollution from both OSDS and WWTP. The former is primarily cesspools, which are unlined, underground holes; septic tanks comprise a smaller percentage, and have holding tanks and leach fields. Hawaiʻi State has ~110,000 OSDS, with 48,596 cesspools on Hawaiʻi Island (Whittier and El-Kadi, 2014; HDOH, 2017; Mezzacapo and Schuler, 2021). The high permeability of basalt on the Hawaiian Islands allows for untreated cesspool septage to rapidly seep into groundwater and be transported to the shoreline posing immediate human health risks (HDOH, 1984; Abaya et al., 2018; Wiegner et al., 2021; Waiki et al., 2025; Kamau, 2025). In 2017, the Hawaiʻi State Legislature passed Act 125, which requires the replacement of all cesspools by 2050 (HDOH, 2015; HDOH, 2017). With the large number of cesspools, more data are needed to assist the State and counties with prioritizing locations for cesspool conversion. Additionally, Hawaiʻi State received a D+ grade on its sewer infrastructure from the American Society of Civil Engineers (ASCE) because of its deterioration and substantial need for funding, especially in light of climate change impacts associated with sea level rise and increased number and severity of storms (ASCE, 2025). New and expanded sewer infrastructure is needed as cesspools are phased out (ASCE, 2025).

Hilo is one location in Hawaiʻi State potentially polluted by wastewater from both OSDS and a WWTP (outfall and sewer infrastructure). Wastewater from OSDS leaches into the groundwater and is discharged into Hilo's rivers and at nearshore springs within hours to days (Waiki et al., 2025). Along Hilo's southeast shoreline, the substratum is pāhoehoe basalt and the soils are shallow Folists histosols, both of which drain freely (USDA NRCS, 2009). Therefore, when septage leaches from OSDS, it percolates fast through the porous and permeable basalt reaching the water table, with little time for biological processes to remove septage contaminants (Waiki et al., 2025). Further inland, OSDS-contaminated groundwater discharges into rivers and streams, where it is biologically modified, especially during low discharge conditions, which is the majority of the year. However, during storms, when there is surface runoff, other pollution sources can contribute to the pollution load making it harder to detect wastewater (Fujioka, 2001; Li et al., 2019; Lin et al., 2019). In contrast to OSDS, wastewater from the WWTP's outfall is released directly into the ocean. Hilo is an ideal place to examine wastewater pollution because many coastal waters throughout Hawaiʻi State, the Pacific Island region, and elsewhere in the world are impacted by wastewater from OSDS, WWTP, or both (Curiel-Ayala et al., 2012; Kirs et al., 2017; Shuler et al., 2019; Knapp et al., 2020; Tonga et al., 2025). However, these wastewater sources are generally studied individually, even in areas that are potentially impacted by both (i.e., Smith et al., 1981; Laws and Allen, 1996; Kirs et al., 2017; Aguiar et al., 2023), and only one study to our knowledge has separated out pollution contributions from each one (Waiki et al., 2025). Hilo can also serve as a model system providing insights into the importance of different freshwater sources (groundwater vs. rivers) in transporting wastewater to coastal areas. This is especially true in areas with high annual rainfall, where it may be difficult to detect wastewater because of significant dilution from high freshwater inputs (Mezzacapo et al., 2020; Waiki et al., 2025). Previous Hilo studies examined wastewater pollution under various river flow conditions focusing on FIB, but they did not employ the use of stable N isotopes and did not include shoreline sampling, where people recreate, subsistence fish and gather, as well as perform cultural practices (Wiegner et al., 2013; Wiegner et al., 2017; Economy et al., 2019). There are only three studies of groundwater-influenced areas without rivers in Hilo, and only one study examined wastewater pollution in a small geographical area (Carlson and Wiegner, 2016; Steadmon et al., 2024; Waiki et al., 2025).

Objectives of this study were to: 1) evaluate the presence of wastewater in nearshore waters through measurements of wastewater indicators, 2) examine the relative importance of groundwater and rivers in transporting wastewater pollution, and 3) assess the efficacy of wastewater indicators to detect wastewater in a tropical setting with high freshwater inputs. Our hypotheses were: 1) areas with high groundwater inputs will have higher concentrations of wastewater indicators and wastewater pollution scores, 2) wastewater indicators will be positively correlated with OSDS density, and 3) wastewater from OSDS will comprise a larger percent of the wastewater pollution in Hilo as quantified through stable isotope measurements of NO_3^-. This study provides the first comprehensive water quality survey along the entire Hilo shoreline. These data are essential for prioritizing locations for cesspool conversion. This information will also serve as a baseline for future comparisons with cesspool removal, WWTP upgrades, and inundation of OSDS and sewer infrastructure with sea level rise (McKenzie et al., 2021; Kiefer and Felton, 2024; Kamau, 2025). Additionally, there are over 9000 high islands with coral reefs like Hawaiʻi Island throughout the Pacific, and our study provides a framework for natural resource managers in this region tasked with addressing wastewater pollution in their coastal waters (Nunn et al., 2016).

2. Methods

2.1. Site description

This study was conducted in the Hilo area, located on the east side of Hawaiʻi Island, HI, USA. Hilo has a tropical climate (average annual rainfall ~7000 mm at high elevations and 3000 mm near sea level), with generally wetter winters (November–April) and drier summers (May–October), although storms can occur year-round (Giambelluca et al., 2013). The Hilo Bay watershed drains the volcanoes Mauna Kea and Mauna Loa and their lava flows create two distinct geological zones. To the north, are the Mauna Kea flows (Hāmākua and Laupāhoehoe) and tephra deposits (~7500–10,000 y and > 11,000 y). To the south, are the Mauna Loa flows (Kaʻu basalts) and small tephra deposits, which are highly porous and permeable (<7500 y; USDA NRDC, 2009). The Hilo Bay watershed has two major rivers: Wailoa and Wailuku, and multiple streams, including Honoliʻi Stream (Fig. 1). The Wailoa River watershed encompasses 255.4 km^2 and has a maximum elevation of 2964 m, with land use consisting of 70.9% conservation, followed by 20.4% agriculture, and 8.7% urban development near the lower reaches (Parham et al., 2008). The Wailoa River is not gauged, but earlier measurements recorded baseflow discharge as low as 52 m^3/d and peak stormflow at 691,200 m^3/d (Wiegner and Mead, 2009). The Wailuku River watershed is the largest in Hawaiʻi State, with an area of 653.2 km^2 and a maximum elevation of 4200 m (Parham et al., 2008). The dominant land use is conservation, followed by agriculture, and urban development at 76.9%, 22.4%, and 0.7%, respectively (Parham et al., 2008). During our study, the lowest baseflow discharge recorded at the Wailuku River was 22,263 m^3/d and the peak stormflow was 2,471,041 m^3/d (USGS station 16,704,000, https://waterdata.usgs.gov/monitoring-location/16704000/). Honoliʻi Stream's watershed has an area of 39.2 km^2 and a maximum elevation of 1867 m, consisting predominantly of conservation land, followed by agriculture, and urban development at 78.4%,

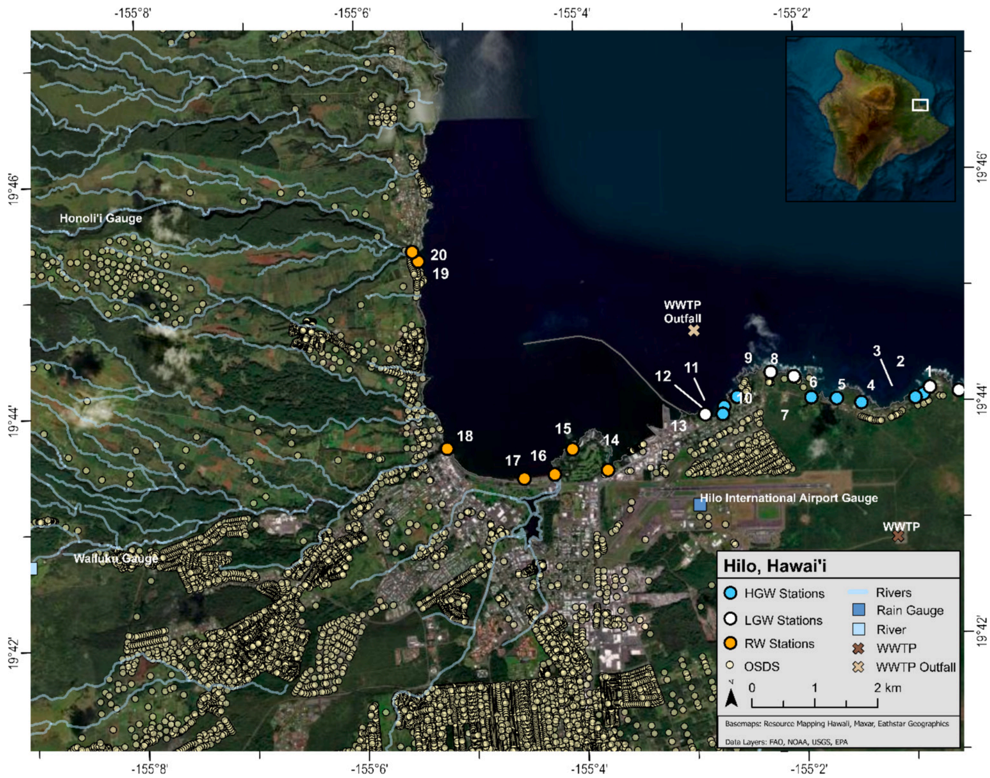

Fig. 1. Map of 20 shoreline stations (white circles with station numbers) sampled monthly for wastewater pollution indicators from July 2020 to October 2021 in Hilo, Hawaiʻi, Map shows the Hilo Wastewater Treatment Plant (WWTP) and its outfall, the Hilo International Airport National Weather Service (NWS) station (#91285021504), rivers (south to north: Wailoa (St. 16), Wailuku(St. 18), Honoliʻi (St. 20)), United States Geological Survey (USGS) river gauges (Wailuku River gauge (#16704000), Honoliʻi Stream gauge (#16717000). Onsite sewage disposal systems are shown as colored circles by freshwater influence groups; light blue: high groundwater (HGW), white: low groundwater (LGW), and yellow river water (RW) stations. The inset is of Hawaiʻi Island, with the study location outlined. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

21.0%, and 0.6%, respectively (Parham et al., 2008). During our study, the lowest baseflow discharge measured at Honoliʻi Stream was 21,945 m^3/d and the peak stormflow was 347,413 m^3/d (USGS station 16,717,000, https://waterdata.usgs.gov/monitoring-location/16717000/). In addition to freshwater inputs from rivers, a significant amount of freshwater enters the bay as submarine groundwater discharge (SGD). Estimated total SGD fluxes for this area is 4,664,563 m^3/d (Hilo + Onomea aquifers; Engott, 2011). The Hilo Bay shoreline consists of rugged rocky shores, along with black sand beaches interspersed with a few white sand pockets, numerous freshwater springs, and a wave-exposed fringing coral reef with spur and groove formations.

Hilo town is Hawaiʻi Island's population center and home to ~43,263 residents (US Census Bureau, 2021). The town fronts Hilo Bay, which has a 3.07-km breakwater, which shelters the Port of Hilo, but also reduces ocean circulation and traps sediment and pollutants from the rivers (M&E Pacific, 1980). Hilo has 10,374 cesspools, with a density of 15.8 OSDS/km^2, discharging 24,000 m^3/d of untreated wastewater into groundwater and rivers (Whittier and El-Kadi, 2014; Mezzacapo and Schuler, 2021). A fixed-film secondary-treated, chlorinated effluent is produced at the Hilo WWTP and is discharged at a rate of 13,638 m^3/d into Hilo Bay through an outfall pipe located in Puhi Bay ~1.3 km offshore (Fig. 1; USEPA, 2010). The WWTP services ~30% of Hilo's homes. The WWTP has been fined by the United States Environmental Protection Agency (USEPA) repeatedly since 2016 for the derelict conditions of the plant and illegal discharges into the ocean (Dobbyn, 2024). The plant no longer operates within the USEPA standards under the Clean Water Act (Brestovansky, 2024). Effective March 2024, the USEPA placed Hawaiʻi County under an administrative consent order to rehabilitate and replace corroded equipment, address deferred maintenance, and develop a program to systematically repair, rehabilitate, and replace aging infrastructure by 2035 (USEPA, 2024). The consent order also includes that Hawaiʻi County must expand sewer service and create a cesspool conversion plan by 2026 (Dobbyn, 2024).

2.2. *Station characterization and shoreline sampling*

Twenty stations were sampled monthly from July 2020 to October 2021 on two consecutive days along 15 km of the Hilo shoreline ($n_{per\ station}$ = 16). Stations were selected based on observations of groundwater discharge from field salinity surveys, county beach park status, and community significance. A priori emphasis was placed on salinity for station selection and for later freshwater-influence (FW) groupings. Stations were sampled on the lowest possible tides of each month within ±2 h of sunrise. Sampling at these times minimizes dilution of groundwater with seawater and reduces sunlight inactivation of FIB (Fujioka et al., 1981). Our experimental design (monthly sampling at each station over 16 months) captured the variability under a range of hydro/hydrogeological conditions, while controlling for the tides and the time of day. Season did not affect the concentration of the most influential wastewater indicators (FIB and δ^{15}N-macroalgae; Nakoa, 2022). To examine the relative importance of groundwater and rivers in transporting wastewater to Hilo's coastal waters, stations were grouped according to the freshwater source that most influenced their water quality. Salinity data and geographic attributes of the station, like proximity to river mouths, were used. Stations east of the breakwater (St. 1–13) all had visible springs, but varying mean salinity. A natural break at a mean salinity of about 9 (Jenks Natural Breaks Optimization) was used to group these stations. Stations with an average salinity <9 that had a visible spring were considered to have a high groundwater-influence (HGW). Stations with an average salinity >9 throughout the study, despite having a visible spring, were considered to have a low groundwater-influence (LGW). River water-influenced (RW) stations were located at river mouths or inside the breakwater. Water inside the breakwater typically has a residence time of several days, allowing time for river water to spread throughout the bay (M&E Pacific, 1980). Eight stations were HGW, five LGW, and seven RW (Fig. 1). Stations were also examined individually for water quality.

Triplicate water samples were collected from each station in sterilized, acid-washed 1-L polypropylene bottles. Salinity and temperature were measured in the field using a calibrated YSI ProSolo. Macroalgal species present at each station were haphazardly collected based on availability and proximity to the water sample collection point. Water and macroalgae samples were stored on ice until processed in the laboratory, ~2 h after sampling started.

2.3. Water quality analyses

Enterococcus spp. were quantified using the Enterolert MPN method and samples below the detection limit (DL) are reported as 5 MPN/100 mL, half of the DL (IDEXX Laboratories Inc.). *Clostridium perfringens* was quantified by filtering 100 mL of sample through a 0.45-μm pore size cellulose nitrate filter (Whatman™) and plating the filter onto membrane-*C. perfringens* (m-CP) agar base (Oxoid Ltd., Basingstoke, Hampshire, England), incubated in an anaerobic jar at 45 °C for 18–24 h, followed by fumigation with ammonium hydroxide; pink colony forming units (CFU) were counted as positive detection (Bisson and Cabelli, 1979). All FIB samples were processed within 6 h of sample collection.

In March and April 2022 following completion of sampling for this study, triplicate samples were collected from each station for human-associated *Bacteroides* marker HF183. For human-associated *Bacteroides* marker HF183 analysis, 250–500 mL of water were filtered through a 0.22-μm mixed cellulose ester membrane filter (Millipore, USA) and stored frozen in Qiagen bead beating tubes (Cat. No.: 12888–100-PBT) until analysis. For DNA extraction, samples were thawed at room temperature 1–2 h. DNA extraction proceeded via the MP Bio FastDNA spin kit for soil following the manufacturer's instructions, with a modification: cells were mechanically lysed using a MP Biomedicals FastPrep-96 bead beater for 45 s at maximum speed, 6.5 m/s. DNA was stored at −40 °C until sequencing. Detection of *Bacteroides* was performed using the established qPCR assay HF183 in triplicate (Table S1). 25-μL reaction mixtures containing 10 μL of Kapa Probe Force Master Mix (Cat. No. KK4301), 400 nmol/L of each specific primer, 200 nmol/L of each specific probe, and 1 μL of sample or standard DNA were amplified using an Eppendorf Realplex2 Mastercycler®. A 2-min hot start at 95 °C was followed by 45 cycles of 95 °C denaturing for 30 s, and 60 °C elongation for 30 s. Standard curves were constructed from serial dilutions of synthetic DNA fragments. For each assay, eight dilution steps ranging from 50,000–10 gene copies per well were performed in triplicate on every run plate. The DL was10 copies per well with a mean cycle threshold of 43. R^2 values ranged from 0.96 − 0.99 and PCR efficiency ranged from 88 to 96%.

Samples for nutrient analyses were taken from one of the triplicate bottles from each station. Water was filtered through a muffled (600 °C, 5 h) GF/F filter (Whatman, 0.7-μm pore size) and stored frozen until analysis. Nutrient samples were analyzed with a Lachat Quikchem 8500, flow-injection autoanalyzer for $NO_2^- + NO_3^-$ [DL 0.07 μmol/L, USEPA 353.2], NH_4^+ [DL 0.36 μmol/L, USEPA 349], Total Dissolved Phosphorus [TDP, DL 0.25 μmol/L, USGS I-4650-03, USEPA 365.5], PO_4^{3-} [DL 0.03 μmol/L, USEPA 365.5], and H_4SiO_4 [DL 1 μmol/L, USEPA 366]. Total Dissolved Nitrogen (TDN) was analyzed by high-temperature combustion, followed by chemiluminescent detection of nitric oxide using a Shimadzu TNM-1 (DL 5 μmol/L, ASTM D5176). Concentrations for samples that were below the DL are reported as half of the DL. Filters from nutrient sample processing were stored in amber plastic vials at −80 °C until analysis for Chlorophyll *a* (Chl *a*). Filters were analyzed using a Turner 10-AU fluorometer, with a DL of 0.05 μg/L (USEPA, 445.0). Turbidity was measured on a calibrated HACH 2100Q turbidimeter (Version 1.03/9 #14100C035910, DL 0.01 NTU). All nutrient analyses were conducted by the University of Hawaiʻi at Hilo (UH Hilo) Analytical Lab.

2.4. Stable isotopes

Composite intertidal macroalgal samples from each station were rinsed with deionized water to remove debris. Vouchers of each species present in the samples were preserved in 4% formalin-seawater solution for identification. Remaining algal tissue was dried at 60 °C until a constant weight was achieved, then ground and homogenized using a Wig-L-Bug® grinding mill, and folded into 3.5 × 5 mm foil tins for stable isotope analysis. Samples were analyzed for $\delta\ ^{15}N$ using a Thermo Delta V isotope ratio mass spectrometer (IRMS). Nitrogen isotope data were normalized to United States Geologic Survey (USGS) standards 40 ($\delta\ ^{15}N$ vs Air = −4.5‰) and 41 ($\delta\ ^{15}N$ vs Air = +47.6‰), accurate to 0.2‰, expressed as per mil (‰), $\delta\ ^{15}N = [(R_{sample}\text{-}R_{standard})/R_{standard}] \times 1000$, $R = {}^{15}N/{}^{14}N$, and reported as %N and $\delta\ ^{15}N$ vs N_2 in air. Macroalgal stable isotope samples were analyzed at the UH Hilo Analytical Lab. Composite samples were used because the same species of macroalgae were not present at every station, and species varied among stations. Previous studies have successfully used composite samples to map out wastewater pollution hotspots, including on Hawaiʻi Island (Derse et al., 2007; Dailer et al., 2012; Wiegner et al., 2016; Abaya et al., 2018). Additionally, previous studies found that $\delta^{15}N$ from different macroalgal species collected at the same location have similar $\delta^{15}N$ values (Derse et al., 2007; Wiegner et al., 2016). Preserved samples were identified using a Leica ZOOM 2000 dissecting microscope, an Olympus BX41 compound microscope, and identification books (Abbott, 1999; Abbott and Huisman, 2004; Huisman et al., 2007).

Water samples for $\delta^{15}N$- and $\delta^{18}O$-NO_3^- analysis were filtered through a 0.45-μm nylon filter (Corning) and analyzed on a Thermo-GasBench II (Thermo Electron) interfaced to an IRMS (McIlvin and Altabet, 2005). Isotope data were normalized to USGS standards (USGS32, USGS34, USGS53) and expressed as per mil (‰), $\delta\ ^{15}N$, $\delta\ ^{18}O = [(R_{sample}\text{-}R_{standard})/R_{standard}] \times 1000$, $R = {}^{15}N/{}^{14}N$, ${}^{18}O/{}^{16}O$. Analyses were conducted at the UH Hilo Analytical Lab.

The percent contribution of specific N sources to the NO_3^- pool for the FW-influence groups and at each station were determined using the $\delta^{15}N$ and $\delta^{18}O$ of NO_3^- in water in a stable isotope mixing model in the program R, the "*simmr*" package (version 0.4.5). Potential NO_3^- sources sampled were treated effluent from the Hilo WWTP pre-outfall manhole (WWTP, $n = 15$) and outfall discharge plume ($n = 12$), inland groundwater from drinking water wells (GW, $n = 7$), and ocean water ($n = 3$) (Table S2). NO_3^- concentrations for the outfall discharge plume and ocean water were below the minimum NO_3^- concentration needed for stable isotope analysis (2 μmol/L); therefore, they were not used as NO_3^- sources (Table S2). Stable isotope data for OSDS on Hawaiʻi Island were obtained from Wiegner et al. (2016) and Abaya et al. (2018) consisting of cesspools, septic systems, and ATU (OSDS, n = 15). Stable isotope values for the Wailuku River were obtained from Colo (2025) for sampling dates that overlapped with our study (RIV, $n = 11$). This model structure examined the contributions of each potential source to the NO_3^- pool for each FW-influence group by averaging the stable isotope values for the stations within each FW-influence group. *Simmr* uses Bayesian methods and NO_3^- concentration dependence to account for natural variation and uncertainty in the model in order to estimate proportional contributions of each source to the NO_3^- concentration in each sample (Fig. S1; Parnell et al., 2010; Parnell et al., 2013). Normality of $\delta^{15}N$ and $\delta^{18}O$ values was tested using the Shapiro-Wilk normality test (*shapiro.test*). Bi-plots were created using average values of $\delta^{15}N$ and $\delta^{18}O$ in NO_3^- for each potential source to remove any sources with substantial overlap from the mixing model (Fig. S2).

Separate models were run for stations outside and inside of the breakwater. Although the breakwater is permeable, it is unlikely that the

WWTP outfall effluent would be a NO_3^- source to areas inside the breakwater because of persistent freshwater flow out of Hilo harbor and the water currents outside of the breakwater move away from it towards the southeast (M&E Pacific, 1980; Puniwai et al., 2016). The model for stations outside of the breakwater analyzed contributions of GW, OSDS, and WWTP to the NO_3^- pool at HGW and LGW stations. The model for stations inside the breakwater analyzed contributions of GW, OSDS, and RIV to the NO_3^- pool at RW stations. Models were also run to assess contributions of NO_3^-sources to each station. For the model that examined contributions of each potential source to the NO_3^- pool at each individual station, stable isotope values for each station were averaged. Note, because the models averaged the stable isotopes differently (by FW-influence groups vs. by station), it is possible that source contributions for a station might be out of the range for the FW-influence group that it is a part of. Mixing model results are presented as Bayesian 50% credibility intervals to account for a larger range of variability.

2.5. Wastewater pollution scores

Median wastewater pollution scores were calculated for FW-influence groups and also, separately for each station using a modified score from Abaya et al., 2018 (Waiki et al., 2025). The score uses FIB (*Enterococcus* spp., *C. perfringens*), nutrient concentrations ($NO_3^- + NO_2^-$, NH_4^+, TDP), and δ^{15}N-macroalgal data. The wastewater indicators selected for this scoring system have either HDOH water quality standards or well-established wastewater pollution values in the literature (i.e., δ^{15}N-macroalgal). The original score was developed for a coastal community in West Hawai'i that has a dry climate and no perennial streams using HDOH open coastal water quality standards (Abaya et al., 2018; HDOH, 2021). The score used in this study and Waiki et al. (2025) was modified to better represent the high freshwater inputs and wet climate in Hilo using HDOH wet conditions standards for estuaries and embayments (HDOH, 2021). Levels for nutrient concentrations were the only part of the original score modified (Waiki et al., 2025), as HDOH FIB water quality standards for marine recreational waters are not specific to climate and weather conditions, and there are no standards for δ^{15}N-macroalgal values (HDOH, 2021). Each indicator was assigned a level based on its measured value in our study and where it falls relative to water quality standards or literature values, with low values indicative of lower pollution levels (1 = low, 2 = medium, 3 = high; level values found in Waiki et al., 2025). Once each indicator was assigned a level (levels 1–3), it was multiplied by a weight factor (weights 1–3 according to Abaya et al., 2018), with the most reliable wastewater indicators from the literature having the greatest weight. *Clostridium perfringens* and δ^{15}N in macroalgae had the greatest weights (weight = 3), as these indicators are more specific to wastewater pollution, more integrative of environmental conditions, and do not fluctuate to the same degree as *Enterococcus* spp. and nutrient concentrations. Justification for weights for the other parameters are provided in Abaya et al. (2018). The equation for calculating the wastewater score was: (*C. perfringens* level x 3) + (δ^{15}N-macroalgal level x 3) + (*Enterococcus* spp. level x 2) + ($NO_3^- + NO_2^-$ level x 1) + (NH_4^+ level x 1) + (TDP level x 1). Wastewater pollution score categories were: low (11–17), medium (18–25), and high (26–33). Note, the effectiveness of this scoring system to detect wastewater pollution hotspots was confirmed by dye tracer tests within our study site (Waiki et al., 2025) and at another location on Hawai'i Island (Abaya et al., 2018).

2.6. Geospatial and statistical analyses

To determine the OSDS density, a 0.8-km radius buffer was created around each station in ArcGIS Pro (2.8.0) for a total buffer area of 2 km^2. This size area was chosen to ensure that stations' buffers did not overlap and to quantify the number of OSDS closest to the stations that would likely have the greatest impact on water quality. Within this buffer, the ocean area was clipped, leaving ~1-km^2 inland buffer around each station. The total number of OSDS within the stations' buffers were isolated using the intersect tool on the "osds_haw" layer (2010) available from HDOH (2017). The spatial join tool was used to combine the land area buffer and OSDS layers to calculate OSDS density within the ~1-km^2 land area buffer for each station.

Statistical analyses examined differences in water quality among FW-influence groups. Because assumptions of parametric statistical tests were not met on raw or transformed data, nonparametric tests were used. Hence, data in this study are presented as medians and ranges. Kruskal-Wallis tests (*kruskal.test*) were used to determine if the *Enterococcus* spp., *C. perfringens*, human-associated *Bacteroides* marker HF183, nutrient concentrations, %N-macroalgae, δ^{15}N-macroalgae, and wastewater pollution scores differed among FW-influence groups. Dunn's tests with a Bonferroni correction were used to determine significant differences among FW-influence groups. Spearman's rank correlation tests (*cor.test*) examined if there were any associations between OSDS density, measured wastewater indicators, and other water quality parameters. All statistical tests were completed using the program R (version 4.0.3) with $\alpha = 0.05$ or 0.017 when a Bonferroni correction was applied.

3. Results

The three FIB varied among FW-influence groups (Fig. 2; Table 1). *Enterococcus* spp. concentrations significantly differed among groups, with the median RW group concentration up to two times higher than those in the LGW and HGW groups (Fig. 2; Table 1). Note, however, the highest single *Enterococcus* spp. concentration was measured at a HGW station (St. 3, 6546 MPN/100 mL, Table S3). There was a 100% positive hit rate for all *Enterococcus* spp. samples across all three FW-influence groups. Overall, *C. perfringens* concentrations were low in all three FW-influence groups, but significantly differed between the RW and the two groundwater groups (Fig. 2; Table 1). However, all three FW-influence groups did occasionally have high *C. perfringens* concentrations up to 10 CFU/100 mL (Table S3). Positive hit rates for *C. perfringens* were the highest for the RW group (55%), followed by LGW (45%), and HGW ones (38%). The human-associated *Bacteroides* HF183 marker was detected in all FW-influence groups, with HGW having the most positive hits per station (7/8 stations, 50% samples), followed by LGW (3/5 stations, 25% of samples), and RW (3/7 stations, 31% of samples). Concentrations ranged from below detection up to 22 gene copies/100 mL (Fig. 2; Tables 1, S3).

Nutrient concentrations varied among FW-influence groups, with HGW stations generally having the highest concentrations (Fig. 2; Tables 1, S3). $NO_2^- + NO_3^-$ and TDN concentrations significantly differed among groups, with concentrations up to two times higher in the HGW group than the LGW and RW ones (Fig. 2; Table 1). NH_4^+ concentrations differed among groups, with the HGW group having the lowest median concentrations compared to the RW and LGW groups (Table 1). PO_4^{3-} and TDP concentrations were up to three times higher in the HGW group compared to the LGW and RW groups (Table 1).

Thirty-two identifiable macroalgal species were collected for %N and δ^{15}N analyses at 19 stations; note, no macroalgae were present at St. 17 for collection. Macroalgae from Phylum Rhodophyta were most prevalent; *Ahnfeltiopsis concinna*, *Chondrus retortus*, and *Polyopes hakalauensis* comprised nearly half of all samples. Percent N content in macroalgae differed significantly among FW-influence groups (Table 1). Percent N content was significantly higher in the HGW group compared to the LGW and RW groups (Table 1). Macroalgal δ^{15}N values also significantly differed among all FW-influence groups (Fig. 2; Table 1). The median RW and LGW groups' values were significantly higher (more +, enriched in ^{15}N) than samples collected at the HGW stations (Fig. 2; Table 1).

All FW-influence groups had a similar median wastewater pollution score of 17, falling into the low-impacted category (Fig. 3; Table 1). Wastewater pollution scores at individual stations ranged from a one-time minimum of 11 to as high as 21 at many stations (Table S4). Additionally, all stations fell within the medium-impacted category at

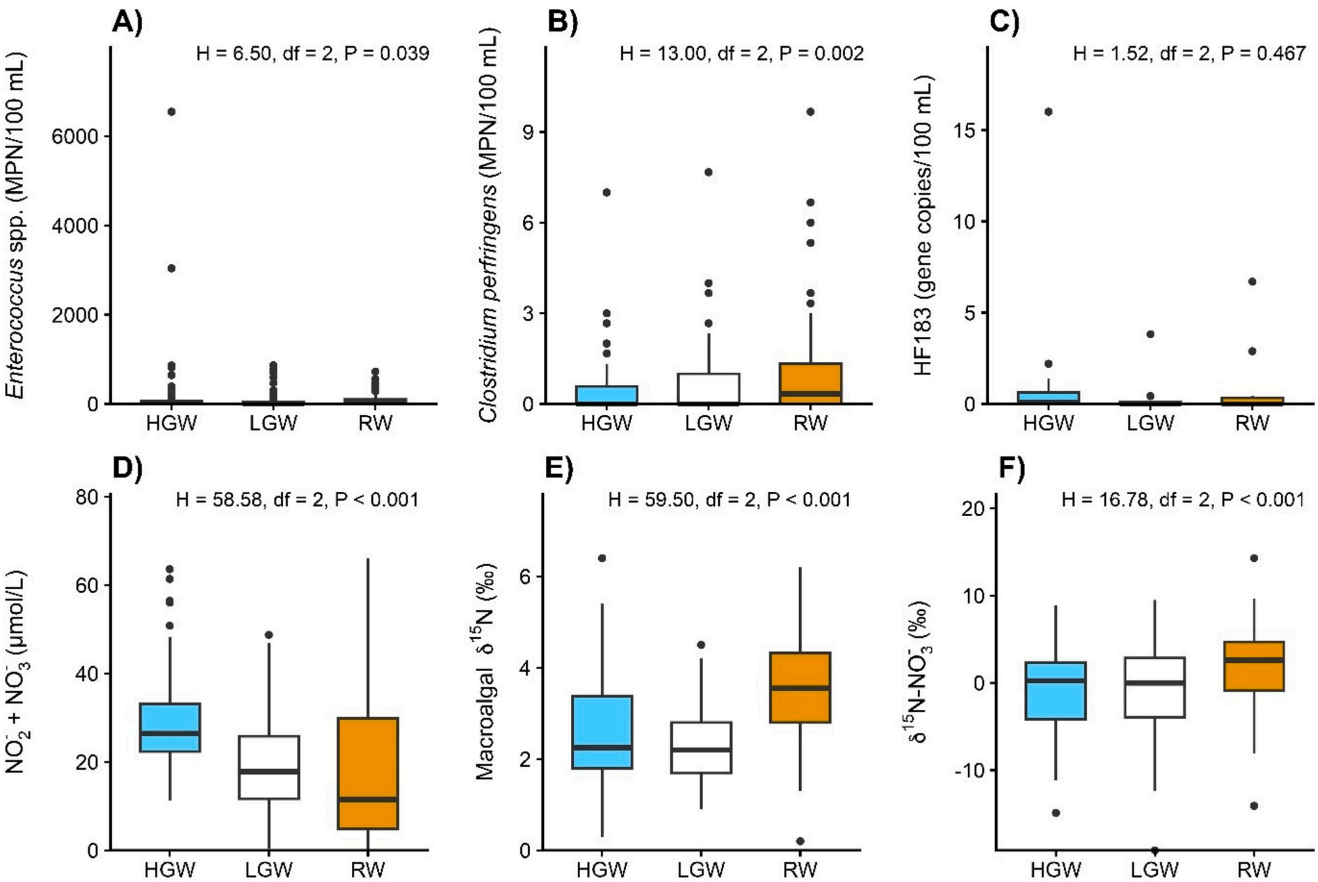

Fig. 2. Distribution of wastewater pollution indicators among high groundwater-influence (HGW), low groundwater-influence (LGW), and river water (RW) shoreline stations in Hilo, Hawai'i. Water samples were collected monthly from July 2020 to October 2021. Boxplots show the median, interquartile range (IQR), whiskers (1.5 × IQR), and outliers for (A) *Enterococcus* spp., (B) *Clostridium perfringens*, (C) human-associated *Bacteroides* marker HF183, (D) $NO_2^- + NO_3^-$ concentration, (E) δ^{15}N-macroalgal, and (F) $\delta^{15}N\text{-}NO_3^-$. Statistical results from Kruskal–Wallis tests comparing freshwater-influence groups are shown in each panel, α = 0.05.

least once. Two stations had median wastewater pollution scores that were in the medium category (St. 7 (HGW), 16 (RW)).

Based on the stable isotope mixing model, the dominant NO_3^- source to stations outside of the breakwater (St. 1–13) was GW, contributing 98% and 96% to the HGW and LGW groups, respectively (Fig. 4A). OSDS contributed the second most to the NO_3^- pool of the HGW (1.5%) and LGW (2.4%) groups (Fig. 4A). Wastewater from the WWTP contributed the least amount of NO_3^- to the HGW (0.8%) and LGW (1.4%) groups (Fig. 4A). For the RW group (St. 14–20), GW contributed the most to the NO_3^- pool (62%), followed by RIV (25%), and OSDS (10%, Fig. 4B). Source contributions were variable by station and region (outside/ inside breakwater) (Fig. S5). Overall, wastewater was a more important source to individual stations than when they were categorized into FW-influence groups (Figs. 4, S5). Outside of the breakwater, source contributions were relatively consistent across St. 1–13, with GW being the dominant NO_3^- source (68–87%), followed by OSDS (7–14%), and WWTP (5–13%, Fig. S5). Outside the breakwater, 12–27% of the NO_3^- was derived from a wastewater source (OSDS +WWTP) (Fig. S5). Of all the stations outside the breakwater, St. 9 had the greatest percentage of NO_3^- derived from wastewater (14% OSDS, 13% WWTP). Inside the breakwater, GW was generally the dominant NO_3^- source (44–59%), except at St. 18, where it contributed 22% (Fig. S5). RIV contributed the second most to the NO_3^- pools at stations inside the breakwater, ranging from 26 to 33% (Fig. S5). OSDS contributions were the smallest to the NO_3^- pool at stations inside the breakwater (8–21%), except at St. 18, where it contributed 49% (Fig. S5).

Salinity was negatively correlated with most nutrient concentrations: $NO_2^- + NO_3^-$, TDN, PO_4^{3-}, TDP, as well as %N of macroalgal tissue (Fig. 5). In contrast, salinity was positively correlated with NH_4^+ concentrations and δ^{15}N-macroalgae (Fig. 5). OSDS density was negatively correlated with *C. perfringens* and NH_4^+, and positively correlated with $NO_2^- + NO_3^-$, TDN, PO_4^{3-}, and TDP (Fig. 5). *Clostridium perfringens* was positively correlated with *Enterococcus* spp., and NH_4^+. Most nutrient concentrations were significantly correlated to one another (Fig. 5).

4. Discussion

4.1. Role of freshwater sources as wastewater vectors

Groundwater is an important vector of wastewater pollution from OSDS and WWTP's injection wells to coastal waters (Knee et al., 2008; Glenn et al., 2013; Amato et al., 2016; Abaya et al., 2018). Results from dye tracer tests showed that wastewater in Hilo is entering into the groundwater from OSDS and being transported and discharged at shoreline springs (Waiki et al., 2025). Because of this, for our study, we hypothesized that stations with high groundwater inputs, as determined by low salinity values, would have higher concentrations of wastewater indicators and higher wastewater pollution scores. We found that HGW stations generally had the highest nutrient concentrations, but not high values of other indicators like FIB, δ^{15}N macroalgae, and $\delta^{15}N\text{-}NO_3^-$ (Fig. 2; Tables 1, S3, S4). When these measurements were combined in the wastewater pollution score, we found similar scores among FW-

Table 1
Median [range] water quality concentrations, %N and δ^{15}N-macroalgal, NO_3^- stable isotopes, onsite wastewater disposal system (OSDS) density, and wastewater pollution score from 20 stations along the Hilo, Hawaiʻi, shoreline. Samples were collected monthly from July 2020 to October 2021. HGW and LGW represent stations influenced by high and low groundwater inputs, and RW represents stations influenced by river water. Results from Kruskal-Wallis test are shown in the table; letters represent groupings determined by Dunn's tests after applying a Bonferroni correction for multiple comparisons (α = 0.017).

Parameter	High groundwater (HGW)	Low groundwater (LGW)	River water (RW)
Salinity	4.47[b] [1.90–25.70]	9.14[a] [3.13–34.80]	9.42[a] [0.04–33.10]
Enterococcus spp. (MPN/100 mL)	20[a] [5–6546]	15[ab] [5–866]	30[a] [5–724]
Clostridium perfringens (CFU/100 mL)	0[a] [0–7]	0[ab] [0–8]	0[b] [0–10]
HF183 (gene Copies/100 mL)	0.00 [0.00–22.22]	0.00 [0.00–6.15]	0.00 [0.00–10.54]
$NO_3^- + NO_2^-$ (μmol/L)	26.44[a] [11.29–63.61]	17.80[b] [0.44–48.76]	11.51[b] [0.04–66.10]
NH_4^+ (μmol/L)	0.83[a] [0.18–11.70]	1.35[b] [0.18–9.57]	1.51[b] [0.18–6.57]
TDN (μmol/L)	35[a] [18–119]	25[b] [8–85]	23[b] [3–110]
PO_4^{3-} (μmol/L)	1.83[a] [0.40–3.43]	1.28[b] [0.02–2.64]	0.54[c] [0.02–3.48]
TDP (μmol/L)	2.34[a] [0.98–4.54]	1.89[b] [0.18–6.05]	1.27[c] [0.13–15.22]
H_4SiO_4 (μmol/L)	378[a] [131–759]	272[b] [7–600]	140[c] [4–713]
Chl *a* (μg/L)	0.64[a] [0.00–108.47]	1.26[b] [0.05–44.37]	1.66[b] [0.00–39.67]
Turbidity (NTU)	0.52[a] [0.06–19.70]	0.92[b] [0.16–34.90]	1.89[c] [0.14–78.00]
Macroalgal N-content (%)	2.35[a] [0.23–4.30]	1.86[b] [0.75–3.63]	2.02[b] [0.76–5.69]
δ^{15}N-Macroalgal (‰)	+2.25[a] [+0.30 – +6.40]	+3.53[a] [+0.90 – +4.50]	+3.53[b] [+0.17 – +6.20]
δ^{15}N-NO_3^- (‰)	−0.40[a] [−14.90 – +11.50]	+2.60[a] [−19.24 – +10.70]	+3.00[b] [−0.40 – +15.10]
δ^{18}O-NO_3^- (‰)	−0.90[a] [−13.19 – +12.48]	+0.50[a] [−10.65 – +17.89]	+3.00[b] [−8.83 – +15.07]
OSDS Density	19 [12–112]	19 [7–63]	25 [0–81]
Wastewater Pollution Score	17 [16–18]	17 [16–17]	17 [17–19]

Abbreviations: TDN = Total Dissolved Nitrogen; TDP = Total Dissolved Phosphorus; Chl *a* – Chlorophyll *a*.

influence groups and they all fell in the low category (Tables 1, S4). In comparison, on the west side of Hawaiʻi Island, at Kailua-Kona, ~50% of shoreline stations had low wastewater pollution scores, and the other 50% had medium scores (Kamau, 2025), whereas at Puakō, most shoreline stations had medium scores and a few were high (Abaya et al., 2018). Hence, our first hypothesis for this study proved to be incorrect along the Hilo shoreline using traditional indicators of wastewater pollution.

While, the nutrient concentrations at the HGW stations were higher than those measured at the LGW and RW stations, they are generally lower in Hilo compared to other locations impacted by wastewater pollution, where they can be up to two orders of magnitude higher (Amato et al., 2016; Lapointe et al., 2017; Abaya et al., 2018; Brewton et al., 2022). *Enterococcus* spp. concentrations in this study, similar to nutrient concentrations, are low compared to other locations with documented wastewater pollution (Curiel-Ayala et al., 2012; Kirs et al., 2017; Abaya et al., 2018; Brewton et al., 2022) and they were infrequently higher than the USEPA marine recreational water standard of 35 MPN/100 mL (33% percent of our samples, 103/310) and the HDOH beach action value of 130 MPN/100 mL (12% of samples, 38/310, Table S3). However, multiple studies have detected human pathogens associated with wastewater in the absence of measurable FIB concentrations (*reviewed in* Mbae et al., 2024). The lower concentrations are likely due to greater dilution from high rates of rainfall and groundwater recharge in the Hilo Bay watershed. The spatially averaged rainfall across the Hilo aquifer is 2500 mm/yr, of which 76% becomes groundwater and flows out at coastal springs (Engott, 2011). In comparison, across the contiguous US, most rainfall is <1880 mm/yr (NOAA, 2023) and < 10% of runoff flows out as SGD (Sawyer et al., 2016). Reduction in wastewater contaminants within an aquifer may occur and the extent of the dilution can vary depending on the recharge rate of the aquifer compared to the load from the OSDS (Mbae et al., 2024).

Similarly, the δ^{15}N-macroalgal values at the HGW stations were relatively low compared to wastewater polluted waters elsewhere even though their nutrient concentrations were relatively high (Moynihan et al., 2012; Abaya et al., 2018; Brewton et al., 2022; Tyre et al., 2023). The majority of our stations (15/19) had a median macroalgal δ^{15}N of +3.9‰, falling in the little to no wastewater-influence category set by a Hawaiʻi State report on cesspool conversion, with the remainder falling within the wastewater-influenced category. These categories were statistically-derived from a statewide assessment of δ^{15}N in macroalgae and OSDS density for 1-km shoreline segments and are: wastewater-dominated waters ≥ +6‰, wastewater-influenced = +4.0 to +5.9‰, and no wastewater influence ≤ +3.9‰ (Smith et al., 2021). However, this method assumes that background δ^{15}N- NO_3^- is similar among watersheds, which is not the case. For example, across Hawaiʻi Island, the highest average δ^{15}N in macroalgae along 1 km of shoreline was at Puakō (mean ± SE; +6.5 ± 1.7‰) (Smith et al., 2021), compared to our HGW and LGW groups with median values of +2.3‰ and + 3.5‰, respectively. However, the background δ^{15}N- NO_3^- in upland drinking water wells are much higher at Puakō (+4.8 ± 0.4‰, Abaya et al., 2018) compared to Hilo (+0.7 ± 0.6‰, Table S2). That is, the upland δ^{15}N-NO_3^- at Puakō are already in the “wastewater-influenced” category and at Hilo it was far below this cutoff. The contribution of ^{15}N-enriched wastewater resulted in a similar increase of +1.7‰ at Puakō and + 1.6‰ to +2.8‰ at Hilo. The low δ^{15}N- NO_3^- in groundwater are within the range for both agricultural fertilizer inputs and nitrogen-fixing vegetation, like *Acacia koa* (koa), *Falcataria falcate* (Moluccan albizia), and *Trema orientalis* (gunpowder tree), all of which occur in the watershed. Nitrogen-fixing trees are more likely the source as wet forest comprises <70% of the land cover, and cultivated agriculture ~20% in the watershed (Jacobi et al., 2017). Similar low δ^{15}N-macroalgal values in an area of high NO_3^- input from OSDS was noted at Spreckelsville and Paia-Kuau on Maui, where legacy fertilizer N from sugarcane cultivation is likely an important component of the background NO_3^- (Smith et al., 2021). While high δ^{15}N in macroalgae is likely a good indicator of wastewater pollution, moderate values require further analysis (methods/approaches) to determine the magnitude of impact from wastewater pollution.

Because individual wastewater indicators can provide conflicting information regarding wastewater pollution as our study shows, we modified an existing wastewater pollution score developed for West Hawaiʻi to Hilo's rainy climate and estuarine water conditions (*see* Methods for details, Abaya et al., 2018; Waiki et al., 2025). All FW-influence groups fell in the low wastewater pollution category. However, when scores were calculated for individual stations, there were dates when some stations had medium wastewater pollution scores (Table S4). The heavy reliance on *C. perfringens* and δ^{15}N-macroalgal values in the score may result in the lower sensitivity in detecting wastewater pollution in Hilo (Abaya et al., 2018). At our stations,

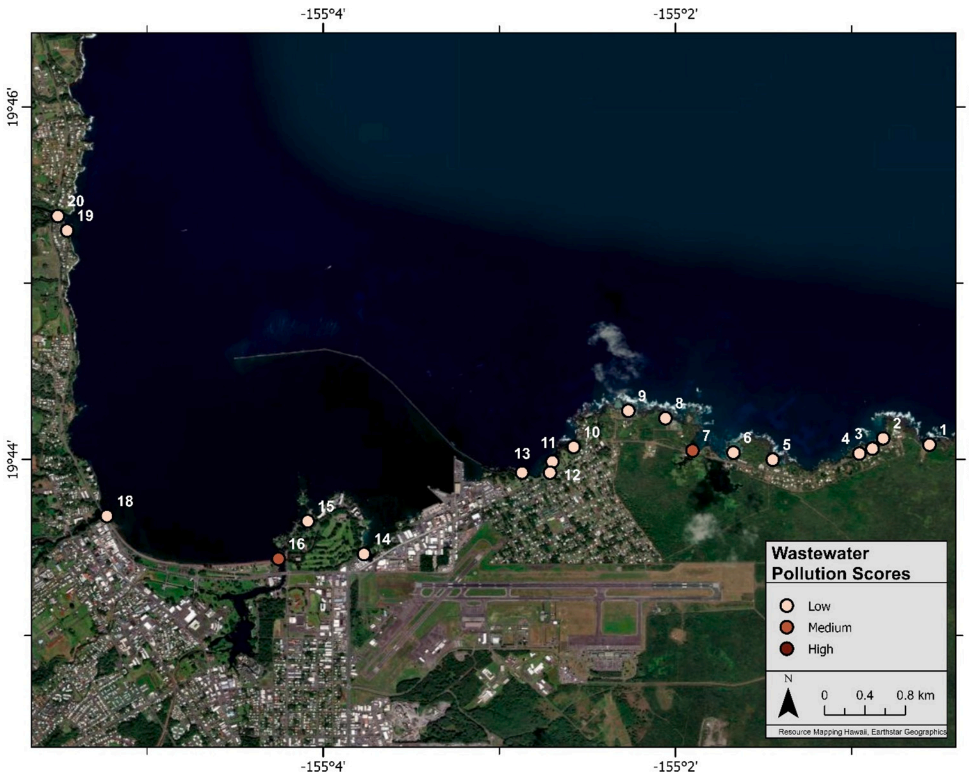

Fig. 3. Wastewater pollution scores for 20 stations (circles with station numbers) along the Hilo, Hawaiʻi, shoreline are based on established and recommended water quality standards, literature values for wastewater indicators, and measured values from this study (fecal indicator bacteria, $\delta^{15}N$ in macroalgae, and nutrients from monthly sampling from July 2020 to October 2021). Wastewater pollution categories were: low (11–17), medium (18–25), and high (26–33). Details on the calculation are provided in the *Methods* section.

C. perfringens was mostly absent or had low concentrations, and $\delta^{15}N$-macroalgal values were often below the range for wastewater likely from freshwater dilution and contributions of other N sources (Tables 1, S3, S4). Potentially including a parameter not impacted by freshwater inputs like OSDS density would improve the sensitivity of the score, especially in an area of high freshwater inputs. OSDS density has been found to be correlated with FIB concentrations (Sowah et al., 2014; Verhougstraete et al., 2015), and other studies have developed a "septic index" to account for OSDS density and proximity to sampling stations (Norat-Ramírez et al., 2019; Smith et al., 2021). To evaluate whether OSDS density improves the sensitivity of the wastewater pollution score, we modified it to include OSDS density, assigning OSDS density a weight of 3 (most reliable wastewater indicator). OSDS density was given a weight of 3 because it has been shown that their septage leaches directly into the groundwater and discharges at shoreline springs in our study area (Waiki et al., 2025). Within the OSDS density parameter, we created three levels: $\leq$ 14 OSDS/km^2 (level 1, least affected by OSDS), 15–61 OSDS/km^2 (level 2, USEPA cutoff for potential groundwater contamination; Yates, 1985), and 62+ OSDS/km^2 (level 3, shellfish bed closures; Duda and Cromartie, 1982). New ranges for the scores are 14–20 (low), 21–31 (medium), and 32–42 (high). Upon modification, 14 additional stations fell in the medium wastewater pollution score range. Four of these stations now have the highest medium scores, St. 11–13, and 18. These four stations have the highest OSDS density (Table S4), with densities >62 OSDS/km^2, significantly above the USEPA OSDS density thought to contribute to groundwater contamination (Yates, 1985). Additionally, St.11–13 were in the vicinity of where dye tracer tests were conducted and presence of wastewater at shoreline springs was confirmed (Waiki et al., 2025).

In addition to groundwater, rivers transport wastewater pollution to coastal waters (Knee et al., 2008; Viau et al., 2011; Verhougstraete et al., 2015; Economy et al., 2019), and our study confirmed their importance to wastewater transport in Hilo. The RW stations had the highest FIB, NH_4^+, and $\delta^{15}N$-macroalgal values (Tables 1, S3, S4). A similar pattern relative to FIB, NH_4^+, as well as sucralose and HF183 was reported in surface waters in Florida near septic tanks (Brewton et al., 2022). Concentrations of the wastewater indicators measured at our RW-influenced stations fall within the range of those previously reported near rivers in Hilo (Wiegner et al., 2013; Strauch, 2017; Wiegner et al., 2017; Economy et al., 2019) and elsewhere in the State of Hawaiʻi (Roll and Fujioka, 1997; Ragosta et al., 2011; Viau et al., 2011). In fact, the highest *Enterococcus* spp. concentrations were consistently observed at the RW stations (Table S3). These findings are supported by output from the NO_3^- stable isotope mixing model, where 8% up to 49% of the NO_3^- at the seven RW stations was from OSDS (Figs. 4, S5). In Hilo, wastewater in the rivers is likely coming from OSDS as the WWTP does not discharge into the rivers and there are no combined sewer overflows. However, there are gravity and force mains on either side of the Wailuku and Wailoa Rivers and pump stations near their mouths, which are old, possibly leaking, and have had documented spills (USEPA, 2024). Storie et al. (unpubl. data) found evidence of both OSDS septage and untreated sewage from microbial community analyses at these locations. Studies elsewhere have found that OSDS density is correlated with riverine FIB concentrations and that OSDS effluent contributes significantly to baseflow discharge (Yang et al., 1999; Burns et al., 2005; Oliver et al., 2014; Sowah et al., 2014; Verhougstraete et al., 2015). The USEPA considers areas with >15 OSDS/km^2 to pose a significant pollution risk to groundwater (Yates, 1985), and the OSDS density within a ~ 1-km^2 around four of the seven RW stations ranged from 25 to 81 OSDS/km^2 (Table S4).

4.2. *OSDS density as a predictor of coastal water quality*

As OSDS septage leaches into groundwater and discharges at

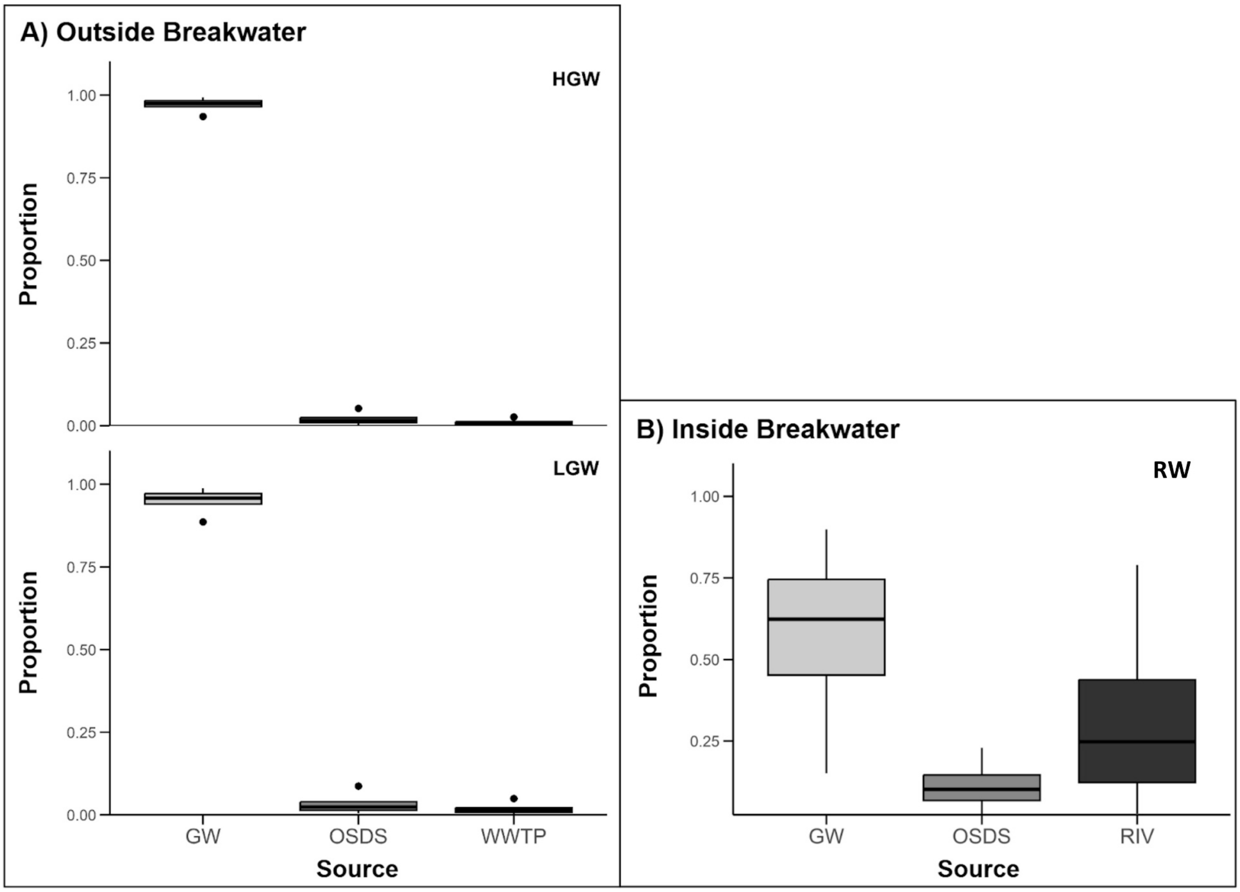

Fig. 4. Estimated proportion contributions to the NO_3^- pool of shoreline stations grouped by freshwater-influence groups (HGW = high groundwater, LGW = low groundwater, RW = river water), and location (A) outside and (B) inside of the Hilo Bay Breakwater in Hilo, Hawai'i. Monthly samples were collected from July 2020 to October 2021. Potential nitrogen sources include groundwater from drinking water wells (GW), on-site sewage disposal systems (OSDS), a manhole prior to treated wastewater from the wastewater treatment plant (WWTP) being discharged at the outfall, and rivers (RIV) (Table S2). The WWTP was only included as an NO_3^- source for stations outside of the Hilo Bay Breakwater as the outfall is located there. RIV is only included as a NO_3^- source inside the Hilo Bay Breakwater as there are only rivers located inside of the Hilo Bay Breakwater. Results are derived from Bayesian stable isotope mixing models in R (*simmr*) using $\delta^{15}N$- and $\delta^{18}O$-NO_3^- values. Boxes represent the 25–75% posterior credible interval, horizontal lines indicate the median (50% quantile), and whiskers represent the 2.5–97.5% posterior credible interval.

shoreline springs (Abaya et al., 2018; Wiegner et al., 2021; Kamau, 2025; Waiki et al., 2025), we expected that concentrations of wastewater indicators would be positively correlated with OSDS density. Our findings are mixed relative to our second hypothesis. OSDS density was positively correlated with most nutrient concentrations: $NO_2^- + NO_3^-$, TDN, PO_4^{3-}, and TDP (Fig. 5). However, it was negatively correlated with *C. perfringens* and NH_4^+ (Fig. 5). Hydrologic and biogeochemical processes occurring within river water likely complicated this analysis. The three stations with the lowest cesspool density (St. 15–17) were all in the RW group. As noted above, for the RW group, wastewater more likely originates from OSDS farther up in the watershed and is transported to the shoreline stations by rivers and coastal currents. However, rivers and groundwater have different background concentrations of wastewater indicators (Table S2). Compared to groundwater collected inland from the area of OSDS, the rivers had lower concentrations of nutrients except for NH_4^+ (Table S2), due to uptake by riparian and in-stream plants and algae during baseflow (Larned and Santos, 2000). Low nutrient concentrations in the river and low cesspool density adjacent to some of the RW stations limited the strength of the positive correlation between nutrients and cesspool density (r = 0.11 to 0.18). Similarly, high *C. perfringens* and NH_4^+ were observed at the RW group stations. *Clostridium perfringens* is one of the most reliable wastewater indicators, and NH_4^+ comprises the greatest percentage of dissolved inorganic N in OSDS septage (Wiegner et al., 2016; Abaya et al., 2018; Wiegner et al., 2021). While these RW group stations have a low cesspool density nearby, the river is bringing freshwater and wastewater from farther away that is transformed by biogeochemical processes to these stations.

4.3. OSDS vs. WWTP contributions to wastewater pollution

In Hilo, the Hawai'i County and State agencies need to know whether OSDS or the WWTP are contributing more to wastewater pollution for management actions. One approach to tease this apart, is to use the stable isotopes of NO_3^- in a mixing model. For stations outside of the breakwall that are directly impacted by both OSDS and WWTP, we found that OSDS and WWTP are contributing a relatively similar percentage of NO_3^- to the NO_3^- pool [OSDS: HGW (1.5%), LGW (2.4%); WWTP: HGW (0.8%), LGW (1.4%)]. Previous studies on Hawai'i Island have found groundwater to be the dominant source of NO_3^-, with wastewater from OSDS contributing up to 40% and WWTP effluent contributing up to 26% of NO_3^- (Wiegner et al., 2016; Wiegner et al., 2021; Panelo et al., 2022; Waiki et al., 2025). At our RW-influenced stations, groundwater was also a significant source of NO_3^- (62%), followed by river water (25%), and OSDS (10%) (Fig. 4). These results suggest that management actions need to be taken to reduce both pollution sources. Note, that contributions of OSDS pollution into this area in Hilo are not solely from the shoreline community, but are an accumulation of pollution from all the homes in the entire watershed.

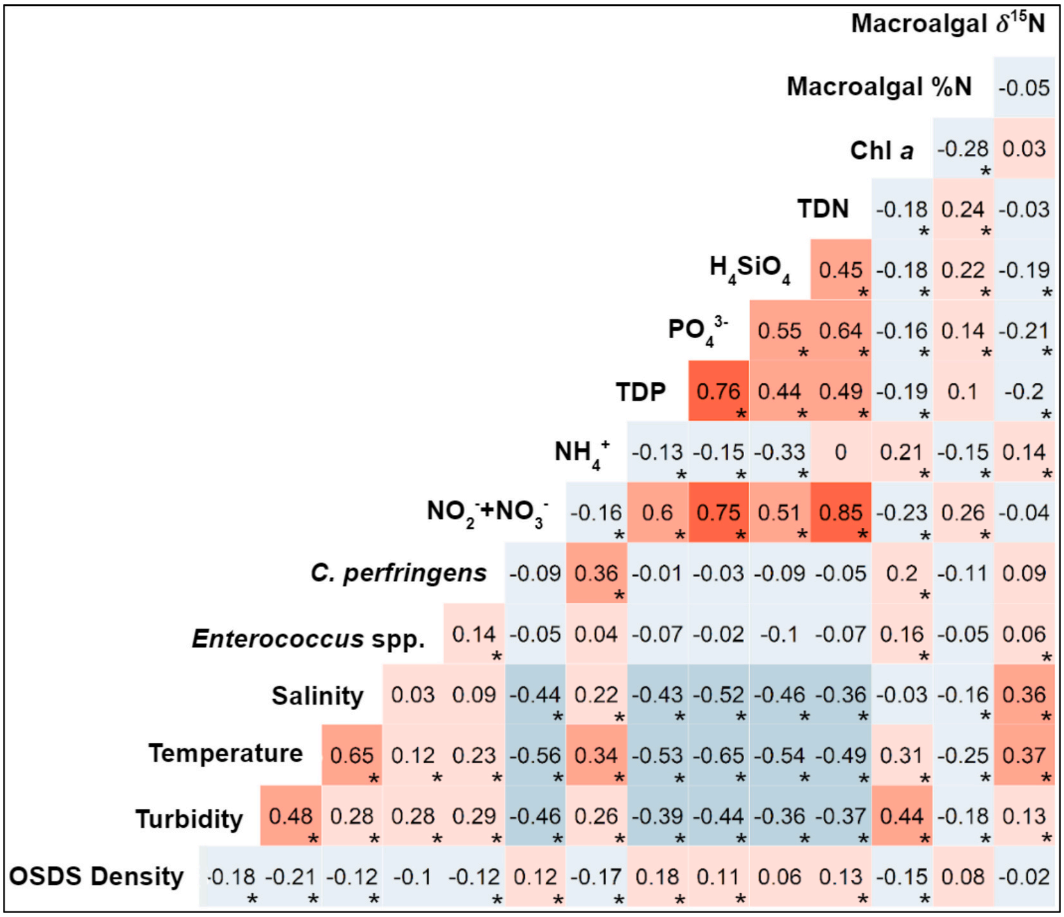

Fig. 5. Correlations between onsite sewage disposal systems (OSDS) density, wastewater indicators, and physiochemical parameters from 20 shoreline stations in Hilo, Hawaiʻi, sampled monthly from July 2020 to October 2021. Spearman's rho (ρ) is shown on each tile with red and blue tiles representing positive and negative correlations, respectively, and asterisks indicating a significant result (α = 0.05). Abbreviations: *C. perfringens* = *Clostridium perfringens*, TDP = Total Dissolved Phosphorus; TDN = Total Dissolved Nitrogen; Chl *a* – Chlorophyll *a*. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

Another potential way to evaluate relative contributions of OSDS and WWTP in an area, can be to use next-generation sequencing to discriminate microbial communities among wastewater sources. A concurrent study to ours found that OSDS septage and WWTP influent (untreated wastewater) have distinct microbial communities, and that both types of wastewaters were detected at these stations (Storie et al., unpubl. data).

5. Conclusion

Groundwater and rivers are important vectors of wastewater to coastal waters worldwide, and both are important to Hilo Bay. However, in areas with high freshwater inputs from both sources, such as Hilo, detection and monitoring of wastewater in the environment is challenging. The lack of accurate detection and adequate monitoring of wastewater using traditional indicators could prevent development and implementation of wastewater collection and treatment improvement projects. If state officials were to assume that wastewater is not an issue because of low wastewater indicator concentrations, then proper policy decisions cannot be made. For example, the use of rainfall in the Hawaiʻi Cesspool Hazard and Assessment and Prioritization Tool (HCPT) led to Hilo being listed as a Priority 2 area in which OSDS pose a "significant contamination hazard", and not a Priority 1 area where OSDS pose a "pronounced contamination hazard" (Mezzacapo and Schuler, 2021). The tool assumes that in higher rainfall areas, OSDS effluent is diluted, reducing its pollution impact (Mezzacapo and Schuler, 2021). Our findings support the first part of this assumption because many of the wastewater indicators we measured had relatively low concentrations compared to other locations known to be impacted by wastewater. However, dye tracer tests in the Hilo area showed that wastewater from homes reached shoreline springs within hours to days and demonstrated that wastewater was an immediate pollution threat (Waiki et al., 2025). This example from the HCPT illustrates that measuring different wastewater indicators and developing geospatial hazard-based prioritization tools do not fully capture the hydrogeology of an area that is needed to understand its wastewater pollution threat shown by dye tracer tests. Additionally, using NO_3^- stable isotopes in a mixing model, we were able to parse out wastewater pollution impacts from OSDS and WWTP. Wastewater pollution from both sources was present in the coastal waters of Hilo. This finding is important because it can inform management actions, such as expanding the sewer line to cover the 70% of Hilo not serviced by the WWTP, as well as renovating the WWTP. The USEPA administrative consent order will ensure that these actions take place. Additionally, relocating the Hilo WWTP outfall to an area where its potential impact to human and ecosystem health is less would greatly improve community relations with Hawaiʻi County. An even better solution to moving the outfall, is to treat the wastewater to a higher degree, like R-1 recycled water (the highest grade of safety for non-potable reuse), so that it can be reused for other purposes and not be discharged to the ocean. Currently, neither are part of the USEPA administrative consent orders nor are they being considered by Hawaiʻi County because of cost. But, there is precedent in Hawaiʻi County to potentially pursue this latter solution. In 2025, Hawaiʻi County reached an agreement to settle a lawsuit that requires the County to upgrade a WWTP in Kailua-Kona (west side of Hawaiʻi Island) to produce high quality effluent suitable for irrigation by 2029 (Dayton, 2025). In Florida, the State government has recognized that OSDS, septic tanks, do not properly function in areas where the water table is high (Brewton et al., 2022); this has also been documented in along Michigan's Lower Peninsula (Verhougstraete et al., 2015), North Carolina (Humphrey Jr et al., 2011; Cahoon et al., 2016), and Jepara, Indonesia (Adyasari et al., 2018). As a result, in 2023, the Florida legislature passed House Bill 1379 (Laws of Florida Chapter No. 2023–169) that requires homes to connect to a central sewer line if available or upgrade to a N-reducing OSDS that can reduce N in the septage by 65%. A similar approach should be considered for Hilo.

Overall, our results illustrate the importance of measuring a variety of wastewater indicators (FIB, nutrients, N stable isotopes in macroalgae/ NO_3^-) and using multiple approaches (dye tracer tests, pollution scoring, stable isotope mixing modeling) to understand the pollution threat in Hawaiʻi, as well as other locations with suspected wastewater pollution, especially those with high freshwater inputs. Generally, funding to address wastewater pollution issues requires documenting impacts to water quality either through measurements of traditional wastewater indicators or dye tracer tests. As our study shows in conjunction with Waiki et al. (2025), the latter approach is better especially in an environment with high freshwater inputs. However, if neither of these types of measurements are needed, a cost-saving approach could be to use OSDS density and proximity to nearshore waters to prioritize which areas undergo OSDS conversion first.

CRediT authorship contribution statement

Joseph W.P. Nakoa: Writing – review & editing, Writing – original draft, Visualization, Supervision, Investigation, Formal analysis, Data curation, Conceptualization. **Tracy N. Wiegner:** Writing – review & editing, Writing – original draft, Supervision, Resources, Project administration, Methodology, Investigation, Funding acquisition, Formal analysis, Data curation, Conceptualization. **Steven L. Colbert:** Writing – review & editing, Writing – original draft, Supervision, Resources, Project administration, Methodology, Investigation, Funding acquisition, Formal analysis, Data curation, Conceptualization. **Karla J. McDermid:** Writing – review & editing, Writing – original draft, Methodology, Conceptualization. **Shayla M.P. Waiki:** Writing – review & editing, Supervision, Data curation, Conceptualization. **Nicholas M. Storie:** Writing – review & editing, Writing – original draft, Data curation. **Devon K. Aguiar:** Writing – review & editing, Supervision, Methodology, Data curation, Conceptualization. **Craig E. Nelson:** Writing – review & editing, Funding acquisition, Conceptualization.

Declaration of competing interest

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.marpolbul.2026.120264.

Data availability

Data will be made available on request.

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