KĀNE‘OHE BAY AND WATERSHED: A REVIEW OF ITS HYDROGEOLOGY, HISTORICAL AND CURRENT SOURCES OF POLLUTION AND METHODS TO QUANTIFY POLLUTANT FLUXES AND SOURCES

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Abstract

Groundwater is an important source of water and nutrients to streams, estuaries and the coastal ocean. Groundwater, however, can also act as a conduit for pollutants derived from land-use (e.g. agriculture, on-site sewage disposal systems (OSDS), industry) to surface waters. Small watersheds, steep topography, and highly conductive geologic strata commonly characterize high volcanic islands, such as O‘ahu, which can result in elevated groundwater fluxes to streams and the coastal ocean. In order to assess the current state of our understanding of nutrient and pollution sources as well as their pathways in Kāne‘ohe watershed, a synthesis of the existing literature is required. This report on the hydrogeology of Kāne‘ohe Watershed and Bay focuses on historical and current sources of ground and surface water contamination to streams and the coastal area of the bay. Additionally, the review will include modern methods to quantify both groundwater inflow and surface water fluxes as well as approaches to quantify and source water contamination.

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1. Introduction

Water resources on high volcanic Pacific Islands are generally sourced almost exclusively from groundwater. This groundwater is stored in a freshwater lens (i.e. Ghyben-Herzberg lens) or in high-level water in dike complexes and is replenished via infiltration of rainfall. This dependence on rainfall as well as the remote locations of Pacific Islands means these water resources are less resilient to naturally- and anthropogenically-sourced stressors. Furthermore, the dependence on groundwater for municipal water supply means that extra care must be taken to not contaminate this reservoir.

From an environmental perspective, groundwater can also be an important source of nutrients to streams, estuaries and the coastal ocean. Groundwater may additionally act as a conduit for pollutants derived from land-use (e.g. agriculture, on-site sewage disposal systems (OSDS), industry) to surface waters. Pollutants are defined as any constituent in a natural system that causes detrimental change to that system (HDOH, 2012). Examples of naturally- and anthropogenically- sourced pollutants to coastal waters and streams can include inorganic nutrients (such as nitrate and orthophosphate), pesticides, and other organic compounds (HDOH, 2012; HDOH, 2015) originating from sources such as natural organic matter decomposition in soils or direct infiltration from cesspools and fertilized agricultural fields, as well as effluent injected into the subsurface from wastewater treatment facilities. Substantial groundwater-derived nutrient fluxes can affect coastal water geochemistry and biology negatively and thus can result in poor water quality (see Table 1; HDOH, 2014), coastal eutrophication, and harmful algal blooms (Valiela et al., 1997; Dailer et al., 2010).

Nutrients and pollutants in groundwater can be delivered to the coastal ocean primarily via two pathways (1) direct groundwater discharge into the coastal ocean by means of submarine groundwater discharge (SGD) and (2) groundwater discharge into streams that then flow to the ocean. Surface water-groundwater interactions can occur in streams, estuaries, and the coastal ocean. SGD inputs are not only volumetrically significant (up to 2 to 4 times the water volume of surface runoff), but can also carry nutrient loads that surpass surface runoff fluxes (Kroeger, et al., 2007). Moreover, SGD is not only an important consideration to water and nutrient budgets on a local scale, but also globally, where terrestrially-sourced SGD is estimated to be 2,400 km^3/year globally (Zekster, 2000). This is particularly the case in island settings, where terrestrially-sourced SGD from major Pacific Islands is estimated to equal 615 km^3/year, or about 25% of the global SGD flux (Zekster, 2000). Despite accounting for a relatively small landmass, high volcanic islands are commonly characterized by small watersheds, steep topography, and highly conductive substrates, resulting in elevated groundwater fluxes to streams and the coastal ocean (Zekster, 2000; Knee & Paytan, 2012; Kwon et al., 2014).

SGD fluxes are influenced by physical processes on both a local and global scale. SGD connects terrestrially-derived groundwater and recirculated seawater with the coastal ocean through the subterranean estuary (STE) (Moore, 1999). The STE is a biogeochemically active area in the subsurface, which can result in chemical transformations of dissolved nutrients and trace metals prior to discharging to the coastal ocean as SGD (Moore, 1999). The STE is influenced by processes such as tidal cycles, sea-level changes, seasonal groundwater withdrawal, and dredging of channels due to the changes in exchange rate between fresh and saline water (Moore, 1999). Changes in the hydraulic gradient between groundwater and the coastal ocean directly impact SGD rates, where SGD responds to both semi-diurnal and semi-monthly tidal fluctuations (Burnett et al., 2006). SGD is expected to intensify during low tide and semi-monthly spring tide when the hydraulic gradient is at a maximum (Burnett et al., 2006).

In addition to direct groundwater near-shore discharge via SGD, groundwater inflows and hyporheic exchange processes (Fig. 1) in streams can also deliver large quantities of groundwater to the coastal ocean. The hyporheic zone is an interface between groundwater and stream water. This zone of interaction can result in natural attenuation of contaminants because of the transition from anoxic to oxic conditions and thus intensified microbial activity (Winter et al., 1998).

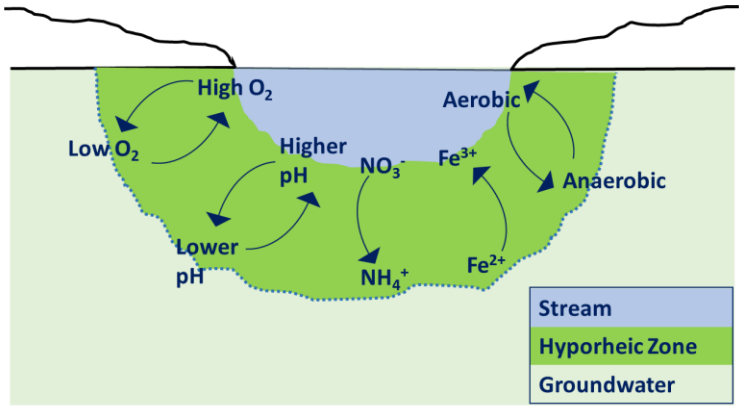

Figure 1. Idealized hyporheic exchange processes (after Winter et al., 1998).

Water quality can be negatively impacted by both groundwater and surface water inputs. The Hawai‘i Department of Health (HDOH) outlines criteria for water quality (Table 1) on the basis of type of reservoir (e.g. streams, wetlands, marine) as well as usage (e.g. recreational, preservation, drinking water). Inland waters, which include streams, reservoirs, and wetlands are classified as Class 1 (includes preservation and drinking water resources; these waters are protected from both point and nonpoint contamination) or Class 2 (primary objectives include the protection of recreational waters and aquatic life). Similar classifications exist for marine waters, which include embayments, coastal and oceanic waters, and are classified as Class A (preservation) or Class AA (protection of recreational waters, aquatic life). For all natural waters in Hawai‘i, elevated bacterial counts, particularly from fecal indicator bacteria such as *Enterococcus*, and *Clostridium perfringens*, may be indicative of pollution from wastewater (Table 1; HDOH, 2014).

Table 1. Selected state of Hawaiʻi water quality criteria for Class II (streams) and Class AA (embayments) by season, defined by the maximum acceptable geometric mean for the criteria of interest (HDOH, 2014). Seasons are distinguished between wet (November 1 through April 30) and dry (May 1 through October 31) seasons. Nutrient concentrations (Total Nitrogen (TN), Total Phosphorus (TP), and Nitrate + Nitrite (NO_3^- + NO_2^-) in ug/L). Turbidity (Nephelometric Turbidity Units or N.T.U.), a measurement of scattered light intensity, commonly increases with storm runoff. Enterococcus counts (colony forming units (cfu)/100 mL) may be indicative of human sewage. Additional water quality parameters not listed include temperature, pH, salinity, and dissolved oxygen.

	TN (ug/L)	NO_3^- + NO_2^- (ug/L)	TP (ug/L)	Turbidity (N.T.U.)	Enterococcus (cfu/100mL)
Streams (Wet Season)	250	70	50	5.0	> 35
Streams (Dry Season)	180	30	30	2.0	
Embayments (Wet Season)	200	8	25	1.5	
Embayments (Dry Season)	150	5	20	0.40	

This review provides an overview on the hydrogeology of Kāneʻohe Watershed and Bay, on the windward side of Oʻahu, Hawaiʻi, focusing both on historical and current sources of ground and surface water contamination to streams and the coastal area of the bay. Additionally, I discuss modern methods to quantify both groundwater inflow and surface water fluxes as well as approaches to quantify and source water contamination.

2. Kāneʻohe Bay and Watershed

Kāneʻohe Bay (21°28′ N, 157°48′ W) is a reef-dominated embayment located on the windward side of Oʻahu and has been historically, as well as currently, susceptible to pollution (Bahr et al., 2015; Smith et al.; 1981). The steep topography of the Koʻolau Mountains coupled with a permeable alluvium results in a relatively fast groundwater flushing time and thus less degradation of pollutants (Jokiel, 2004; Takasaki & Mink, 1985). From 1951 until 1979, municipal sewage effluent (primarily and secondarily treated) was directly discharged to the southern portion of Kāne‘ohe Bay, resulting in low oxygen conditions, high primary productivity in the water column, and coral reef decline (Smith et al., 1981). Excess nutrient loading in Kāne‘ohe Bay has historically caused a shift from a coral-dominated to an algal-dominated system (Smith et al., 1981; Jokiel, 2004; Bahr et al., 2015). While surface runoff has been thought to be the major pathway responsible for delivering pollutants (suspended particulate matter, nutrients, trace metals) to Kāne‘ohe Bay (e.g. De Carlo, et al., 2007; Hoover & MacKenzie, 2009; Hédouin et al., 2009), recent research has indicated that SGD-derived baseline nutrient inputs (Dulai et al., 2016) are greater than those coming from surface runoff.

2.1. Climate

The Hawaiian island chain is located between 19 and 28° N and 154 and 178° W (Lau & Mink, 2006). The yearlong average temperature in Kāne‘ohe is 25.1 ± 3.8° C (Smith et al., 1981). The windward slopes of the Hawaiian Islands receive high quantities of rainfall due to orographic lifting and prevailing trade-wind patterns (Giambelluca et al., 2013). Trade-winds blow from a northeast direction and are intercepted on the island of O‘ahu by the Ko‘olau Mountains (Figure 2; Giambelluca et al., 2013; Lau & Mink, 2006).

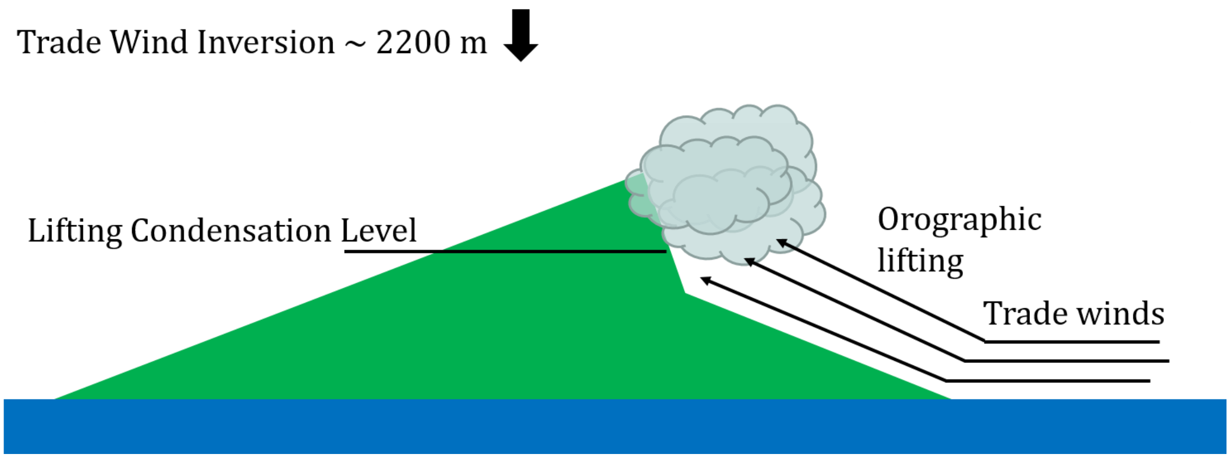

Fig. 2. Schematic of processes behind orographic rainfall in Hawai‘i (after Giambelluca et al., 2013). Trade-winds blow from the northeast and orographic lifting against the mountains causes cloud condensation and orographic rainfall on windward slopes. Trade wind inversion occurs at about 2200 m, where compression of air masses causes air temperature to increase with altitude, limiting the elevation which substantial orographic rainfall occurs (Giambelluca et al., 2013; Lau & Mink, 2006)

While precipitation tends to be relatively consistent in the upper slopes, rainfall on O‘ahu’s coastal plains occurs mostly (about 70% of annual rainfall) from October through April (Hunt, 1996; Leta et al., 2016). Additionally, there can be significant year-to-year differences in rainfall during the wet season due to Kona Storms (or subtropical cyclones), which can result in torrential rain, thunderstorms, and flash flooding (Lau & Mink, 2006). The annual average water budget of the watershed can be broken down to 2400 mm precipitation, 1350 mm evapotranspiration, 800 mm recharge, and 350 mm surface runoff, or about 56%, 33%, and 11% of total precipitation, respectively (Giambelluca et al., 2013; Safeeq et al., 2012).

2.2. Geology

The geologic composition of Kāneʻohe watershed is dominantly basaltic with overlying alluvium. Koʻolau basalt, which is tholeiitic, is the prevailing type of basalt (Stearns & Vaksvik 1935). Individual flows are on average 10 ft thick, however it can range from 2 to 80 ft in thickness (Figure 3; Stearns & Vaksvik 1935, Wentworth, 1951; Lau & Mink, 2006). Interspersed are younger Honolulu volcanics, which are more varied geochemically, but are mostly alkalic in composition. The marginal dike zone is concentrated at the higher elevations, and the dike complex underlies most of Kāneʻohe Bay and Watershed (see Section 2.4 below for further explanation; Takasaki & Mink, 1985). The coastal lowlands have overlying sand, clay, silt, and gravel alluvium, and covers about 60% of Kāneʻohe Bay watershed (Hunt, 1996). Soils in the study area are moderately to quickly permeable and are predominantly utisols (kaolinite-rich, high capacity for P fixation), oxisols (rich in oxide-clay minerals, high capacity for P fixation), and inceptisols (Figure 3; Hawaiʻi Soil Atlas).

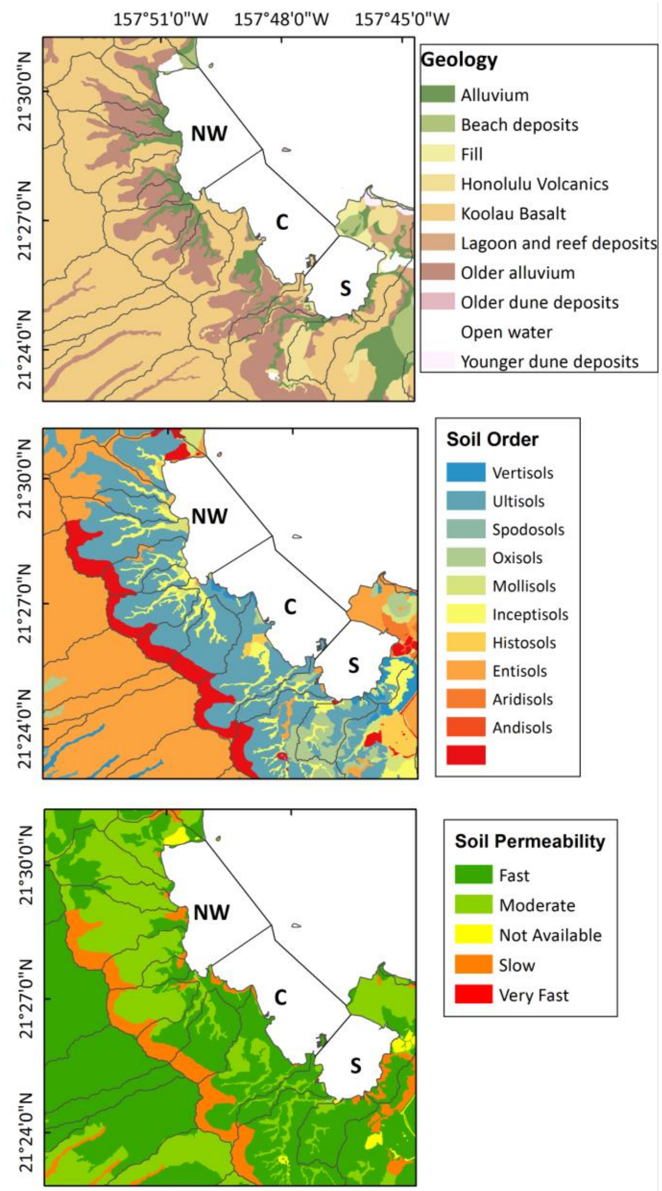

Figure 3. (A) Geology, (B) soil order, and (C) soil permeability of Kāneʻohe Bay and Watershed (Sherrod et al., 2007; Hawaiʻi Soil Atlas).

2.3 Sub-watersheds of Kāneʻohe Bay

Kāneʻohe Bay and Watershed is subdivided into three sectors (northwest, central and south) and further subdivided into fourteen steep amphitheater-shaped sub-watersheds (Laws & Redalje, 1979; Smith et al., 1981; Table 2). Land-use in sub-watersheds in the northwest sector (Hakipuʻu, Waikāne, Waianu, Waiāhole, Kaʻalaea, Haiamoa, and Waiheʻe) is primarily agriculture and preservation land (Dulai et al., 2016). In the central sector sub-watersheds (Kahaluʻu, ʻĀhuimanu, and Heʻeia), land-use shifts to principally low-intensity residential development and preservation land (Koʻolaupoko Watershed Management Plan, 2012). The sub-watersheds in the southern sector (Keaʻahala, Kāneʻohe, Kāwā, and Puʻu Hawaiʻiloa) are considerably more urban, and Puʻu Hawaiʻiloa is almost exclusively military (Koʻolaupoko Watershed Management Plan, 2012). Moreover, the progressive population and urbanization from north to south can be observed through the increasing percentage of impervious surface (defined by paved surfaces such as roads and parking lots) and stream channelization (Table 2). Many streams have been channelized as a means of flood control, however this modification also results in lower residence times for pollutants in the stream so they have a higher chance of reaching Kāneʻohe Bay and eliminates the benefits of hyporheic exchange on contaminant remediation (Koʻolaupoko Watershed Management Plan, 2012).

Table 2. Kāneʻohe sub-watersheds by sector. Baseflow is estimated to be 70% of daily mean stream flow (Koʻolaupoko Watershed Management Plan, 2012). 303(d) originates from the Clean Water Act and is a list of water bodies which do not meet water quality standards due to pollutant(s) exceeding an acceptable level.

Sector	Sub-watershed	Stream length (km)	% Impervious surface	Listed pollutants 303(d)	Stream Baseflow (mgd)
Northwest	Hakipuʻu	3.9	2.03	Not listed	0.41
	Waikāne	14.6	0.25	Not listed	1.94
	Waianu	N/A	2.59	Not listed	N/A
	Waiāhole	10.1	0.69	Nitrite/nitrate	2.89
	Kaʻalaea	N/A	4.99	Not listed	1.29
	Haiamoa	1.5	15.08	Not listed	0.81
	Waiheʻe	15.7	2.52	Nutrients	3.04
Central	Kahaluʻu	27.2	13.06	Turbidity	0.90
	ʻĀhuimanu		1.21	Not listed	2.22
	Heʻeia		18.41	Nitrite/nitrate	1.16
South	Keaʻahala	3	56.99	Nitrite/nitrate, Total N&P, turbidity, trash	1.04
	Kāneʻohe	32.6	22.77	Nutrients, turbidity, dieldrin	5.11
	Kāwā	0.6	40.17	Nutrients, turbidity, SS	1.87
	Puʻu Hawaiʻiloa	2.8	49	Not listed	N/A

2.4. Hydrogeology

The Koʻolau range has numerous vertically intrusive volcanic rocks, or dikes with an average thickness of 1.5 – 1.8 m (Takasaki & Mink, 1985; Shade & Nichols, 1996; Lau & Mink, 2006). These dikes form almost impermeable barriers allowing for storage of water in between individual dikes in the pore spaces of the basalt. The dike complex refers to areas where dikes make up more than 10% of the total mass of rock (Takasaki & Mink, 1985). This is compared to the marginal dike zone, or where dikes comprise less than 5% of the total mass and have a substantial storage capacity for high-level groundwater (Takasaki & Mink, 1985). In some areas, these "dike aquifers" also feed streams. Due to the steep topography and high level dike impounded groundwater that is generally characteristic of windward Hawaiian watersheds, these streams are generally gaining in the dike complex region, losing in upstream reaches and gaining in the coastal plain.

The basal lens, or groundwater which floats on top of seawater according to the Ghyben-Herzberg relation, is found in the lowland coastal plains (Stearns & Vaksvik, 1935; Lau & Mink, 2006). The basal lens and dike complex are hydraulically connected to the coastal zone, and may discharge to both streams as well as through the STE to the nearshore environment (Lau & Mink, 2006).

Surface flow in the watershed primarily occurs through stream flow. Kāneʻohe Bay is fed by seven perennial groundwater-fed streams (Jokiel, 2004). Of streams which flow into Kāneʻohe Bay, Kāneʻohe Stream has the highest discharge rate (7.3 mgd) and Hakipuʻu has the lowest discharge rate (0.59 mgd), on average (Koʻolaupoku Watershed Management Plan, 2012). By sector, average total stream flow into the northwest, central, and southern sectors is 14.8, 6.1, and 11.5 mgd, respectively (Koʻolaupoku Watershed Management Plan, 2012). An estimated 31 billion gallons/year of freshwater enters Kāneʻohe Bay and Watershed (Takasaki & Mink, 1985).

2.5. Nearshore environment

Kāneʻohe Bay is divided into inshore (average depth = 6.1 m) and offshore (average depth = 0.3 – 2.1 m) portions, which comprise of about 66% and 34% of the total surface area, respectively (Jokiel, 2004). The average salinity in the nearshore environment ranges from 31 to 35 (Smith, 1981; Jokiel, 2004). Inner Kāneʻohe Bay has 28 km shoreline, has a mean water depth of 8 m (water depths vary; about 47% of the nearshore bay is at depths greater than 10 m, and about 33% is less than 1.5 m depth), and includes surface area of 31 km^2 (Smith et al., 1981). The southern sector, which includes the historical sewage outfall covers 10.2 km length of shoreline, with a mean water depth of 9.5 m, and covers 8.4 km^2 area (Smith et al., 1981). The central sector stretches 8.8 km of shoreline length, has a mean water depth 9.8 m, and covers 12.3 km^2 area (Smith et al., 1981). The northwestern sector covers 9.0 km of shoreline, has a mean water depth of 6.1 m and includes the remaining 10.8 km^2 surface area of Kāneʻohe Bay (Smith et al., 1981).

3. History of Kāneʻohe Bay – land-use and environmental perturbations

3.1. Polynesian era, European colonization & plantation era

Kāneʻohe Bay area was originally settled by Polynesians around 1250 C.E. By about 1450 C.E. (Kittinger et al., 2011), Kāneʻohe Watershed was originally divided into nine ahupuaʻa or mauka to makai (ridge to reef) management areas (Jokiel, 2004). The ahupuaʻa system considered both agriculture and aquaculture in consort (Jokiel, 2004). Agriculture included loʻi kalo, which diverted stream water for irrigation of taro and other plants (Bahr et al., 2015; Jokiel, 2004). Aquaculture included the construction of over 30 fishponds in the nearshore area of Kāneʻohe Bay, and was estimated to cover about 30% Kāneʻohe Bay's coastline by the 19th century (Devaney et al., 1982). Taro was the dominant crop until the 1870's, when large-scale plantations replaced many smaller loʻi kalo operations (Devaney et al., 1982).

European colonization during the 18th century caused a rapid drop in Hawaiian population due to the influx of unfamiliar disease (Bahr et al., 2015). This population decline and thus shift in power to European colonials led to an increase in single crop (primarily pineapple, sugarcane and rice) plantations (Devaney et al., 1982). At its peak, pineapple production on the windward side of Oʻahu covered nearly over 1000 Ha (Altizer et al., 2011).

With the growth of agricultural activities, population density in Kāneʻohe Watershed increased from 5,000 before 1920 to 56,000 people in 1978 (Smith, 1981), and along with it wastewater production. Until 1963, when the first sewer line was introduced in the area, Kāneʻohe town was served exclusively by OSDS, which were estimated to discharge about 1,000 m^3/day of effluent to the environment (Smith, 1981). This excess loading from OSDS offers a chronic source of nutrients and other contaminants (see Section 4.1.) to natural waters.

Population increases were accompanied by the establishment of military bases in the watershed. The construction of the Kāneʻohe Marine Corps Air Station (1939 – 1945) resulted in substantial dredging and filling of Kāneʻohe Bay (Smith et al., 1981; Bahr et al., 2015). An estimated 11,616,300 m^3 of material was dredged during this period, including 25 out of 79 patch reefs (Smith et al., 1981; Bahr et al., 2015; Jokiel, 2004). Additionally, between 1946 and the 1950's, at least 12 fishponds were filled. These perturbations led to significant coral reef decline (Smith et al., 1981; Bahr et al., 2015; Jokiel 2015).

3.2. Sewage outfall diversion

In addition to major land-use changes and increased erosion and sedimentation, the direct dumping of untreated sewage into Kāneʻohe Bay further impaired water quality and contributed to coral reef cover decline. The pre-sewage diversion experiment lasted from 1951 until 1979 (Smith et al., 1981). From 1951 until 1972, the Kāneʻohe Marine Corps Air Station dumped untreated sewage to the southern portion of the Bay (Smith et al., 1981; Bahr et al., 2015). The City and County of Honolulu Public Works department also dumped secondarily treated sewage to the Bay from 1961 until about 1978 (Smith, 1981). By 1977, the average sewage flux to Kāneʻohe Bay was about 1.8×10^4 m^3/day (Hunter & Evans, 1995). Excess nutrient loading and organic material oxidation resulted in eutrophication, low oxygen conditions, and a shift from a coral-dominated to an algal-dominated environment (Smith, 1981; Bahr et al., 2015; Jokiel, 2004).

Treated sewage was eventually diverted to the open ocean, which led to substantial recovery of coral reef populations and improved water quality. Sewage discharge to southern Kāneʻohe Bay stopped in 1979, following the acknowledgement of the significant coral decline in Kāneʻohe Bay (Smith, 1981; Bahr et al., 2015). Sewage discharge to central Kāneʻohe Bay in ʻAhuimanu continued until 1988, but was eventually connected to the open ocean outfall (Jokiel, 2004). Post-sewage diversion, coral reef cover doubled and particulate nutrient fluxes decreased by 80% for nitrogen and 60% for phosphorus (Smith, et al., 1981).

3.3. Erosion and Urbanization

Population growth, increased land cultivation, and growth of animal husbandry from the late 19th century through the mid-20th century contributed to deforestation, causing increased rates of erosion and sedimentation (Bahr et al., 2015). The area of land for livestock grazing more than quadrupled in Kāneʻohe between 1880 and 1969 (Bahr et al., 2015). In Heʻeia, the major crop shifted from taro (until the 1870's) to rice and sugarcane (1870's to 1910's) to pineapple (1910's to 1920's), reflecting changes in land-use over a short period of time (Devaney et al., 1982). In the mid-20th century, mangroves (which have since migrated to the downstream nearshore environment) were introduced in upland farms as a means to prevent erosion (Devaney et al., 1982). Substantial erosion (which is exacerbated by dominance of moderate to very rapidly permeable soils in the area, Fig. 3) can result in high sediment and nutrient loads to the coastal environment, particularly during high rainfall events, which result in an overall decline in water quality and clarity due to excess particulate nutrient loading.

Rapid urbanization, particularly in the southern sector of Kāneʻohe Watershed, has historically resulted in an increase in trace metals and pesticides. Oysters have had comparatively higher concentrations of trace metals and pesticides in the southern sector compared to the northern sector (Hunter et al., 1995). Specifically, concentrations of lead, chromium, copper, and zinc were statistically significantly higher in oysters collected from the southern (Kāneʻohe Stream, Makani Kai Marina, Marine Base) sector compared to the northern (Waikāne Stream) sector (Hunter et al., 1995).

4. Current and future issues

Current sources of pollutants to Kāneʻohe Bay and Watershed include on-site disposal systems (OSDS), leakage from sewer lines, urbanization, agricultural runoff, and erosion. While the sewage diversion project greatly improved water quality in Kāneʻohe Bay, the high number OSDS in Kāneʻohe Watershed (1485 OSDS units; Figure 4) and aging sewer infrastructure still pose a wastewater-derived threat to the bay's water quality (Whittier & El-Kadi; 2009, 2014). Legacy contaminants, such as dieldrin (an organochloride insecticide), which are

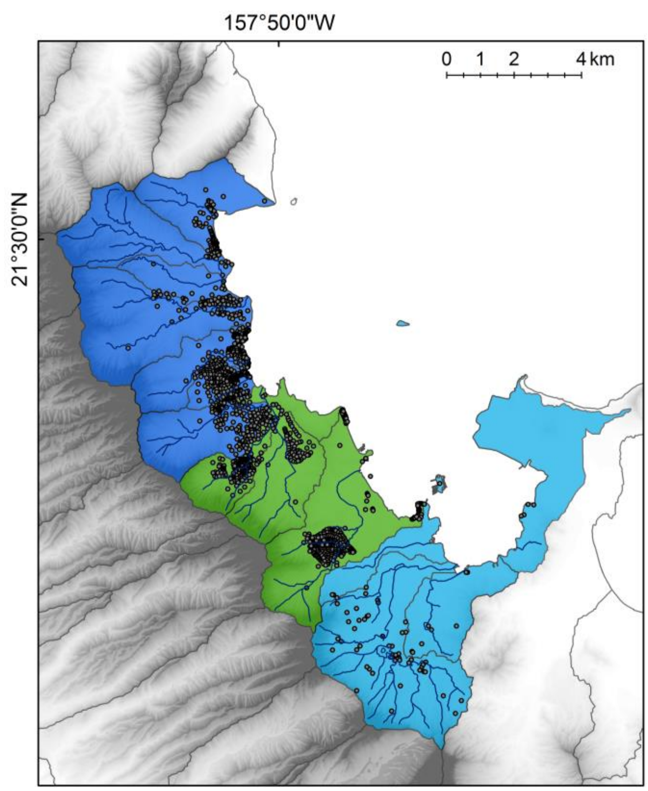

Figure 4. Sectors (Northwest, central, and south in dark blue, green, and light blue, respectively) sub-watersheds, and streams of Kāneʻohe Bay and Watershed. Individual OSDS units indicated by dots (Hawaii Office of Planning and Permitting; Whittier & El-Kadi, 2009).

sourced from past agricultural practices are still quantifiable (Hunter et al., 1995; Koʻolaupoko Watershed Management Plan, 2012). The degree to which these pollutant sources impact water quality in Kāneʻohe Bay and Watershed will likely be further exacerbated by increasing population and sea level rise (SLR), causing coastal groundwater inundation of OSDS units.

4.1. On-Site Sewage Disposal Systems

OSDS are considered a non-point pollutant source that can directly contaminate groundwater. OSDS are individual receptacles for wastewater, most commonly in Hawaiʻi in the form of septic tanks and cesspools. Not all OSDS, however, treat waste equally (Table 3), and cesspools (which only offer basic primary treatment, and can be thought of as a settling tank) are often in direct contact with the groundwater (Whittier & El-Kadi, 2009; 2014; HDOH 2015).

Table 3. Characteristics (average fecal coliform (cfu/100 mL), average nitrogen and phosphorus (mg/L) concentrations) of OSDS effluent by OSDS type (WRRC and Engineering Solutions, 2008; Whittier & El-Kadi, 2009).

Class	Type	Avg. Fecal Coliform (cfu/100 mL)	Avg. nitrogen concentration (mg/L)	Avg. phosphorus concentration (mg/L)
Class I	Soil treatment	13	1	2
Class II	Aerobic unit	1×10^6	24	8
Class III	Septic tank	1×10^6	36	13
Class IV	Cesspool	1×10^6	60.5	16.5

Cesspools, unlike the other forms of OSDS, do not treat sewage beyond removing solids from the wastewater. Cesspools are constructed with open joints along the walls to allow fluid leachate seep to the surrounding soil (Figure 5; HDOH, 2014; EPA, 2002). Solids are trapped in the cesspool and settle by gravity. Other forms of OSDS, such as septic tanks, offer primary treatment followed by some degree of secondary treatment, such as anaerobic microbial digestion of sludge (EPA, 2002). Septic systems also have a leach field, where liquid waste can undergo further treatment by soil percolation and plant uptake (EPA, 2002). Many OSDS in Hawai‘i were constructed without sufficient depth to groundwater (> 25 feet, HDOH 2015; Whittier & El-Kadi, 2009, 2014), which increases the risk of wastewater contamination to water resources. The large number of and lack of treatment associated with cesspools further increases this risk of groundwater contamination.

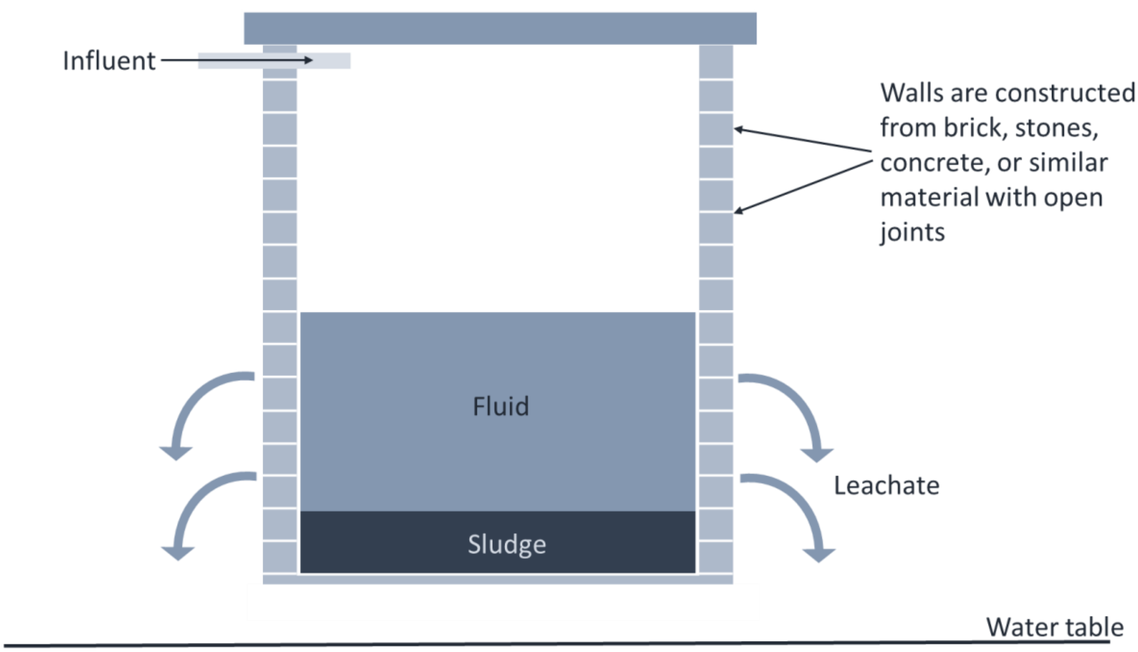

Figure 5. Schematic of a cesspool (after HDOH, 2015). Wastewater influent from a household or building served by cesspool enters the tank through a pipe. Solids settle to the bottom by gravity and accumulate as sludge. Liquids can leach through the open joints in the walls.

In fact, there are over 110,000 OSDS (88,000 of them are cesspools) in the state of Hawai‘i and nearly half of them are deemed a risk to water resources given their proximity to the coast or water resources (HDOH, 2015). Cesspools in Hawai‘i are estimated to leach about 53 million gallons per day (estimated 800,000 gallons per day in Kāne‘ohe Watershed alone) of raw sewage into the ground, in addition to substantial fluxes of nutrients, which are estimated at 23,700 and 6,000 pounds per day for nitrogen and phosphorus, respectively (Whittier & El-Kadi, 2009; 2014; HDOH; Hunter & Evans, 1995).

The quantity and density of OSDS in Kāne‘ohe Watershed increase with longitude, or increase from the southern to the northwestern sector (69, 601, and 815 units for the southern, central, and northwestern sectors, respectively) as access to sewer connectivity decreases (Whitter & El-Kadi, 2009, HDOH). Moreover, areas where OSDS density exceeds 40 units/mi^2 are deemed at high risk for groundwater contamination on the basis of average volume of effluent per household and historical occurrences of OSDS-derived wastewater contamination of drinking water wells (USEPA, 1977; Yates, 1985; USEPA, 2002; Whitter & El-Kadi, 2009; 2014). In Kāne‘ohe Watershed, OSDS density exceeds this threshold in both the central (> 40 units/mi^2 in some areas) and northwestern (> 100 units/mi^2 in some areas) sectors (Whittier & El Kadi, 2009).

Groundwater contamination from OSDS leachate is a potentially major threat to not only water quality but also drinking water resources. In addition to raw sewage, pathogens associated with wastewater, metals (from plumbing systems), and excreted pharmaceuticals, OSDS leachate is bioreactive due to high concentrations of organic matter. Leachate from OSDS travels along the groundwater gradient and undergoes biogeochemical transformations, mediated by chemoautotrophic bacteria (Figure 6).

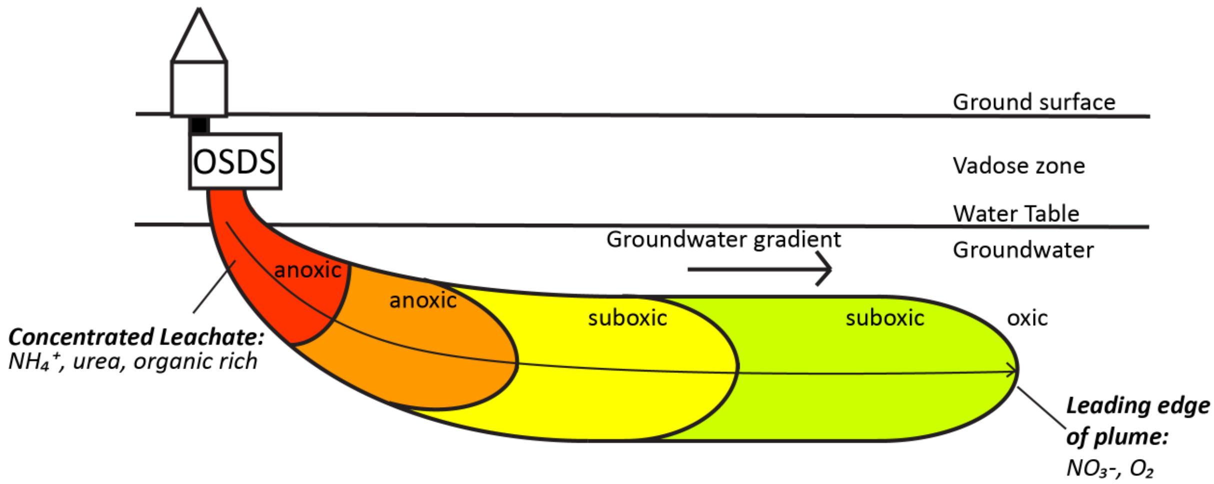

Figure 6. Idealized biogeochemical transformations and movement of an OSDS plume (Spiteri et al., 2007).

Concentrated leachate from the OSDS is rich in ammonia/ammonium (~ 70-90% of total nitrogen), dissolved organic nitrogen (DON; ~ 10-30% of total nitrogen), dissolved inorganic carbon (DIC) and organic matter, and is anoxic (Spiteri et al., 2007; Lusk et al., 2017). Dissolved organic nitrogen (DON) includes urea and free amino acids, which are transformed to ammonia/ammonium by ammonification (nitrogen mineralization) mediated by bacteria such as *Clostridium*, *Pseudomonas*, and *Bacillus* in the plume (Lusk et al., 2017). The OSDS plume becomes progressively oxidized with increasing dispersion and diffusion from contact with the relatively oxygen-rich groundwater (Spiteri et al., 2007; Lusk et al., 2017). The leading edge of the OSDS plume is rich in nitrate (through nitrification mediated by chemoautotrophic nitrifying bacteria), which can then be converted to N_2O and N_2 gases through denitrification (Kendall, et al., 2007; Lusk et al., 2017).

The high concentration of OSDS units, which are dominantly cesspools, are a major concern for groundwater, stream, and coastal water pollution in Kāneʻohe Bay and Watershed. Issues with water contamination from cesspools have already surfaced in Kahaluʻu, where

Kahaluʻu Lagoon and Beach have been closed since 2014 by the Hawaiʻi Department of Health due to high fecal indicator bacteria counts (McKenzie et al., unpublished; HDOH). The potential for OSDS-derived water contamination will only increase, as systems age and stressors from climate change (i.e. sea level rise causing groundwater inundation) decrease efficiency of OSDS performance (Habel et al., 2017; Cooper, et al., 2016).

4.2. Agriculture and erosion

While plantation-based (sugarcane and pineapple) agriculture have subsided in Kāneʻohe Watershed, the application of fertilizers for over 100 years resulted in large quantities of added nitrogen and phosphorus to the watershed. In addition, legacy contaminants, such as dieldrin and other pesticides, which do not easily degrade in the environment, still remain detectable in trace quantities in the water (Hunter et al., 1995; Koʻolaupoko Watershed Management Plan, 2012). Large-scale agriculture was dominantly based in the southern sector of Kāneʻohe Watershed, compared to smaller-scale operations in the northeastern sector. Concentrations of pesticides in oysters in Kāneʻohe Bay reflect these differences, where concentrations of dieldrin ranged from 8 to 200 ng/g in the southern sector and were below detection in the northeastern sector (Hunter et al., 1995).

Kāneʻohe Watershed has the highest erosion rate on the island of Oʻahu (DNLR, 2015). Erosion, both from natural and anthropogenic sources, offers an additional threat to Kāneʻohe Watershed and Bay water quality due to the high potential of excess sediment loading to these waters. Natural erosion generally occurs episodically, following high rainfall events (Hoover & MacKenzie, 2009). Storm events can result in substantial suspended particulate matter (SPM) and particulate nutrient fluxes to the bay (De Carlo, et al., 2007). Additionally, storm-derived stream runoff can cause dramatic increases in dissolved inorganic nitrogen (DIN) and dissolved inorganic phosphorus (DIP) concentrations (De Carlo et al., 2007; Hoover, 2002). Storm-derived DIN: DIP ratios (reported values range from 27 to greater than 40) to Kāneʻohe Bay are generally high, resulting in a P-limiting condition in a body of water that is generally considered N-limiting (Hoover, 2002; De Carlo et al., 2007). In southern Kāneʻohe Bay, excess particulate loading from streams often also results in stratification of the water column, lasting for up to a week (De Carlo et al., 2007).

4.3. Future: Climate Change and Sea Level Rise

Additionally, while increasing population and urbanization in Kāneʻohe Watershed pose an increasing risk to water quality in Kāneʻohe Bay and Watershed, the projected impacts from climate change may pose a greater threat. Anticipated climate change impacts in Hawaiʻi include increased sea level and sea water temperature, and decreased rainfall during the wet seasons (Giambelluca et al., 2013; Leta, et al., 2016). A decrease in orographic rainfall is anticipated due to a decrease in Pacific trade winds caused by increasing atmospheric temperatures (Vecchi et al., 2006; Garza et al., 2012; Giambelluca et al., 2013). Since 1860, Pacific trade wind strength has decreased by about 3.5%, and is projected to further decrease by 10% by the end of the 21st century (Vecchi et al., 2006). This decrease will particularly affect the windward upslope sides of the Hawaiian Islands which receive about 70% of yearly rainfall through orographic precipitation (Hunt, 1996). Increased sea-level by 0.6 m is projected to result in appreciable groundwater inundation, additionally fully inundating many OSDS units (Habel et al., 2017). Groundwater inundation occurs when the unsaturated space in soils becomes saturated, causing flooding. Because many OSDS units are located along the coastline and were constructed without sufficient depth to groundwater, these units are projected to become partially or fully inundated by groundwater (Whittier & El-Kadi, 2009; 2014; HDOH; Habel et al., 2017; Cooper et al., 2016).

5. Methods for quantifying water fluxes and contaminant transport

Given the current and likely future perturbations to Kāneʻohe Bay and Watershed, an understanding of how to trace and source water is important. This section will discuss various methods of tracing ground and surface waters, as well as sourcing contaminants to natural waters.

5.1. Natural tracers for groundwater

Naturally occurring radiogenic uranium daughter products, including ^{222}Rn (radon), and radium isotopes (^{223}Ra, ^{224}Ra, ^{226}Ra, ^{228}Ra), have been used successfully as tracers for groundwater in a wide array of environments. Radon (^{222}Rn) is a radiogenic ($t_{1/2}$ = 3.8 days) noble gas that is unique to groundwater. Radon is produced in the aquifer from the decay of uranium-bearing rocks and has been used successfully as a conservative tracer for groundwater inflow to surface waters (Moore et al., 1996; Burnett & Dulaiova, 2003; Charette et al., 2008; Dulaiova et al., 2010). Radium isotopes include both short (^{223}Ra, ^{224}Ra, $t_{1/2}$ = 11 and 3.5 days, respectively) and long (^{226}Ra, ^{228}Ra, $t_{1/2}$ = 1600 and 5.75 years, respectively) lived isotopes which can then be used to determine mixing processes, and SGD rates (Burnett & Dulaiova, 2003). Coupling these radiogenic tracers with stable isotopes can offer a powerful means for tracing groundwater sources and origins, estimating residence times and mixing ratios, and determining groundwater age (Charette et al., 2001; Burnett & Dulaiova, 2003; Liu et al., 2014).

Ground and surface water fluxes can be determined by implementing a radon mass balance box-model in combination with field measurements from stream gaging. Total SGD fluxes in $m^3 d^{-1}$ (which includes both fresh and saline SGD) along the coastline can be calculated using Equation 1, where $A_{Rn_{cw}}$ and $A_{Rn_{gw}}$ are the coastal and groundwater ^{222}Rn activities Bq m^{-3}, V is the volume of water represented by the box (m^3), and τ is the coastal residence time of the water.

$$QSGD_{tot} = \frac{A_{Rn_{cw}} \times V}{\tau \times A_{Rn_{gw}}} \qquad \text{Eq. 1.}$$

Groundwater fluxes in streams can be calculated using a similar box model in regular intervals downstream with Equation 2 (Cartwright & Hofmann, 2016) where $\frac{dQ}{dx}$ is the change in stream discharge per box, I is the groundwater inflow into the box ($m^3\ m^{-1}\ day^{-1}$), E accounts for evaporation ($m\ day^{-1}$), and w is the stream width (m).

$$\frac{dQ}{dx} = I - Ew \qquad \text{Eq. 2.}$$

Radon and radium isotopes are useful for determining not only groundwater fluxes, but can also be used to estimate nutrient and other dissolved constituent fluxes.

5.2. Methods for contaminant source tracking

Identification of wastewater contamination, as well as its source(s), to natural waters can be challenging. These challenges originate from not only discerning the type of source (e.g. leaky sewers, septic tanks, cesspools), but also accounting for spatiotemporal variability, groundwater flow paths, and differentiating human sewage from animal waste (Kendall et al., 2007). Modern wastewater tracers include isotopic approaches (such as $\delta^{15}N$ and $\delta^{18}O$ of nitrate), microbial source tracking (MST), and dye tracer techniques, but each of these methods have limitations. Stable isotopes of nitrate are useful as wastewater source trackers, but the results are not always conclusive due to the potential for overlap in source isotopic composition. Additionally, the isotopic values typically associated with wastewater do not differentiate between human and animal waste (Kendall et al, 2007). MST generally relies on fecal indicator bacteria (FIB) levels to determine presence of sewage, however, many FIB are associated with both human and warm-blooded animal waste and not all FIB are appropriate to use in all environments (Leeming et al., 1996; Tran et al., 2015). Dye tracer tests can directly demonstrate a flow path from its source, however it is reliant on both knowing the exact source of contamination as well as having access to that source. Dye tracer tests were successfully used at Lahaina Wastewater Reclamation Facility (LWRF), where Fluorescein dye was added to wastewater injection wells and later detected along the coastline (Glenn et al., 2012). It is necessary to use a multi-tracer approach to determine both source and transport pathway.

$\delta^{15}N$ and $\delta^{18}O$ values of nitrate are increased via microbial pathways, allowing one to recognize the source of nitrogen in the water (Figure 7). Nitrate is common in groundwater and has

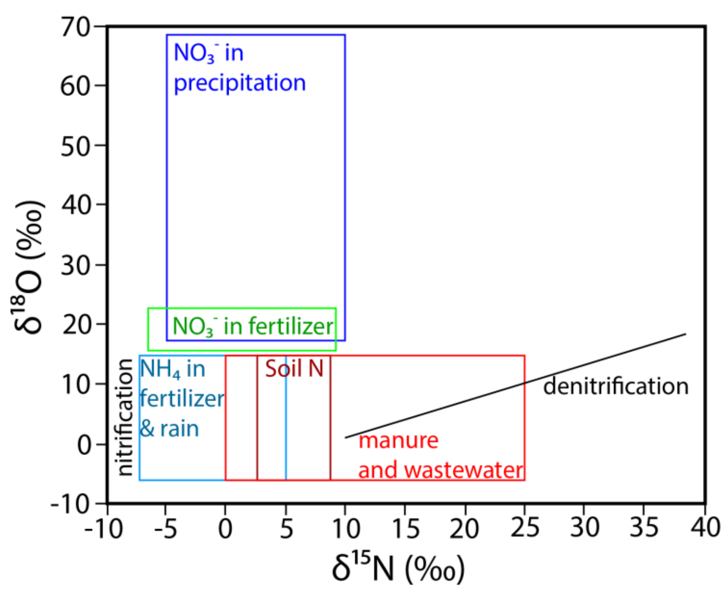

Figure 7. Typical ranges in ‰ for $\delta^{15}N$ and $\delta^{18}O$ of nitrate sources in water (after Kendall et al., 2007).

numerous natural (e.g. atmospheric deposition, weathering of organic-rich soils) and anthropogenic (e.g. sewage, artificial fertilizer, manure) sources. The isotopic composition of nitrate can be changed by isotope fractionation associated with biological or mixing processes which occur during the nitrogen biogeochemical cycle (Kendall, 1998). Major processes in the nitrogen cycle include denitrification, ammonia volatilization, anammox, nitrification, and N_2 fixation (Table 4).

Table 4. Major biogeochemical processes in the nitrogen cycle and their typical $\delta^{15}N$ values (Kendall, 1998; Granger & Wankel, 2016).

Process	Pathway	Reaction	Typical $\delta^{15}N$ values (‰)
Denitrification	$NO_3^- \rightarrow NO_2^- \rightarrow N_2O$ (g) $\rightarrow N_2$ (g)	$CH_2O + NO_3^- \rightarrow N_2$ (g) $+ H_2O + CO_2 + HCO_3^-$	-40 to +30
Ammonia Volatilization	$NH_3 \rightarrow NH_4^+ \rightarrow NO_3^-$	$CO(NH_2)_2 \rightarrow NH_3 \leftrightarrow NH_4^+ \rightarrow NO_3^-$	> 20
Anammox	$NH_4^+ \rightarrow N_2$ (g); $NO_2^- \rightarrow N_2$ (g)	$NH_4^+ + NO_3^- \rightarrow N_2$ (g) $+ H_2O + H^+$	
Nitrification	$NH_3 \rightarrow NO_2^- \rightarrow NO_3^- + NO_2^- \rightarrow NO_3^-$	$NH_4^+ + O_2 \rightarrow NO_3^- + H_2O + H^+$	-15 to +15
N_2 fixation (natural)	$N_2 \rightarrow NH_3$	$N_2 + H^+ + MgATP \rightarrow 2\ NH_3 + H_2 + MgADP$	-3 to + 1
N_2 fixation (Haber-Bosch)		$N_2 + 3\ H_2 \rightarrow 2\ NH_3$	

Kinetic isotope fractionation occurs during denitrification, a process mediated by bacteria, leaving the residual nitrate isotopically heavier (Kendall et al., 2007; Lusk et al., 2017). An isotope mass balance can be used to evaluate the contribution of sources (in the case of denitrification, these sources may include wastewater, animal manure, and soil nitrogen) to a mixed sample. Equation 3 is an example of an isotope mass balance for denitrification, where the product refers to the mixed sample of interest, and the sources refer to processes which can change the $\delta^{15}N$ value of nitrate.

$$[NO_3^-]_{product}\,\delta^{15}N_{product} = \sum[NO_3^-]_{sources}\delta^{15}N_{sources} \qquad \text{Eq. 3}$$

To parse potential denitrification in the aquifer and differentiate it from a wastewater input, one can take several approaches. One way would be evaluating the degree of isotope fractionation (Griggs et al., 2003) on the basis of Rayleigh Distillation. Another method of recognizing denitrification includes quantifying the presence and/or absence of excess N_2 gas (Kendall, 1998). This approach assumes a closed system, where both argon (which should behave conservatively in the aquifer) and N_2 gases are measured and an isotope mass balance is applied (Equation 3). If excess N_2 gas is found, then denitrification is indeed occurring in the aquifer. Yet another means of identifying denitrification would be to plot the $\delta^{15}N$ and $\delta^{18}O$ of nitrate values (such as in Figure 7) and look for a slope with a constant increasing trend indicative of denitrification and/or anammox (anticipated slope would be 1 or 0.5, depending on the process). Oftentimes, $\delta^{15}N$ and $\delta^{18}O$ values of nitrate can provide somewhat ambiguous results, which is further complicated by the potential for denitrification to occur in low oxygen conditions, and thus it is helpful and informative to combine with other indicators of water source.

Due to recent improvements in mass spectrometry resolution, contaminants of emerging concern (CECs), are increasingly used as wastewater tracers. CECs include pharmaceuticals, endocrine disrupting and industrial chemicals, pesticides, and other organic anthropogenically-sourced compounds that are environmentally pervasive due to their refractory nature and human influence on land-use (Lapworth, et al., 2012). CECs are environmentally persistent, generally in trace quantities (ng/L to µg/L), due to poor removal efficiencies in wastewater treatment plants, sewage leaks, and agricultural and industrial waste leaking into soils (Lapworth et al., 2012; Yang, et al., 2017). Pharmaceuticals may be used to trace wastewater contamination in a watershed. In a similar manner, pesticides may also be used to trace agricultural contamination. Some CECs, such as caffeine and carbamazepine, have been successfully used as novel wastewater tracers in multiple environments, including Hawaiʻi (e.g. Knee et al., 2010; Lusk et al., 2017).

While there have been a limited number of studies on pharmaceuticals conducted in Hawaiʻi, compounds including carbamazepine, sulfamethoxazole, caffeine, and estrogen have been detected downstream from wastewater treatment plants (Hunt & Rosa, 2009; Knee et al., 2010; D'Alessio et al., 2014). Analysis of caffeine in water samples in Kauai ranged from below detection to 88 ng/L (Knee et al., 2010), and were proposed for use as a potential tracer for wastewater contamination to natural waters. Estrogen compounds in treated wastewater effluent from Honoululiuli Wastewater Treatment Plant (Ewa Beach, Oʻahu, Hawaiʻi) were found in concentrations up to 10 ng/L (D'Alessio et al., 2014). In treated wastewater effluent in Kihei, Maui, carbamazepine and sulfamethoxazole were found in concentrations in coastal springs up to 102 and 24 ng/L, respectively (Hunt & Rosa, 2009).

There are several advantages and disadvantages for using pharmaceuticals as wastewater tracers. One, many have an exclusively anthropogenic (i.e. human sewage) source, owing to the fact that pharmaceuticals are designed to persevere through the body which means a sometimes substantial percentage of bioactive ingredient is excreted (Lusk et al., 2017). Pharmaceuticals also have different environmental properties and degradation pathways (Table 5), which can be seen as both an advantage and disadvantage for wastewater-tracking purposes.

Table 5. Major environmental properties and degradation pathways of CECs (Lapworth et al., 2012; Wang et al., 2017; Lusk et al., 2017; PubChem Database; ChemSpider Database).

Property/Pathway	Description	Metric(s)	Example CECs
Volatilization	Likelihood of attenuation of compound in gaseous form to the atmosphere	Henry's law constant (higher = more likely to volatilize)	Volatile organic compounds (VOCs), such as 1,4-dichlorobenzene
Sorption	Likelihood to sorb to soil matrix	Octanol-water partitioning coefficient (K_{ow}; higher log K_{ow} = more likely to sorb); pK_a water pH	Triclosan; Carbamazepine
Solubility	Likelihood to dissolve in water	Organic carbon-water partitioning coefficient (K_{oc}; lower = more soluble in later); pK_a water pH	Caffeine; Ciprofloxacin
Photolysis	Likelihood to degrade by sunlight	Primary photolytic pathway: direct (CEC absorbs photons, leading to degradation) or indirect (CEC interacts with other photosensitive compounds)	Triclosan; Sulfamethoxazole; Fluoroquinolones
Biodegradation	Likelihood to degrade by biotransformation	Temperature; dissolved oxygen concentration (CECs are more likely to degrade in aerobic conditions) ; pH; bacteria present	Caffeine

High concentrations of dissolved organic carbon (DOC) were shown to enhance the transport of estrogen compounds in treated wastewater effluent (D'Alessio et al., 2014). Moreover, because OSDS-leachate is anticipated to have high concentrations of DOC, estrogens (which would likely occur in higher concentrations) in OSDS-derived wastewater would have lower residence times, thus less potential for remediation (D'Alessio et al., 2014; Lapworth et al., 2012).

6. Conclusion

Kāne‘ohe Bay and Watershed have had a long history of ongoing environmental pollution from a variety of contaminant sources. Recognizing these historical, modern, and potentially future contaminant sources, in addition to the characteristics of the environment itself, can help inform scientific knowledge and management decisions. Currently, major pollutant sources to Kāne‘ohe Bay and Watershed stem from erosion, wastewater, and urbanization. While erosion, sediment runoff, direct sewage discharge, and the impacts of increasing populations have been well-studied in Kāne‘ohe, wastewater impact from OSDS, SGD's role in delivering pollutants, and future impacts from SLR are currently understudied. Moreover, having the ability to trace these contaminants in the environment is not straightforward and using a multi-tracer approach may lead to more accurate conclusions, thus improving water quality and quantity. The specific methods mentioned in this report for groundwater tracing are advantageous to use in environments that include fresh to saline water. Moreover, because OSDS leachate impacts groundwater, coupling groundwater tracers with wastewater tracers offers a means to trace the pathway of pollution and perhaps localize source(s) of pollution. These methods combined offer a means to evaluate water quality mauka to makai, ideally informing those who make management decisions on how best to not only remediate pollution to natural waters, but also protect natural and drinking water resources from wastewater pollution and projected impacts from SLR. Development of additional contaminant source-tracking techniques is vital to making this process not only easier, but more informative.

References

Altizer, K., Lance, J., Hammatt, H.H. (2011). Draft Archaeological Literature Review and Field Inspection for the Mahuahua Ai o Hoi: Heeia Wetland Restoration Project, Heeia Ahupuaa, Koolaupoko District, Oahu Island, TMK: [1] 4-6-016:001 & 002.

Burnett, W.C., Dulaiova, H., (2003). Estimating the dynamics of groundwater input into the coastal zone via continuous radon-222 measurements. *Journal of Environmental Radioactivity*, 69(1): 21-35

Burnett, W. C., Aggarwal, P. K., Aureli, A., Bokuniewicz, H., Cable, J. E., Charette, M. A., … Turner, J. V. (2006). Quantifying submarine groundwater discharge in the coastal zone via multiple methods. *Science of the Total Environment*, 367(2–3), 498–543. https://doi.org/10.1016/j.scitotenv.2006.05.009

Bahr, K. D., Jokiel, P. L., & Toonen, R. J. (2015). The unnatural history of Kāneʻohe Bay: coral reef resilience in the face of centuries of anthropogenic impacts. *PeerJ*, *3*, e950. https://doi.org/10.7717/peerj.950

Cartwright, I., & Hofmann, H. (2016). Using radon to understand parafluvial flows and the changing locations of groundwater inflows in the Avon River, southeast Australia. *Hydrology and Earth System Sciences*, 20(9), 3581–3600. https://doi.org/10.5194/hess-20-3581-2016

Charette, M. A., Buesseler, K. O., & Andrews, J. E. (2001). Utility of radium isotopes for evaluating the input and transport of groundwater-derived nitrogen to a Cape Cod estuary. *Limnology and Oceanography*, 46(2), 465-470. doi:10.4319/lo.2001.46.2.0465

Charette, M.A.,W.S. Moore and W.C. Burnett, (2008). Uranium- and thorium-series nuclides as tracers of submarine groundwater discharge. Radioactivity in the Environment, ed. S. Krishnaswami, J.K. Cochran, vol: "U-Th Series Nuclides in Aquatic Systems", 13, 155-191.

Chemspider Database. *Royal Society of Chemistry.*

Cooper, J. A., Loomis, G. W., & Amador, J. A. (2016). Hell and high water: Diminished septic system performance in coastal regions due to climate change. *PLoS ONE*, 11(9), 1–18. https://doi.org/10.1371/journal.pone.0162104

Dailer, M.L., Ramey, H.L., Saephan S., Smith, C.M. (2010). Using δ15N values in algal tissue to map locations and potential sources of anthropogenic nutrient inputs on the island of Maui, Hawaii, USA. *Marine Pollution Bulletin* 60: 655-671.

D'Alessio, M., Vasudevan, D., Lichwa, J., Mohanty, S. K., & Ray, C. (2014). Fate and transport of selected estrogen compounds in Hawaii soils: Effect of soil type and macropores. *Journal of Contaminant Hydrology*, 166, 1–10. http://doi.org/10.1016/j.jconhyd.2014.07.006

De Carlo, E. H., Hoover, D. J., Young, C. W., Hoover, R. S., & Mackenzie, F. T. (2007). Impact of storm runoff from tropical watersheds on coastal water quality and productivity. *Applied Geochemistry*, 22(8 SPEC. ISS.), 1777–1797. https://doi.org/10.1016/j.apgeochem.2007.03.034

Devaney, D.M., Kelly, M., Lee, P.J., Motteler, L.S. (1982). Kaneohe: a history of change (1778–1950). Honolulu: Bishop Museum Press

Dulai, H., Kleven, A., Ruttenberg, K., Briggs, R., Thomas, F. (2016). Evaluation of submarine groundwater discharge as a coastal nutrient source and its role in coastal groundwater quality and quantity. Chapter 8 In: Emerging Issues in Groundwater Resources, Part of

the series Advances in Water Security, 187-221, Ali Fares, ed. pp. DO 10.1007/978-3-319-3008-3; ISBN 978-3-319-32006-9.
Dulaiova, H., Camilli, R., P. B. Henderson, and M. A. Charette, (2010). Coupled Radon, Methane And Nitrate Sensors For Large-Scale Assessment Of Groundwater Discharge And Non-Point Source Pollution To Coastal Waters, *Journal of Environmental Radioactivity*, 101(7), 553-563,.doi: 10.1016/j.jenvrad.2009.12.004.
Garza, J.A., Chu, P., Norton, C.W., Schroeder, T.A., (2012). Changes of the prevailing trade winds over the islands of Hawaii and the North Pacific. *Journal of Geophysical Research*, 117(11), 1-18. doi:10.1029/2011JD016888,
Giambelluca, T. W., Chen, Q., Frazier, A. G., Price, J. P., Chen, Y. L., Chu, P. S., … Delparte, D. M. (2013). Online rainfall atlas of Hawai'i. *Bulletin of the American Meteorological Society*, 94(3), 313–316. https://doi.org/10.1175/BAMS-D-11-00228.1
Glenn, C.R., Whittier, R.B., Dailer, M.L., Dulaiova, H., El-Kadi, A.I., Fackrell, J., Kelly, J.L.. Waters, C.A. and Sevadjian, J., (2012). Lahaina Groundwater Tracer Study—Lahaina, Maui, Hawai'i, FINAL REPORT. 502p.
Granger, J., & Wankel, S. D. (2016). Isotopic overprinting of nitrification on denitrification as a ubiquitous and unifying feature of environmental nitrogen cycling. *Proceedings of the National Academy of Sciences*, 113(42). doi:10.1073/pnas.160138311
Griggs, E. M., Kump, L. R., & Böhlke, J. (2003). The fate of wastewater-derived nitrate in the subsurface of the Florida Keys: Key Colony Beach, Florida. *Estuarine, Coastal and Shelf Science*, 58(3), 517-539. doi:10.1016/s0272-7714(03)00131-8
Habel, S., Fletcher, C. H., Rotzoll, K., & El-Kadi, A. I. (2017). Development of a model to simulate groundwater inundation induced by sea-level rise and high tides in Honolulu, Hawaii. *Water Research,114*, 122-134. doi:10.1016/j.watres.2017.02.035
Hawaii Department of Health (HDOH), (2012). Hawaii Administrative Rules (HAR). Chapter 11-55. Water Pollution Control. *Hawaii Department of Health - Clean Water Branch*
Hawaii Department of Health (HDOH), (2014). Hawaii Administrative Rules (HAR), Chapter 11-54, Water Quality Standards. *Hawaii Department of Health - Clean Water Branch*
Hawaii Department of Health (HDOH). (2015). Hawaii non-point source management plan: 2015 to 2020. *Hawaii Department of Health - Clean Water Branch*. Retrieved from: http://health.hawaii.gov/cwb/files/2013/05/2015-Hawaii-NPS-Management-Plan.pdf
Hawaii Office of Planning and Permitting: GIS Database
Hawaii Soil Atlas (2014). CTAHR, University of Hawaii at Manoa. Retrieved from http://gis.ctahr.hawaii.edu/SoilAtlas
Hédouin L., Reichelt-Brushett, A.J., Gates, R.D. (2009). Assessment of metals and a metalloid in sediments from Hawaii coral reef ecosystems. *Marine Pollution Bulletin*, 58, 1739-1765. https://doi.org/10.1016/jmarpolbul.2009.07.021
Hoover D.J. (2002). Fluvial nitrogen and phosphorus inputs to Hawaiian coastal waters: storm loading, particle-solution transformations and ecosystem impacts. *Unpublished Ph.D. dissertation*, Department of Oceanography, University of Hawaii
Hoover, D. J., & MacKenzie, F. T. (2009). Fluvial fluxes of water, suspended particulate matter, and nutrients and potential impacts on tropical coastal water Biogeochemistry: Oahu, Hawai'i. *Aquatic Geochemistry*, 15(4), 547–570. https://doi.org/10.1007/s10498-009-9067-2
Hunt, C. D. (1996). Geohydrology of the Island of Oahu, Hawaii. *U.S. Geological Survey Professional Paper* 1412-B.

Hunt, C.D., Rosa, S.N. (2009). A multitracer approach to detecting wastewater plumes from municipal injection wells in nearshore marine waters at Kihei and Lahaina, Maui, Hawaii: *US Geological Survey Scientific Investigations Report* 2009-2553. 166 pp.

Hunter, C. L., & Evans, C. W. (1995). Coral Reefs in Kaneohe Bay, Hawaii: Two Centuries of Western Influence and Two Decades of Data. *Bulletin of Marine Science*, 57(2).

Hunter, C. L., Stephenson, M. D., Tjeerdema, R. S., Crosby, D. G., Ichikawa, G. S., Goetzl, J. D., … Newman, J. W. (1995). Contaminants in oysters in Kaneohe Bay, Hawaii. *Marine Pollution Bulletin*, 30(10), 646–654. https://doi.org/10.1016/0025-326X(95)00039-P

Jokiel, P. L. (2004). Illustrated Scientific Guide To Kane'Ohe Bay, Oahu. Hawaii Institute of Marine Biology, (January 1991), 1–65. https://doi.org/10.13140/2.1.3051.9360

Kendall, C. (1998). Isotope Tracers in Catchment Hydrology. Elsevier Science B.V., Amsterdam.

Kendall, C., Elliott, E. M., & Wankel, S. D. (2007). Tracing anthropogenic inputs of nitrogen to ecosystems, 375–449.

Kittinger, J. N., Pandolfi, J. M., Blodgett, J. H., Hunt, T. L., Jiang, H., Maly, K., … Wilcox, B. A. (2011). Historical reconstruction reveals recovery in Hawaiian coral reefs. PLoS ONE, 6(10). https://doi.org/10.1371/journal.pone.0025460

Koolaupoku Watershed Management Plan. (2012). Prepared for the Honolulu Board of Water Supply by Townscape Inc.

Knee, K. L., Gossett, R., Boehm, A. B., & Paytan, A. (2010). Caffeine and agricultural pesticide concentrations in surface water and groundwater on the north shore of Kauai (Hawaii, USA). *Marine Pollution Bulletin, 60*(8), 1376-1382. doi:10.1016/j.marpolbul.2010.04.019

Knee, K. L., & Paytan, A. (2012). Submarine Groundwater Discharge: A Source of Nutrients, Metals, and Pollutants to the Coastal Ocean. *Treatise on Estuarine and Coastal Science* (Vol. 4). Elsevier Inc. https://doi.org/10.1016/B978-0-12-374711-2.00410-1

Kroeger, K. D., Swarzenski, P. W., Greenwood, W. J., & Reich, C. (2007). Submarine groundwater discharge to Tampa Bay: Nutrient fluxes and biogeochemistry of the coastal aquifer. *Marine Chemistry*, 104(1–2), 85–97. https://doi.org/10.1016/j.marchem.2006.10.012

Kwon, E. Y., Kim, G., Primeau, F., Moore, W. S., Cho, H. M., Devries, T., … Cho, Y. K. (2014). Global estimate of submarine groundwater discharge based on an observationally constrained radium isotope model. *Geophysical Research Letters*, 41(23), 8438–8444. https://doi.org/10.1002/2014GL061574

Lapworth, D. J., Baran, N., Stuart, M. E., & Ward, R. S. (2012). Emerging organic contaminants in groundwater: A review of sources, fate and occurrence. *Environmental Pollution*, 163, 287–303. https://doi.org/10.1016/j.envpol.2011.12.034

Lau, L. S., & Mink, J. F. (2006). *Hydrology of the Hawaiian Islands*. University of Hawaii Press.

Laws, E. A., & Redalje, D. G. (1979). Effect of seawage enrichment on the phytoplankton population of a subtropical estuary. *Pac Sci*, 33(2), 129–144.

Leeming, R.; Ball, A.; Ashbolt, N.; Nichols, P. (1996). Using faecal sterols from humans and animals to distinguish faecal pollution in receiving waters. Water Res. 30, 2893–2900

Leta, O. T., El-Kadi, A. I., Dulai, H., & Ghazal, K. A. (2016). Assessment of climate change impacts on water balance components of Heeia watershed in Hawaii. *Journal of Hydrology: Regional Studies*, 8, 182–197. https://doi.org/10.1016/j.ejrh.2016.09.006

Liu, Q., Charette, M.A., Henderson, P.B., McCorkle, D.C., Martin, W., Dai, M. (2014). Effect of submarine groundwater discharge on the coastal ocean inorganic carbon cycle. *Limnol. Oceanogr.* 59(5), 1529-1554

Lusk, M. G., Toor, G. S., Yang, Y. Y., Mechtensimer, S., De, M., & Obreza, T. A. (2017). A review of the fate and transport of nitrogen, phosphorus, pathogens, and trace organic chemicals in septic systems. *Critical Reviews in Environmental Science and Technology*, 47(7), 455–541. https://doi.org/10.1080/10643389.2017.1327787

Moore, W. S. (1996). Large groundwater inputs to coastal waters revealed by 226Ra enrichments. *Nature* 380: 612-614.

Moore, W. S. (1999). The subterranean estuary: A reaction zone of ground water and sea water. Marine Chemistry, 65(1–2), 111–125. https://doi.org/10.1016/S0304-4203 (99)00014-6

PubChem Database. *National Center for Biotechnology Information.*

Safeeq, M., Mair, A., & Fares, A. (2012). Temporal and spatial trends in air temperature on the Island of Oahu, Hawaii. *International Journal of Climatology,33*(13), 2816-2835. doi:10.1002/joc.3629

Shade, P. J., and W. D. Nichols. (1996). Water Budget and the Effects of Land-Use Changes on Ground- Water Recharge, Oahu, Hawaii. USGS Professional Paper 1412-C.

Sherrod, D.R., Sinton, J.M., Watkins, S.E., Brunt, K.M. (2007). Geologic Map of the State of Hawaii. *United States Geological Survey*, Open-File Report 2007-1089.

Smith, S. V, Kimmerer, W. J., Laws, E. A., Brock, R. E., & Walsh, T. W. (1981). Kaneohe Bay sewage diversion experiment: perspectives on ecosystem responses to nutritional perturbation. Pacific Science, 35(4), 279–395.

Spiteri, C., Slomp, C. P., Regnier, P., Meile, C., & Cappellen, P. V. (2007). Modelling the geochemical fate and transport of wastewater-derived phosphorus in contrasting groundwater systems. *Journal of Contaminant Hydrology*, 92(1-2), 87-108. doi:10.1016/j.jconhyd.2007.01.002

Stearns, H. T., Vaksvik, K.N. (1935). Geology and ground-water resources of the island of Oahu, Hawaii. *Hawaii Division of Hydrography Bulletin 1*, 479 pp.

Takasaki, K. J., & Mink, J. F. (1985). Evaluation of major dike-impounded ground-water reservoirs, Island of Oahu. U.S. Geological Survey Water Supply Paper 2217, 77 p.

Tran, N. H., Gin, K. Y., & Ngo, H. H. (2015). Fecal pollution source tracking toolbox for identification, evaluation and characterization of fecal contamination in receiving urban surface waters and groundwater. *Science of the Total Environment*, 538, 38-57. doi:10.1016/j.scitotenv.2015.07.155

United States Environmental Protection Agency (USEPA). (1977). The Report to Congress: Waste Disposal Practices and Their Effects on Ground Water. EPA/570/9-77/001.

United States Environmental Protection Agency (USEPA). (2002). Onsite Wastewater Treatment Systems Manual. EPA/628/R-00/008.

Valiela, I., Mcclelland, J., Hauxwell, J., Behr, P. J., Hersh, D., & Foreman, K. (1997). Macroalgal blooms in shallow estuaries: Controls and ecophysiological and ecosystem consequences. *Limnology and Oceanography, 42* (5part2), 1105-1118. doi:10.4319/lo.1997.42.5_part_2.1105

Vecchi, G.A., Soden, B.J., Wittenberg, A.T., Held, I. M., Leetmaa, A., Harrison, M.J., (2006). Weakening of tropical Pacific atmospheric circulation due to anthropogenic forcing. *Nature*, 441(1). 73-76. doi:10.1038/nature04744

Wentworth, C. K. (1951). Geology and ground-water resources of the Honolulu-Pearl Harbor area, Oahu, Hawaii. Honolulu: *Board of Water Supply*, City and County of Honolulu.

Whittier, R.B. & El-Kadi, A.I., (2009). Human and Environmental Risk Ranking of On-site Sewage Disposal Systems. Final. Prepared for the Hawaii Dept. of Health by the University of Hawaii at Manoa. Department of Geology and Geophysics.

Whittier, R.B. & El-Kadi, A.I., (2014). Human and Environmental Risk Ranking of On-site Sewage Disposal Systems for the Hawaiian islands of Kauai, Molokai, Maui, and Hawaii. Prepared for the Hawaii Dept. of Health by the University of Hawaii at Manoa. Department of Geology and Geophysics.

Winter, T. C., Harvey, J. W., & Franke, O. L., Alley, W.M. (1998). Ground water and surface water: A single resource. *USGS Publications*, 79. https://doi.org/10.3389/fpsyg.2012.00044

WRRC and Engineering Solutions, Inc. (2008). Onsite Wastewater Treatment Survey and Assessment. Prepared for the State of Hawaii, Department of Business, Economic Development and Tourism Office of Planning, Hawaii Coastal Zone Management Program; and the Department of Health.

Yang, Y., Ok, Y. S., Kim, K.-H., Kwon, E. E., & Tsang, Y. F. (2017). Occurrences and removal of pharmaceuticals and personal care products (PPCPs) in drinking water and water/sewage treatment plants: A review. Science of the Total Environment, 596–597, 303–320. https://doi.org/10.1016/j.scitotenv.2017.04.102

Yates, M.V. (1985). Septic Tank Density and Groundwater Contamination. Groundwater. 23(5). pp. 586-591

Zekster, I.S. (2000). Groundwater and the Environment: Applications for the global community. Lewis Publishers, Boca Raton, pp. 175