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Developing nitrogen bioextraction economic value via off-bottom oyster aquaculture in the northwestern Gulf of Mexico

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ABSTRACT

Eutrophication remains a persistent water quality issue throughout much of the United States, leading to changes to ecosystem health in valuable coastal habitats. Oysters help to buffer against eutrophication by removing nitrogen from the water column by feeding on phytoplankton and other seston, a process referred to as "bioextraction". Recent legislation in Texas has allowed oysters to be grown off-bottom (suspended in cages). To understand the connections between bioextraction and off-bottom oyster aquaculture, the Assessment of Estuarine Trophic Status (ASSETS) model was applied, indicating nutrient-related degradation of water quality. The Farm Aquaculture Resource Model (FARM) was used to determine that a typical oyster farm can remove about 4900–7100 lb. N yr^{-1}, with an approximate value of $41,966 to $232,511 based on engineered (wastewater treatment plant) technologies. A promising and innovative nutrient management strategy, bivalve mariculture can be utilized as an additional strategy complementary to existing nutrient management strategies.

1. Introduction

Eutrophication is defined as the "increase in the supply of organic matter to an ecosystem" (Nixon, 2009) and is typically associated with nutrient over-enrichment to a waterbody. The acceleration of organic matter production can occur naturally (e.g., transported from an inland watershed, tidal inflow of offshore production, etc.) or through anthropogenic activities (e.g., industrial activities or from fertilizers in agricultural runoff) (de Jonge et al., 2002). In marine systems where nitrogen is the nutrient typically limiting primary production (Malone et al., 1996), over-enrichment can determine the abundance and composition of primary producers (Ryther, 1954; Glibert, 2017). These shifts can propagate into other effects such as excessive algal blooms that lead to hypoxia, displace organisms, decrease abundance or cause die-off of submerged vascular plants, disrupt water clarity, and impact fisheries (Nixon, 1995, Burkholder et al., 2007; Thronson and Quigg, 2008; Nixon, 2009; Breitburg et al., 2009).

Approaches to nutrient management within the Clean Water Act (1972) in the United States have sought reductions in nutrient pollution to maintain ecosystem health (e.g., National Pollutant Discharge Elimination System, Total Maximum Daily Load, and Effluent Limitation Guidelines) (EPA, 2023). Despite nutrient reductions from these management strategies, eutrophication remains an issue within the nation's waters. Nationwide assessments, such as the National Estuarine Eutrophication Assessment (NEEA), found that >40 % of the total U.S. estuarine area studied exhibited high expressions of eutrophication and that 84 of 138 estuaries had moderate to high eutrophication status (Bricker, 1999). Similarly, the Environmental Protection Agency's (EPA) National Coastal Condition Assessment (NCCA), found that eutrophication is the most widespread problem in U.S. estuaries, with >60 % rated as "fair" to "poor" (EPA, 2021). High eutrophic conditions were found in the Gulf of Mexico with both the NEEA and NCCA (Bricker et al., 1999; EPA, 2021).

Supplementation of land-based nutrient remediation with that of bivalves has been proposed in the U.S., Europe, Japan, Australia, and New Zealand (Carmichael et al., 2012; Rose et al., 2015a,b). The recent passing of Texas House Bill 1300 and Senate Bill 682 in 2019 has

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enabled off-bottom cultivated oyster aquaculture within Texas waters, allowing for new forms of nutrient management within the state. Oysters naturally remove nitrogen by feeding on phytoplankton and other seston suspended in the water column, thereby removing nutrients from the water via sequestration of nutrients into tissue and shell, a process called 'bioextraction' (Lindahl et al., 2005; Bricker et al., 2018). In some U.S. states (e.g., Massachusetts), nutrient removal by oyster and clam cultivation has been approved for crediting and use within comprehensive coastal management plans for nutrient removal (Town of Mashpee, 2015; Reitsma et al., 2017). Within the U.S. Chesapeake Bay, a regional approach for crediting nutrient removal by harvest of oyster tissue from aquaculture was approved in 2016 (Cornwell et al., 2016). These same nutrient management mechanisms can be applied to Texas's growing oyster aquaculture industry.

The amount and value of nitrogen removal by oysters has been studied within the Gulf of Mexico and other jurisdictions in the North and Eastern U.S. (Pollack et al., 2013; Bricker et al., 2015; Parker and Bricker, 2020; Lai et al., 2020). In Texas, evaluations of nitrogen removal within the 540 km^{-2} Mission-Aransas Estuary estimated a total annual removal of 35,315 kg N from physical transport (i.e., harvest; 21,665 kg N), denitrification of biodeposits (9100 kg N), and burial of biodeposits into sediments (4550 kg N) (Pollack et al., 2013). In 1070 km^{-2} Mobile Bay, nitrogen removal services totaling 34,911 $\pm$ 5032 kg N yr^{-1} (mean $\pm$ 1sd) were estimated as the combination of calculated harvest (1769 $\pm$ 876 kg N), denitrification (22,095 $\pm$ 3305 kg N), and burial (11,047 $\pm$ 1652 kg N) (Lai et al., 2020). Using a replacement cost method of engineered solutions from wastewater treatment plants, the total value of nitrogen regulation and removal provided by these oysters were estimated at \$293,993 yr^{-1} in the Mission-Aransas Estuary (Pollack et al., 2013) and \$76,455 $\pm$ \$11,020 yr^{-1} in Mobile Bay (Lai et al., 2020). This builds upon previous research, estimating the nitrogen content and value of oysters grown in gear suspended in the water column to support the development of nutrient credit trading in the Gulf of Mexico. As suggested by previous work in Texas and other locations that support oyster populations, nutrient credit trading can be a viable option in Texas, as bivalve aquaculture is already of interest to the state (HB 1300 and SB 682) and at the federal level (NOAA National Shellfish Initiative) and could be used to supplement nutrient management on the Texas coast.

Three objectives are used in this study: 1) determine the eutrophication status in Copano Bay, TX, using the Assessment of Estuarine Trophic Status (ASSETS), 2) use the Farm Aquaculture Resource Management (FARM) model to estimate nitrogen removal via physical transport in a typical farm, and 3) Use the avoided or replacement costs approach calculate the monetary value of nitrogen removed by oysters cultivated in off-bottom configurations by comparing the cost equivalency of the same removal of nitrogen by wastewater treatment plants (WWTP).

2. Materials and methods

2.1. Study site and nitrogen loading

Copano Bay is a 180 km^{-2} secondary bay system and estuary located within the Mission-Aransas National Estuarine Research Reserve (NERR), the eastern portion of which connects to Aransas Bay. Copano Bay has an average depth of 2 m, with high winds and low stratification throughout the water column (Mooney and McClelland, 2012; Bruesewitz et al., 2013; Spalt et al., 2020). Salinity is highly variable (10–25 in wet years, 30–45 in dry years, A. Ramos, 2016) and is influenced by variable freshwater inflow from the Mission and Aransas Rivers (Pollack et al., 2012). The two primary sources of freshwater input are present in the bay are the Mission and Aransas Rivers, which provide approximately 29 % of annual gaged inflow (Schoenbaechler et al., 2011). Freshwater inputs into Copano Bay have been found to have symptoms of eutrophication, such as high chlorophyll (Mission Bay, Aransas River, Chilitipin Creek, and Port Bay) as well as low dissolved oxygen (Aransas River) and nitrate (Aransas River) (TCEQ, 2020).

The surrounding terrestrial landscape contributing to the Copano Bay watershed consists primarily of scrubland (31.57 %), pasture/hay (24.49 %), and cultivated crops (22.74 %) (Wagner and Moench, 2009). Copano Bay is in Zone 32 of the Texas Department of State Health Service's Harvesting Classification Areas, where oyster reefs occur naturally (Texas Department of State and Health Services, 2024). Areas immediately adjacent to surface freshwater inflow are generally not open for oyster harvesting. Copano Bay encompasses oyster reefs relied on by local fishers who contribute to Texas's \$21.9 million commercial oyster fishery annually (2012–2022) (National Marine Fisheries Service, 2023). Oysters have been found to form reefs perpendicular to the shoreline (Spalt et al., 2018; Legare and Mace, 2017).

The U.S. Geological Survey's Spatially Referenced Regression on Watershed (SPARROW) Southwest 2012 model integrates monitoring data with landscape information to estimate contaminant transport from inland watersheds to larger bodies of water (Preston et al., 2009). While the SPARROW model estimates total phosphorus (T.P.), total nitrogen (T.N.), suspended sediment, and streamflow for U.S. waterbodies, here we focus on nitrogen because it is commonly considered the most limiting resource for primary production in Texas bays and other estuarine systems (Conley and Malone, 1992, Gardner et al., 2006). The total nitrogen watershed loading to Copano Bay is 1,009,172 kg N yr^{-1}. The SPARROW model nitrogen load estimates for Copano are a combination of inputs from the three watersheds that comprise the area directly surrounding Copano Bay (Fig. 1). The leading sources of nitrogen across the Hydrologic Unit Codes (HUC) surrounding Copano Bay are farm fertilizer (63.8 %), municipal wastewater treatment (18.6 %), and atmospheric deposition (10.3 %), presenting nutrient management challenges for nonpoint sources (Fig. 2).

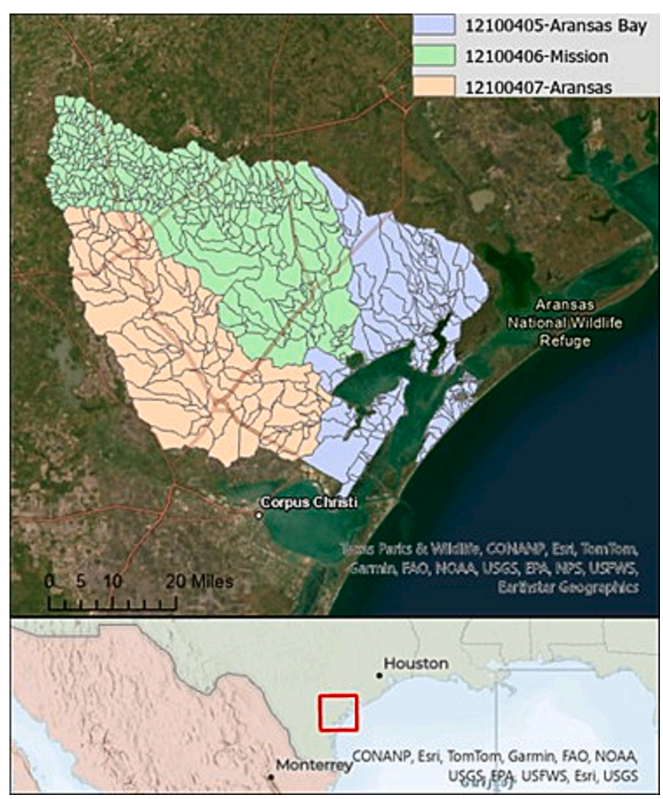

Fig. 1. USGS SPARROW shapefiles surrounding Copano Bay, TX.

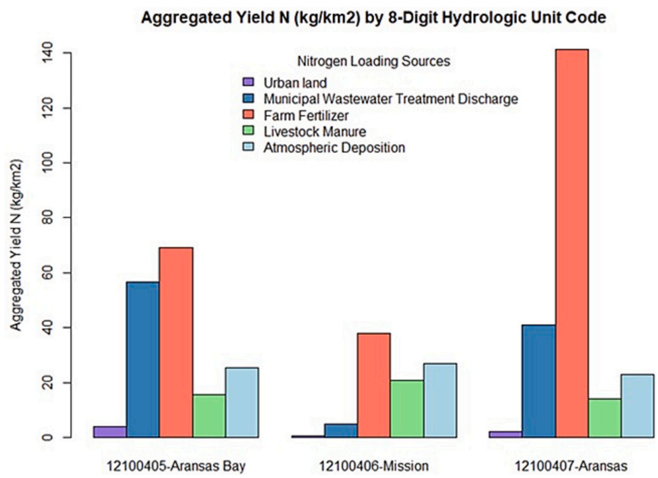

Fig. 2. USGS SPARROW Nitrogen loading sources of the three Hydrologic Unit Codes (HUC) surrounding Copano Bay indicate inland watershed nitrogen input into Copano Bay.

2.2. Assessment of Estuarine Trophic Status (ASSETS) model

ASSETS is a eutrophication screening tool used to understand the susceptibility to, status of, and potential future change in eutrophication of a waterbody (Bricker et al., 2003), with widespread use in the U.S. (Bricker et al., 1999, 2003, 2008, 2014; Whitall et al., 2007), Portugal (Ferreira et al., 2003), and China (Xiao et al., 2007), among many other locations. Primary symptoms of eutrophication in ASSETS include chlorophyll *a* and macroalgae, while secondary symptoms -indicating more serious degradation - include dissolved oxygen, submerged aquatic vegetation loss, and the occurrence of harmful algal blooms (Bricker et al., 1999; Ferreira et al., 2007a,b; Whitall et al., 2007). The indicators used in ASSETS are based on the extreme concentrations observed over an annual cycle, the spatial coverage of worst-case conditions, and the frequency of occurrence of the worst-case conditions. The macroalgae and nuisance and toxic bloom indicators are heuristically determined and include whether they are detrimental to any biological resource, the duration of a bloom, and the frequency of occurrence of blooms. The seagrass indicator considers the area of seagrass and whether seagrass area has been lost (Bricker et al., 1999; Borja et al., 2008).

The ASSETS method was selected for use in this study due to more robust sample timeframes capturing more frequent variability (e.g., as illustrated with percentiles approach of chlorophyll *a* and oxygen indicators), as well as the inclusion of additional eutrophication index parameters (macroalgae, SAVs, toxic blooms), than some other eutrophication assessment methods. While the Mission-Aransas NEER system encompasses Mission Bay, Aransas Bay, Copano Bay, and Mesquite Bay, only data from Copano Bay were utilized in the ASSETS model. Data for 2010–2021, covering many annual cycles and seasonal variations from the two Copano Bay water quality monitoring stations, were used to calculate an eastern and western percentile 90th concentration for chlorophyll *a*nd 10th percentile concentration for dissolved oxygen (Table 1). The two stations were used to represent conditions across Copano Bay. The values for chlorophyll *a* and dissolved oxygen were averaged from both stations to represent extreme conditions in Copano Bay: highest chlorophyll *a* (90th percentile of 12.056 μg l^{-1}) and lowest oxygen (10th percentile of data being 5.65 mg l^{-1}) during that period. Freshwater inflow was calculated using annual averages from the USGS Water Dashboard, combining the top three freshwater sources, Copano Creek and the Mission and Aransas Rivers, a total of 47.26 m^{-3}/s, with summer rains typically contributing the most to riverine freshwater inflow throughout an annual cycle.

Table 1
ASSETS data requirements and sources.

Data category	Data sources
System Information	National Estuarine Research Reserve Tunnell et al., 2010
Nitrogen Loading	USGS Sparrow Model (Wise et al., 2019) USGS Dashboard (USGS, 2023) National Estuarine Research Reserve System
Susceptibility and Hydrology	USGS Water Dashboard (USGS, 2023) National Estuarine Research Reserve System
Eutrophic Conditions	National Estuarine Research Reserve System Mooney and McClelland, 2012 Personal Communication (NERRS Staff, 2021)

2.3. Farm Aquaculture Resource Management (FARM) modeling

The Farm Aquaculture Resource Management (FARM) model uses biogeochemical, shellfish growth models, and eutrophication screening tools to estimate bivalve growth and eutrophication assessment (Ferreira et al., 2007a,b). The FARM model estimates nitrogen removal by mass balance of the intake of food, assimilation of part of the digested food for growth, and release of the remainder back to the water as ammonia, pseudofeces, or feces (Ferreira et al., 2007a,b). The FARM model has been used to assess the nitrogen removal rates of bivalves in the U.S., China, Chile, and Europe (Rose et al., 2015a,b). These rates are site and species-specific, with estimates of 105–1356 lb. of N $acre^{-1}$ yr^{-1} (mean of 520 lb. of N $acre^{-1}$ yr^{-1}) based upon nitrogen assimilation into shell and tissue (Rose et al., 2015a,b). FARM was chosen to estimate nitrogen removal by oysters via sequestration into tissue and shell for this research study due to the availability of a calibrated version of the model for Copano Bay (Pollack et al., 2012; Fox, 2022; NCCOS, 2023). Other models, such as ShellGIS and ShellSim, have had fewer comparable studies to utilize when considering the nitrogen removal aspects of shellfish modeling (Hawkins et al., 2013; Newell et al., 2013). Additionally, ShellGIS and ShellSim also require more data inputs in the form of bathymetry, elemental ratios, aerial exposure, and multiple set mortality configurations that require additional data in comparison to FARM (Newell et al., 2013; Hawkins et al., 2013; Ferreira et al., 2007a,b).

Data requirements for FARM are divided into; 1) time series of environmental data representing the conditions at the cultivation site such as temperature, salinity, dissolved inorganic nitrogen (DIN), total suspended solids (TSS), particulate organic matter (POM), Chlorophyll *a*, current speed and direction, etc., 2) farm dimensions, and 3) cultivation practices (Ferreira et al., 2007a,b; Ferreira et al., 2009). This study used bimonthly data collected from a 76-cage adjustable long-line system in Copano Bay, Texas from June 2020 to November 2021. The model farm was stocked with 80 adult oysters per cage (typically 7.6 cm in shell height) for a total of 6080 oysters, roughly 100 m offshore at a depth of 1–1.5 m. Water samples were obtained approximately 10 ft upstream and downstream of the farm site and 1 ft below the water's surface. These samples (raw and filtered bay water) were frozen for subsequent analysis of various chemical and physical properties: dissolved inorganic nitrogen (DIN, ammonia, nitrate, nitrite), total suspended solids (TSS), and chlorophyll *a* (Table 2). A YSI Pro DSS data sonde (Yellow Spring Instruments, Yellow Springs, OH) was used to get instant temperature, salinity, and dissolved oxygen results from a point at the center of the farm. Two Tilt Current Meter Model 1 (TCM-1, Lowell Instruments LLC, East Falmouth, MA) were used to determine water velocity during neap and spring tide.

Culture practice inputs for FARM modeling were compiled from publicly available data submitted to TPWD during the leasing process from perspective oyster farmers. Two years of data representing different salinity conditions were used to establish a range of minimum and maximum expected possibilities for oyster growth. Salinity is known to be a major driver of oyster growth, with optimal salinity generally between 19 and 24 PSU in the Gulf of Mexico (Wang et al., 2008, Lowe et al., 2017). In 2010, salinity was typically around 5–10 PSU, briefly

Table 2
Methods used for FARM water quality parameters.

Parameter	Method	Source
DIN (Nitrate, Nitrite, Ammonium)	EPA 353.2	Woods Hole Oceanographic Institution
		University of Maryland Center for Environmental Science
TSS	PA Method 160.2 and Standard Methods 208 E	Texas A&M University-Corpus Christi Center for Coastal Studies
		University of Maryland Center for Environmental Science
Chlorophyll	Fluorometric EPA 445.0, SM10200H.3; Spectrophotometric EPA 446.0, SM10200H.2	Texas A&M University-Corpus Christi Center for Coastal Studies
		University of Maryland Center for Environmental Science

reaching 0–2 PSU in three separate events through the year. In 2014, salinity remained at 35–40 PSU. Both FARM scenarios used the same farming practice for compatibility (i.e., 25 % mortality over the cultivation cycle, a leased area of 7 acres, a stocking density of 250 diploid oysters m^{-2}, 240-day cultivation cycle, 1.5 g seed weight, 70 g as minimum harvest weight (a weight corresponding roughly to 2.5 in. required for harvest), and first seeding day of April 30th (Julian day 120) for comparability.

2.4. Replacement cost valuation modeling

There are generally three forms of monetary valuation when assigning value to nutrient removal ecosystem services: 1) replacement/avoided cost for the mitigation option or equivalent, 2) observe payments made using nutrient offset credit trading programs that reflect public preferences, and 3) ask for willingness to pay for a given nutrient reduction (Shabman and Batie, 1978; Barrett et al., 2022). The replacement/avoided cost is perhaps the most common within shellfish nitrogen bioextraction ecosystem service literature (Pollack et al., 2013; Bricker et al., 2019; Grabowski et al., 2012; Lai et al., 2020). Although replacement/avoided costs are typically seen as closely associated in the literature, this research will use the term “replacement cost” as it is most closely associated with assigning monetary values to ecosystem services and considers the cost of a substitute service to an engineered service (Pollack et al., 2013; Mehvar et al., 2018; Lai et al., 2020). Few studies have data developing the economics of observed payments or willingness to pay in Texas due to oyster aquaculture only being recently implemented in the state.

A replacement cost monetary valuation analysis was used within this study because recently published engineered wastewater treatment costs provide detailed cost estimates among multiple wastewater treatment plant configurations (EPA, 2021). This study used “Life Cycle Cost and Assessments of Nutrient Removal Technologies in Wastewater Treatment Plants” (EPA, 2021) as a foundation for costs associated with wastewater treatment plants used to calculate replacement costs and, therefore, the bioextraction value. The “Life Cycle Cost and Assessments of Nutrient Removal Technologies in Wastewater Treatment Plants” contains two cost estimates, one generated by the Computer Assisted Procedure for Design and Evaluation of Treatment Systems (CAPDETWorks™) and the other referencing previous cost estimates in Falk et al., 2011, (EPA, 2021). Prior to CAPDETWorks™, the EPA collected data such as capacity, nutrient removal rates, operational, material, energy, and lifetime costs for WWTPs throughout the country, which are highly variable in function and difficult to compare (EPA, 2007). Both CAPDETWorks™ and Falk et al., 2011 have five performance levels used in this study, indicating different technologies.

The following three steps were used to determine the cost of removing 1 lb. of nitrogen:

1. Calculate TN removal of WWTP given theoretical inputs outlined in the “Life Cycle Cost and Assessments of Nutrient Removal Technologies in Wastewater Treatment Plants” (EPA, 2021) (Table 3).

2. Divide the NPV by the planning period (20 years), establishing an annualized cost of WWTP operation. The NPV combines capital, operation, and maintenance costs into a single cumulative value (EPA, 2021). The planning period is the amount of time monetarily planned for during this development stage, and WWTP's total life expectancy may be longer.

$$\frac{NPV}{20\, year\ planning\ period} = Annualized\ Cost\ of\ WWTP$$

3. Divide the EPA (2021) determined nitrogen removed (in lb., Table 3) by the wastewater treatment process by the annualized NPV to yield price per lb. of N removed by each of the various WWTP configurations (Eq. 2). This is the cost that will be used to assign a monetary replacement value to the estimated removal of nutrients by oysters.

$$\frac{Annualized\ Cost\ of\ WWTP}{Nitrogen\ Removed} = Nitrogen\ Removal\, Per\, lb.$$

3. Results

3.1. ASSETS model

The ASSETS model application results indicate a high level of nutrient-related water quality degradation in Copano Bay. Influencing Factors (I.F.) exhibited a “High” level of expression, primarily due to high nitrogen inputs and low tidal ranges, thus causing low flushing and high residence time. Eutrophic Condition (E.C.) exhibited a “Moderate High” level of expression due to the combination of low chlorophyll and high macroalgal expression, giving the primary conditions a score of “High.” There are no problems with dissolved oxygen or associated with submerged aquatic vegetation, but the moderate score of nuisance and toxic blooms gives a score of “Moderate” for secondary conditions. The combination of high primary and moderate secondary conditions gives an overall eutrophication condition rating of Moderate-High. Future Outlook (F.O.) remains unchanged due to the expectation that the populations within the local watershed will remain stable, with no obvious major catalysts for growth or future change in nutrient loads. Changes farther north in the broader watershed may still add additional pressure to Copano Bay. The overall ASSETS rating is “Bad” because of

Table 3
Five WWTP configurations outlined in the “Life Cycle Cost and Assessments of Nutrient Removal Technologies in Wastewater Treatment Plants” (EPA, 2021). Annual TN loading in all scenarios is 1,220,000 lb., with a concentration of 40 mg l^{-1}.

WWTP configuration	Long-term average concentration (mg l^{-1})	Annual load (lb. yr^{-1})	T.N. removed (lb. yr^{-1})	T.N. removal (removed/influent)
Level 1, AS	30	908,000	312,000	26 %
Level 2-1, A2O	8	244,000	976,000	80 %
Level 3-1, B5	6	183,000	1,037,000	85 %
Level 4-1, B5/Denit	3	91,100	1,128,900	93 %
Level 5-1, B5/RO	0.78	23,800	1,196,200	98 %

the "High" Pressure, "Moderate High" overall eutrophic condition, and expectations of "No Change" in future nutrient pressures.

3.2. FARM modeling

Environmental data from 2 separate years, 2010 and 2014, were used in FARM simulations because they represent observed extreme (i.e. low and high) salinities within Copano Bay. Therefore, FARM results represent the range of possible oyster growth and nutrient removal under those scenarios. Data from 2010 represents a wet year with low salinity, while 2014 represents a dry year with higher salinity. Both scenarios assume a 7-acre lease, with 5.5 acres of productive growing areas divided into 150 m-by-150 m areas, 250 oysters m^{-2}, 25 % mortality, and a 240-day cultivation cycle. The scenario in 2010 resulted in an estimated 4905 lb. of N removal, with no oysters reaching harvestable size under these conditions. Although the oysters are not harvestable, nitrogen is still assimilated into tissue and shell, providing nutrient-regulating services. Using environmental data from 2014, the model resulted in an estimated 7112 lb. of N removal with 95,620 oysters harvested.

3.3. Valuation results

The NPV of each of the five WWTP configurations generally increases with level (i.e. lower target for T.N.) because of the higher costs to remove nutrients from lower concentration effluent. The estimations by Falk et al. (2011) are lower in almost all circumstances except for Level 5-1, B5/RO. Dividing the NPV by the 20-year planning period gives an annualized cost. In the case of Level 1, AS, it would cost $10,200,000 to pay for 20 years of initial costs, O&M, and overall costs consistent with use (Table 4). A few key points explain variations in costs between these two sources. CAPDETWorks™ accounts for operational labor, maintenance labor, materials, chemicals, and energy, while Falk et al., 2011 only include chemicals and energy (EPA, 2021). Falk et al. (2011) also used higher costs associated with construction assumptions (i.e., sheeting, shoring, and higher concrete costs) (EPA, 2021). Differences in annual NPV are due to a 5 % discount rate and a 3.5 % escalation rate for capital, energy, and non-energy components, while the CAPDETWorks™ used a 3 % discount rate and did not escalate costs (EPA, 2021).

Table 4
Annualized Net Present Value per WWTP Configuration (derived from Table 5-3, EPA, 2021) in $2014.

WWTP configuration (target effluent N concentration)	CAPDETWorks™ Annualized Net Present Value (EPA, 2021)	Falk et al. (2011) Annualized Net Present Value (EPA, 2021)
Level 1, AS (no target specified)	$10,200,000	$6,150,000
Level 2-1, A2O (8 mg N l^{-1})	$11,800,000	$8,350,000
Level 3-1, B5 (4–8 mg N l^{-1})	$18,900,000	$10,050,000
Level 4-1, B5/Denit (3 mg N l^{-1})	$13,350,000	$11,700,000
Level 5-1, B5/RO (<2 mg N l^{-1})	$13,750,000	$16,750,000

The discount rate is typically used for the marginal pretax rate of return on investment (EPA, 2021) and reflects the rate of interest on applied future cash flows, reflecting costs to finance projects and their productive outputs. Escalation rates include inflation rate to reflect purchasing power in the future. These costs reflect actual costs across a long period in which costs may be compounding.

Values of nitrogen removal vary between a low of $8.56 for removal of one pound via the "Level 2-1 Anaerobic/Anoxic/Oxic" system valuation performed by Falk et al. (2011), to a high a of $32.69 per lb. using a "Level 1 Activated Sludge (AS)" system as modeled by the CAPDETWorks™ software, with an average cost of $14.87 per pound (EPA, 2021) (Table 5).

Applying TN removal cost per lb. in comparison to nitrogen removal of oysters within the 2010 (4905.285 lb.) and 2014 (7112.113 lb.) results in costs ranging from $41,966 to $232,511 (Table 6).

4. Discussion

4.1. Eutrophication impacts continue

The overall ASSETS rating in Copano Bay is "Bad" because of the "High" Pressure, "Moderate-High" overall eutrophic condition (State), and 'Moderate' Future Outlook (Response). ASSETS performed in nearby estuaries in previous studies in Texas provide some context for comparison. Aransas Bay's overall condition is considered "Moderate," partly buffered by surface water first entering Copano Bay prior to mixing. Further north, Matagorda and San Antonio Bay are considered "Moderate Low" based on results of the ASSETS modeling application south of the Copano Bay study area, the upper Laguna Madre, Baffin Bay, and Corpus Christi Bay all received "High" ratings, indicating that the bays are experiencing symptoms of eutrophication (Bricker et al., 1999, 2007). These results reaffirm findings that South Texas estuaries continue to experience symptoms of eutrophication as found in the NEEA in 1999 and an update in 2007 (Bricker et al., 1999, 2007), as well as the NCCA (EPA, 2021).

Symptoms of eutrophication are also found in local studies in South Texas. Baffin Bay (Wetz et al., 2017; Bugica et al., 2020), Galveston Bay (Bugica et al., 2020), and Oso Bay (Wetz et al., 2016; Bugica et al., 2020) have all experienced high organic nitrogen concentrations and long-term chlorophyll increases. Although many estuaries in Texas exhibit symptoms of eutrophication, some areas, such as Baffin Bay, have 2–5 times the amount of total Kjeldahl nitrogen (TKN) and chlorophyll *a* found in other estuaries in Texas (Wetz et al., 2017), making these waterbodies an even greater concern. The consistency of results of the various eutrophication assessments suggest that the Texas estuaries require additional nutrient management. These studies also highlight the challenge of addressing nutrient-related water quality degradation, as bay systems are unique in subspeciality and response to over-enrichment.

4.2. Nitrogen regulating processes

Estimations of nitrogen removal by the FARM model simulations range from 4905 lb. in 2010, where no oysters were harvested, to 7112 lb. in 2014, with a harvest of 95,620 oysters. Whether or not oysters are

Table 5
Final replacement value per lb. of nitrogen removed using WWTP costs from the "Life Cycle Cost and Assessments of Nutrient Removal Technologies in Wastewater Treatment Plants" (EPA, 2021).

WWTP configuration (target effluent N concentration)	Total N removed (lb.)	CAPDETWorks™ (EPA, 2021) (Cost of removal per lb. of T.N.)	Annualized Falk et al. (2011) (EPA, 2021) (Cost of removal per lb. of T.N.)
Level 1, AS (no target specified)	312,000	$32.69	$19.71
Level 2-1, A2O (8 mg N l^{-1})	976,000	$12.09	$8.56
Level 3-1, B5 (4–8 mg N l^{-1})	1,037,000	$18.23	$9.69
Level 4-1, B5/Denit (3 mg N l^{-1})	1,128,900	$11.83	$10.36
Level 5-1, B5/RO (<2 mg N l^{-1})	1,196,200	$11.49	$14.00

Table 6
Replacement cost values based on the nutrients removed by the farm oyster population in 2010 (4905 lb.) and 2014 (7112 lb.) estimated by the FARM model. The year 2010 represents a year of lower salinity with poor growth, while 2014 represents a year with high stable salinity.

WWTP configuration	2014 scenario (value using EPA, 2021)	2014 scenario (value using Falk et al., 2011)	2010 scenario (value using EPA, 2021)	2010 scenario (value using Falk et al., 2011)
Level 1, AS	$232,511	$140,191	$160,365	$96,691
Level 2-1, A2O	$85,987	$60,846	$59,306	$41,966
Level 3-1, B5	$129,623	$68,926	$89,402	$47,539
Level 4-1, B5/Denit	$84,106	$73,710	$58,008	$50,839
Level 5-1, B5/RO	$81,752	$99,589	$56,385	$68,687

harvested, they are still gaining biomass and consuming nitrogen containing material from the water column, and thus they still contribute to nitrogen removal (as long as they are living and growing). This range represents possibilities dependent on environmental conditions and is a calculation of nitrogen assimilation into the shell and tissue of the oyster. Oysters can regulate other chemical processes, such as denitrification or burial, that act as a sink for nitrogen (Vitousek et al., 1997; Cerco, 2015; Rose et al., 2015a,b; Barrett et al., 2022). Denitrification is a process whereby microbial communities that live in the tissue and shell of oysters, reefs, and sediment near oyster populations convert biologically available nitrogen into dinitrogen gas (N_2) and return it to the atmosphere (Ray et al., 2021). Burial occurs as oysters' nitrogen containing biodeposits (feces and pseudofeces) settle on the sediment surface and remove nitrogen from the water column as they are buried (Newell et al., 2005). Oysters within an off-bottom configuration (i.e., floating containers or "rafts" used in this study) may differ in nitrogen regulating services from the wild reefs in other studies (Pollack et al., 2013; Lai et al., 2020). Estimates within this study only calculate sequestration into tissue and shell, and should be considered a conservative or underestimate of the total amount of nitrogen removed within other processes. When considering these other nitrogen mitigation processes, nitrogen removal rates associated with oyster aquaculture may be higher, as seen in Pollack et al. (2013) and Lai et al. (2020), and were also discussed in the works of Bricker et al. (2018, 2020), where other nitrogen regulating pathways are calculated. One important point to note is that most nitrogen inputs to Copano Bay are from non-point sources, which cannot be addressed by wastewater treatment but can be addressed via oyster bioextraction.

4.3. Developing monetary costs of nitrogen bioextraction

Under influent parameters and nitrogen removal rates provided by the "Life Cycle Cost and Assessments of Nutrient Removal Technologies in Wastewater Treatment Plants" (EPA, 2021), higher levels of technology are typically more economical at larger scales, depending on the desired goals and foci of the WWTP. From level 1 to level 5, the nitrogen removal rate increased from 26 % (312,000 lb.) to 98 % (1,196,200 lb.), representing an increase of 283.4 % in nitrogen removal (Fig. 3). In contrast with cost, the estimated $10,200,000 annualized NPV at Level 1, AS, and $13,750,000 at Level 5-1, B5/RO indicate a 34.8 % cost increase. Falk et al. (2011) level 1 estimation is $6,150,000 and $16,750,000 at level 5, representing an increased cost of 172.4 %. The EPA's CAPDETWorks™ cost estimations are higher than Falk et al. (2011), with the exception of Level 5-1, B5/RO, a five-state reverse osmosis configuration. The most significant difference in cost estimation is at Level 3-1, B5, which is the 5-stage Bardenpho System Wastewater Treatment Configuration. The 5-stage Bardenpho process uses a series of anaerobic, primary anoxic, primary aerobic, secondary anoxic, and secondary aerobic processes (Li et al., 2013). A major consideration within the 5-stage Bardenpho process is the increased costs associated with storage tanks and space, leading to variable space and material costs depending on location (Singureanu and Woinaroschy, 2017).

The costs and removal rates would change under different influent parameters (T.N., flow rate, and other contaminants). The EPA notes that "The study results do not represent a specific, existing WWTP" and that "The key consideration in selecting a functional unit is to ensure the wastewater treatment configurations are compared on the basis of equivalent performance. In other words, an appropriate functional unit allows for an apples-to-apples comparison" (EPA, 2021). Local, site-specific considerations are paramount, and this study and the "Life Cycle Cost and Assessments of Nutrient Removal Technologies in Wastewater Treatment Plants" are used to value and conceptualize nutrient management processes and not to determine a "one-size fits all" approach. Comparisons among CAPDETWorks™, Falk et al. (2011) and Bricker et al. (2018) can therefore be used as a general metric for comparison, and other WWTP metrics can also be used comparatively when the necessary data exists.

Among nutrient reduction options, engineered WWTPs targeting 8 mg l^{-1} T.N. are a comparable data point between Bricker et al. (2018), Falk et al. (2011), and CAPDETWorks™ (EPA, 2021) (Table 7). The costs

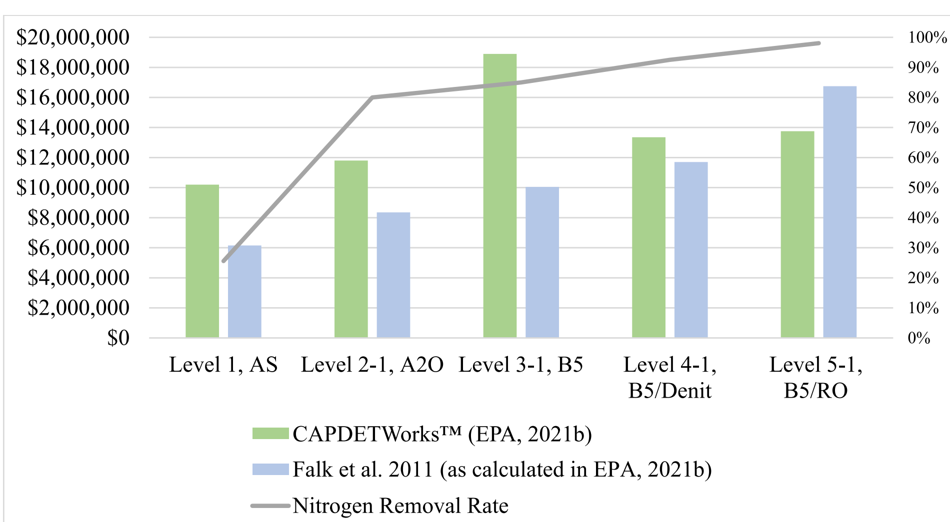

Fig. 3. Annualized NPV of WWTP configurations using two estimations (EPA, 2021) and nitrogen removal rate.

Table 7
Comparison of costs generated through the "Life Cycle Cost and Assessments of Nutrient Removal Technologies in Wastewater Treatment Plants" (EPA, 2021) using the EPA's CAPDETWorks™ and Falk et al., 2011 (EPA, 2021) compared to Bricker et al. (2018) converted to $2014.

Source	Nutrient reduction measure (mg N l^{-1})	Capital costs ($, million)	O and M ($, million)	Annualized cost ($, million)	Nitrogen removed 10^3 lb. yr^{-1}	Average cost $ $lb.^{-1}$ yr^{-1}
Bricker et al. (2018)	WWTP (8)	447	8.39	30.88	2070	$14.86
	WWTP (5)	147	4.29	11.58	677	$17.09
	WWTP (3)	326	12.8	28.85	634	$45.52
	Agricultural BMP	–	–	7.8	1310	$5.99
	Full Urban BMP	–	–	165.57	1040	$161.51
CAPDETWorks™ (EPA, 2021)	Level 1, AS (30)	204	5.14	10.2	312	$32.69
	Level 2-1, A2O (8)	236	5.47	11.8	976	$12.09
	Level 3-1, B5 (6)	378	5.8	18.9	1037	$18.23
	Level 4-1, B5 (3)	267	6.84	13.35	1128	$11.83
	Level 5-1, B5/RO (0.78)	275	8.32	13.75	1196	$11.49
Falk et al. (2011) (EPA, 2021)	Level 1, AS (30)	123	1.02	6.15	312	$19.71
	Level 2-1, A2O (8)	167	1.41	8.35	976	$8.56
	Level 3-1, B5 (6)	201	2.62	10.05	1037	$9.69
	Level 4-1, B5 (3)	234	3.57	11.7	1128	$10.36
	Level 5-1, B5/RO (0.78)	335	5.57	16.75	1196	$14.00

varied from a high of $14.63 (Bricker et al., 2018) and $12.09 to $8.56 with the CAPDETWorks™ and Falk et al. (2011), respectively. WWTP targets of 3 mg l^{-1} T.N. are also comparable, from $45.52 (Bricker et al., 2018), $11.83 (EPA, 2021), and $10.36 (Falk et al., 2011). Estimations here are impacted by source information. Some key differences among the cost methodology used by Falk et al. (2011) include a higher discount rate (5 %) and the use of an escalation rate (3.5 %) compared to the lower discount rate (3 %) used in CAPDETWorks™ which also did not escalate any costs (EPA, 2021). Bricker et al. (2018) utilizes results of Evans (2008) cost analysis of 151 WWTP facilities across New Hampshire, Vermont, Massachusetts, Connecticut, and Connecticut. Both CAPDETWorks™ and Falk et al., 2011 assume the same input parameters, 10 mgd and 40 mg l^{-1} of TN in a hypothetical WWTP, while many of the design flow and concentration parameters found in Evans, 2008 have significantly less flow (average of 1.96 mgd), roughly 20 % of the flow capacity of the theoretical WWTP. Influent data is not listed for 151 sites, but target configurations are also highly variable.

4.4. Site-specific solutions and nutrient credit trading

A comparison of costs to remove nutrients using several different nutrient management strategies shows that the lowest cost per lb. of nitrogen removal is agricultural best management practices (BMPs) ($5.99), and the highest costs are full urban BMPs ($161.51) (Rose et al., 2015a,b; Bricker et al., 2018). According to the USGS SPARROW model results, fertilizers comprise over 40 % of the nitrogen loading into Copano Bay, suggesting that agricultural BMPs are well suited in the sparsely populated surrounding areas (Wise et al., 2019). The availability of land relative to a city center also encourages agricultural BMPs. The coastal bend provides sorghum, corn, and cotton as agricultural commodities (Fernandez et al., 2012; USDA, 2017). The USDA Economics Research Service (ERS) estimates around 143.5 lb. of nitrogen fertilizer were applied per acre of corn production in 2010, among the highest in agricultural commodities tracked by the ERS (USDA, 2017). In 2006, 35 % of U.S. field crop was estimated to utilize nitrogen applications with proper rate, timing, or method as established by USDA's Natural Resources Conservation Service (NRCS) BMP (Ribaudo et al., 2011). Subsequent expansion of corn production is a possible driver for increased nitrogen loading to the Copano Bay/South Texas coastal area. Municipal wastewater treatment discharge is also a significant nitrogen source, and retrofitting or design updates may be an alternative method for further reductions. Agricultural BMPs consist of many practices (policy instruments, cropland to forest conversion, fertilizer reduction, alternative nitrogen sources, no-till, etc.) with varying costs and nitrogen removal efficacies (Stephenson et al., 2010; Ribaudo et al., 2011). Developing a more accurate agricultural BMP cost for the coastal Texas plains would involve a more targeted approach to understanding farming practices, nitrogen use, soil chemistry, and other factors influencing nitrogen use efficiency (Ribaudo et al., 2001; Cassman et al., 2002).

In addition to engineered improvements with WWTP and agricultural BMPs, Copano Bay can also utilize marine space as nutrient management. Estimations of the Mission-Aransas Estuary have found approximately $65.56 per acre of nitrogen regulation services (denitrification, burial, and physical transport from the system via harvest) across the system, totaling $293,993, that are already occur naturally as an ecosystem service (Pollack et al., 2013). As of September 2023, seven farms totaling 33 acres have been approved by the Texas Parks and Wildlife Department (n.d.), over four times the lease acreage utilized in this study, with many more pending and conditional which will greatly increase the nutrient removal and monetary value of those acres. Oyster growers directly contribute to nitrogen regulation in Copano Bay, and provide a monetary benefit from harvested crop and regulating services.

In some U.S. states (e.g., Massachusetts), nutrient removal by oyster and clam cultivation has been approved for crediting and use within comprehensive coastal management plans for nutrient removal (Town of Mashpee, 2015; Reitsma et al., 2017). The two Chesapeake Bay states have nitrogen credit trading programs in development, Maryland (since 2019) and Virginia (since 2005), with research suggesting that coastal managers in other states may seek to replicate the program to add to the suite of nutrient management practices (Rose et al., 2014). Within the U.S. Chesapeake Bay, a regional approach for crediting nutrient removal by harvest of oyster tissue from aquaculture was approved in 2016 (Cornwell et al., 2016). The nutrient-regulating services of oyster aquaculture can supplement existing nutrient management methods and allow flexibility for coastal managers dealing with nonpoint nutrient reduction (Stephenson and Shabman, 2017).

4.5. Implications and future work

This research calculated nitrogen bioextraction potential in a model 7-acre oyster farm using a floating configuration at a density of 250 oysters m^{-2}, resulting in a range of nitrogen removal of 4905–77,112 lb. yr^{-1}, or around 700–1016 N lb. $acre^{-1}$. This represents <1 % of the 1.15 million lb. nitrogen total loading to Copano Bay estimated by the SPARROW model. However, Copano Bay is over 50,000 acres, meaning that scaling up oyster aquaculture is possible, even considering siting limitations. Other farm configurations, such as higher oyster densities or alternate equipment, could increase nitrogen removal (Clements and Comeau, 2019). The nitrogen bioextraction value is similar and as efficient to other approved best management practices, further validating oyster nitrogen bioextraction as used in tandem with other nitrogen reduction measures (Rose et al., 2015a,b; Bricker et al., 2018; Parker and Bricker, 2020).

The utilization of bivalves in nutrient management plans and nutrient credit trading programs is becoming increasingly more accepted and is gaining traction as a mechanism for non-point nitrogen removal (Ribaudo et al., 2005; Ferreira and Bricker, 2016; Bricker et al., 2020). As the oyster aquaculture industry in Texas expands, its relevance and value in large-scale management of estuaries in southern Texas will also become increasingly important. Developing feasibility studies and early examples of nutrient credit trading as seen in other states such as Maryland (Weber et al., 2018; MDE, 2023), will simultaneously incentivize entrepreneurs in the oyster aquaculture in Texas while maintaining a network of nitrogen mitigation options in valuable coastal waters.

5. Conclusions

With continued increases in population, industry, and agriculture development, nutrient pollution continues to be a growing concern. Eutrophication, caused in part by nutrient pollution to a waterbody, is a complex problem that causes many functional changes to an ecosystem. The implementation of oyster aquaculture is a potential solution to be used alongside other large-scale nutrient management strategies. This study aimed to develop an estimate for potential nutrient removal and to estimate the monetary value of nitrogen bioextraction by cultivated oysters in the newly approved off-bottom aquaculture industry compared to WWTP configurations. Nitrogen removal costs were used from several theoretical, generalized WWTPs and applied to the bioextracted nitrogen to assign a monetary value. This study found, based on FARM model results, that a typical 7-acre oyster farm could potentially remove between 4905 and 7112 lb. of nitrogen with a potential range in value of \$41,966 to \$232,511, based on WWTP configuration and environmental growing conditions. This represents \$8.56–\$32.69 per pound of removed nitrogen. Note that these monetary value estimates for nitrogen removal are underestimates of the value of total ecosystem services provided by oyster. Bivalve cultivation has many regulating services (wave energy dissipation, habitat-forming, nutrient uptake, etc.) that add value outside the cultivated species' sales (Alleway et al., 2019) but have not been measured. Here the value has been assigned only to nitrogen removal via assimilation into tissue and shell of the farm oyster population.

In some cases, agricultural BMPs can represent a cost-effective alternative as a nutrient reduction strategy (\$5.90 per lb. estimation by Bricker et al., 2018; \$0.1–470 cost per lb. estimation by Rose et al., 2015a,b) compared to other management techniques. As a predominantly agricultural area, fertilizer usage is the dominant nitrogen source in the ecosystems surrounding Copano Bay. Enhancing BMPs and adjusting fertilizer usage to increase nitrogen use efficiency at a lower cost may prevent the need for further costly mitigation tools. Development of oyster populations in Texas through aquaculture can offer estuarine systems further buffering capability against nitrogen over-enrichment while offering other ecosystem services simultaneously.

As environmental goals feature more functional and ecosystem-based management strategies, the ecological value of these species is increasing. Compilations of these values and comparison to traditional nutrient management strategies can assist coastal management by understanding marine bivalves' relevance and broader value. Expanding this work should alternatively compile additional benefits of ecosystem services of oyster populations and provide values for aspects such as fishery habitats and local workforce benefits. Furthermore, bivalve nitrogen bioextraction should be explored in different estuarine conditions, particularly across areas with well-defined spatial gradients of nitrogen pollution where their efficacy can be compared.

CRediT authorship contribution statement

Anthony R. Lima: Writing – original draft, Visualization, Software, Methodology, Formal analysis, Data curation, Conceptualization. **Jennifer Pollack:** Writing – review & editing, Supervision, Resources, Project administration, Funding acquisition, Conceptualization. **Joe M. Fox:** Writing – review & editing, Supervision, Project administration, Methodology, Funding acquisition, Data curation. **João G. Ferreira:** Software, Methodology. **Alhambra Martínez Cubillo:** Software, Formal analysis, Data curation. **Anthony Reisinger:** Software, Methodology, Formal analysis, Data curation. **Suzanne Bricker:** Writing – review & editing, Writing – original draft, Validation, Supervision, Software, Resources, Project administration, Methodology, Formal analysis, Conceptualization.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Data availability

Data will be made available on request.

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