2017

Differential effects of bivalves on sediment nitrogen cycling in a shallow coastal bay

Ashley Smyth
Virginia Institute of Marine Science

Anna E. Murphy
Virginia Institute of Marine Science

Iris C. Anderson
Virginia Institute of Marine Science

BK Song
Virginia Institute of Marine Science

DIFFERENTIAL EFFECTS OF BIVALVES ON SEDIMENT NITROGEN CYCLING IN A SHALLOW COASTAL BAY

Ashley R. Smyth[1+*], Anna E. Murphy[1], Iris C. Anderson[1], Bongkeun Song[1]

[1] Virginia Institute of Marine Science, School of Marine Science, College of William and Mary, Gloucester Point, Virginia, 23062, USA

[+]Present Address: Soil and Water Sciences Department, Tropical Research and Education Center, University of Florida, Homestead, FL 33031, USA

* Corresponding author. E-mail: arsmyth@gmail.com

Running head: N Dynamics in Clam Aquaculture and Oyster Reefs

Keywords: denitrification, DNRA, *Mercenaria mercenaria*, *Crassostrea virginica*, nitrogen, eutrophication

ABSTRACT

In coastal ecosystems, suspension-feeding bivalves can remove nitrogen though uptake and assimilation or enhanced denitrification. Bivalves may also retain nitrogen through increased mineralization and dissimilatory nitrate reduction to ammonium (DNRA). This study investigated the effects of oyster reefs and clam aquaculture on denitrification, DNRA, and nutrient fluxes (NO_x, NH_4^+, O_2). Core incubations were conducted seasonally on sediments adjacent to restored oyster reefs (*Crassostrea virginica*), clam aquaculture beds (*Mercenaria mercenaria*) which contained live clams, and bare sediments from Smith Island Bay, Virginia, USA. Denitrification was significantly higher at oyster reef sediments and clam aquaculture site than bare sediment in the summer; however DNRA was not enhanced. The clam aquaculture site had the highest ammonium production due to clam excretion. While oyster reef and bare sediments exhibited seasonal differences in rate processes, there was no effect of season on denitrification, DNRA or ammonium flux at the clam aquaculture site. This suggests that farm management practices or bivalve metabolism and excretion may override seasonal differences. When water column nitrate concentration was elevated, denitrification increased in clam aquaculture site and oyster reef sediments but not in bare sediment; DNRA was only stimulated at the clam aquaculture site. This, along with a significant and positive relationship between denitrification and sediment organic matter, suggests that labile carbon limited nitrate reduction at the bare sediment site. Bivalve systems can serve as either net sinks or sources of nitrogen to coastal ecosystems, depending mainly on the type of bivalve, location and management practices.

INTRODUCTION

Eutrophication, frequently caused by excess nitrogen (N) inputs, affects coastal systems worldwide (Diaz and Rosenberg 2008). Excess N can fuel primary production leading to algal blooms, dead zones, and habitat loss (Paerl 1997, Hauxwell et al. 2001). N can be removed from ecosystems by sediment denitrification, a stepwise reduction pathway of nitrate to dinitrogen gas (N_2). Modification to the timing and rate of carbon delivery, concentration of nitrate, the terminal electron acceptor, and accumulation of sulfide, may promote dissimilatory nitrate reduction to ammonium (DNRA) instead of denitrification, resulting in N retention rather than removal (Burgin and Hamilton 2007, Hardison et al. 2015). DNRA and denitrification compete for nitrate produced via nitrification, the oxidation of ammonium to nitrate/nitrite, or supplied directly from the water column. The potential for coastal systems to remove N and combat the negative consequences of eutrophication relies in part on the competition between these two nitrate reducing processes.

There is growing interest in using shellfish to mitigate the effects of eutrophication and manage N pollution (Bricker et al. 2014, Kellogg et al. 2014, Rose et al. 2014). Suspension feeding bivalves such as oysters and clams, provide top-down control of phytoplankton biomass and enhance sediment N cycling through benthic-pelagic coupling (Dame et al. 1984, Newell 2004, Smyth et al. 2013a, Murphy et al. 2016a). These organisms can also serve as habitat for nitrifying and denitrifying bacteria (Welsh and Castadelli 2004, Stief 2013, Welsh et al. 2015). As bivalves feed, particulate N contained in phytoplankton and other organic matter is removed from the water column. A portion of this N is assimilated, of which a fraction is excreted as dissolved N directly to the water column; the remainder is egested as biodeposits and transferred to the sediments. N in the biodeposits can be buried or utilized by the microbial community (e.g.

46 remineralized, assimilated, nitrified) and may enhance denitrification (Newell et al. 2002,
47 Kellogg et al. 2013, Smyth et al. 2013a). Additionally, the delivery of organic carbon to the
48 sediments in biodeposists may stimulate DNRA, which is favored over denitrification in high
49 carbon and low nitrate conditions (Tiedje 1988, Burgin and Hamilton 2007, Hardison et al.
50 2015).

51 To date, evidence of the effectiveness for bivalves to control N availability is equivocal
52 (Table 1). The amount of N that is recycled rather than removed seems to depend on species,
53 environmental characteristics and grow-out practices. For example, commercial hard clam
54 (*Mercenaria mercenaria*) aquaculture in a shallow polyhaline tributary of Chesapeake Bay
55 contributed to reduced conditions in the sediments and accumulation of sulfide, which resulted in
56 N regeneration via DNRA rather than removal via denitrification (Murphy et al. 2016a).
57 However, in the highly eutrophied Po River Delta (Italy) sediments from clam (*Tapes*
58 *philippinarum*) aquaculture sites had higher rates of denitrification than DNRA (Nizzoli et al.
59 2006). Oyster (*Crassostrea virginica*) reefs tended to increase denitrification relative to bare sites
60 (Piehler and Smyth 2011, Smyth et al. 2013b, Kellogg et al. 2013, Humphries et al. 2016) but
61 had a minimal influence on sediment N cycling in eutrophic systems (Hoellein and Zarnoch
62 2014). The majority of studies on oyster reefs have focused on denitrification and estimates of
63 DNRA are limited (Table 1). Based on the few studies which are available, denitrification is
64 favored over DNRA in oyster reef sediments and sediments affected by oysters, either adjacent
65 to oyster reefs or floating aquaculture cages, tended to have higher rates of DNRA than bare
66 sediment (Smyth et al. 2013b, Erler et al. 2017).

67 Within the same ecosystem, the effects of bivalves on sediment biogeochemistry will
68 likely differ based on the type of bivalve, growing conditions and physical substrates that can be

69 colonized by denitrifying and nitrifying bacteria. For example, in Chesapeake Bay, the largest
70 estuary in the US, clams (*Mercenaria mercenaria*) and oysters (*Crassostrea virginica*) are the
71 predominate bivalve species. Clams may enhance nitrification by supplying ammonium and
72 increasing O_2 penetration depth through bioturbation, leading to more coupled nitrification-
73 denitrification (Pelegrí and Blackburn 1994, Welsh 2003, Nizzoli et al. 2006). Additionally, the
74 shell, soft tissue and digestive system can be colonized by nitrifying and denitrifying bacteria
75 and these areas exhibited high rates of these processes (Welsh and Castadelli 2004, Stief 2013,
76 Welsh et al. 2015). However, in an aquaculture context some of the natural function of clams
77 may be impacted by the method of cultivation. In the Chesapeake Bay region, the high density of
78 clams and use of predator exclusion nets, which restrict clam movement, modify water flow and
79 serve as habitat for macroalgae, may affect exchanges at the sediment-water interface (Secrist
80 2013). The predator exclusion net is a plastic netting placed flush to the sediment surface (over
81 the clams) and held in place with reinforcing bars and sandbags. Macroalgae grows rapidly using
82 the NH_4^+ generated by from the clams requires periodic removal (Murphy et al. 2015). The
83 predator exclusion net and the high density of may contribute to greater organic enrichment of
84 sediment (Newell 2004), resulting in reduced and sulfidic conditions that promote DNRA
85 (Murphy et al. 2016a). While natural clam beds still exists in the US, clam aquaculture is
86 becoming a more common feature in the coastal landscape (Murphy et al. 2016b, Emery 2015).
87 Oysters reefs have dramatically declined in area in coastal ecosystems due to factors such
88 as disease and over-fishing (Beck et al. 2011). Oysters are sessile, epifanual suspension feeding
89 bivalves that form biogenic reefs. The three-dimensional reef structure on top of the sediment
90 helps concentrate organic matter on the sediment surface. The reef structure increases
91 biogeochemical cycling compared to unstructured sediment (Smyth et al. 2016) and provides

habitat and refugia for diverse infaunal and epifaunal communities, which include many bioturbating organisms (Kellogg et al. 2013). While the reef structure alone can increase biogeochemical cycling, the oysters can have a direct impact on nitrogen cycling. Filtration and associated biodepostion delivers organic C and N to the sediments, leading to elevated rates of nitrogen cycling compared to bare sediment (Newell et al. 2002, Kellogg et al. 2013, Smyth et al. 2013a). At the same time, the oysters add ammonium directly through excretion and consume oxygen through respiration (Kellogg et al. 2013, Smyth et al. 2013a). Additionally, the shell and gut of oysters can serve as habitat for nitrifying and denitrifying bacteria (Caffrey et al. 2016). Clam aquaculture and oyster reefs have been shown to affect both DNRA and denitrification relative to bare sediments in a variety of coastal systems (Table 1); however, few studies have measured both processes simultaneously. Given the different ecological features between clam aquaculture and oyster reefs, it is expected that the effect of these organisms on benthic metabolism and nitrogen cycling would be different.

As aquaculture continues to expand, there is competition for available space with oyster reef restoration efforts, since the range of conditions tolerated by clams and oysters is similar. In order to aid in the evaluation of ecosystem services provided by clam aquaculture and oyster reef restoration, we assessed sediment N dynamics at a restored oyster reef (C. *virginica*) and clam aquaculture site (M. *mercenaria*). Specifically, we investigated the relative importance of DNRA and denitrification in sediments adjacent to a restored oyster reef and from a clam aquaculture bed as well as a reference bare subtidal flat (bare site) sediment, seasonally in Smith Island Bay, Virginia, USA. We hypothesized that sediments from oyster reefs and clam aquaculture would enhance microbial N cycling activity compared to the bare site due to increased C and N delivery to the sediments through bivalve biodeposition. We expected restored oyster reef sediments to

have higher rates of denitrification than sediments from clam aquaculture sites, but that clam aquaculture sites would have greater DNRA than oyster reef sediments because of the high clam density and the use of predator exclusion nets that enhance organic matter accumulation. Additionally, we expected seasonal differences in rate processes associated with temperature and food availability for the bivalves.

MATERIALS AND METHODS

Study site and field sample collection

To determine how clam aquaculture and oyster reefs affect sediment N cycling, sediment samples were collected from a clam aquaculture site which contained littleneck clams (3.5 cm to 5.2 cm long) and from sediment adjacent to a restored oyster reef in Smith Island Bay, VA (37°08'57.08" N, 75°53'06.81" W). Smith Island is one of the southern barrier islands of the Delmarva Peninsula and part of the Virginia Coastal Reserve Long Term Ecological Research site (VCR LTER; Fig 1). Smith Island Bay has an average water depth of 0.4 m, semidiurnal tides that range about 1.2 m and a residence time from 4-12 days (Safak et al. 2015). Sampling sites included (1) a restored oyster reef located on an intertidal flat, (2) a clam aquaculture bed, which is part of a commercial aquaculture lease with predator exclusion nets and (3) a bare subtidal flat (bare site) located approximately 50 m from the reef and aquaculture operation. Smith Island Bay was selected for this study because both clam aquaculture and restored oyster reefs are found within close proximity to each other. Samples were taken for sediment biogeochemical flux experiments and sediment physico-chemical properties (sediment organic matter, benthic microalgal biomass and pore-water nutrients) seasonally in April, July, and November 2014.

Continuous Flow Incubations

Continuous flow core incubations were used to examine rates of N exchanges at the sediment-water interface. Triplicate sediment cores (9.5 cm inner diameter x 10 cm sediment depth) were collected by hand from each of the three sampling sites. Sediment samples from the oyster reef were collected adjacent to the reef and did not contain live oysters. For samples from the clam aquaculture site, the predator exclusion net was removed prior to sample collection. Associated infauna were left undisturbed for all samples; therefore, if present, live clams were not removed from the sediment cores collected at the clam aquaculture site. Water chemistry was assessed at the site with a YSI 6600 datasonde, (YSI, Inc., Yellow Springs, OH, USA). Approximately 170 L of water were collected from Smith Island Bay for use in the continuous flow core incubations and for dissolved nutrient analyses.

Upon collection, sediment cores and water were transported to an environmental chamber set to *in situ* temperature at the Virginia Institute of Marine Science (VIMS) in Gloucester Point, Virginia, USA. At VIMS, cores were submerged in aerated site water and held in the dark for 12-16 hrs. The following day each core was sealed with a gas tight lid equipped with inflow and outflow ports and incubated in a continuous flow system (Gardner and McCarthy 2009). Unfiltered, aerated site water was passed over the cores at a flow rate of 3 ml per minute. Dark conditions were maintained throughout the course of the incubation to reduce the effects of photosynthetic algae (An and Joye 2001)and to prevent the formation of bubbles that affect dissolved gas concentrations (Reeburgh 1969).

Cores were acclimated for 24 hours before sampling to allow the system to reach steady state. Samples for dissolved nutrients (combined nitrate/nitrate (NO_x) and ammonium (NH_4^+)) and gasses (O_2 and Ar) were collected three times over the course of 24 hours after the initial

pre-incubation period. A bypass line that flowed from the replacement water tank directly into the sample vial and not through a core tube was used to determine the concentration of dissolved constituents entering the cores. After the initial 24-hour sampling period, replacement water was enriched with ^{15}N-$NaNO_3$ (98atm%) to a final concentration of ~100 μmol L^{-1} for isotopic pairing experiments (Nielsen 1992, Risgaard-Petersen et al. 1995, Yin et al. 2014). After a 24-hour equilibration period, samples were collected three times over the following 24-hours to measure DNRA and denitrification. At the end of the 4-day experiment average outflow oxygen concentration ranged from 4.8 ± 0.15 mg/l O_2 (69% of saturation) in the summer to 7.8 ± 0.9 mg/l O_2 (91% of saturation) in the fall.

Sediment physico-chemical properties

Upon completion of the flux incubations, clams were removed and counted. The upper 2 cm of sediment were homogenized and analyzed for sediment organic matter (SOM), determined by loss on ignition (n=3). Sediments were dried for 24 hours at 60°C then combusted at 525°C for 4 hours. The difference between the weights of dried and combusted samples constituted SOM, expressed as a percentage of the total sediment mass. Pore-water (n=3) was collected in the field from the clam aquaculture site, bare sediment and oyster reef sediment using a stainless steel push-point sampler (2 cm screen; MHE Products, East Tawas, MI, USA), inserted 5-7 cm into the sediment. Pore-water nutrient samples were immediately filtered (0.45 μm Whatman polyethersulfone) and frozen until analysis for dissolved inorganic nitrogen (DIN). Pore-water sulfide samples were immediately fixed with zinc acetate and analyzed on a spectrophotometer within a week of collection (Cline 1969). Sediment samples for benthic chlorophyll *a*, a proxy for algal biomass (n=3), were collected at the clam aquaculture site, bare sediment and oyster reef adjacent sediment from the upper 0.3 cm of sediment using a cut-off syringe (1.1cm ID) and

183 stored frozen until analysis. Samples for sediment chlorophyll *a* were extracted with 10 ml of
184 90% ethanol, sonicated for 30 seconds and extracted at -15°C for 24 hours. Benthic algal
185 biomass was determined using spectrophotometry (Lorenzen 1967), modified to include the
186 extraction of the sediment in 10 ml of solvent (Pinckney et al. 1994). Samples for pore-water and
187 benthic chlorophyll *a*, were collected prior to removing the predator exclusion net at the clam
188 aquaculture site.
189
190 **Analytical Methods and Calculations**
191 Samples for nutrient analysis were immediately filtered through a 0.45 µm Whatman
192 polyethersulfone (PES) filter and frozen until analysis. Filtrate was analyzed with a Lachat
193 Quick-Chem 8000 (Lachat Instruments, Milwaukee, WI, USA) automated ion analyzer for NO_x
194 (combined NO_3^- and NO_2^-) and NH_4^+. Detection limits for NO_x and NH_4^+ were 0.20 and 0.36
195 µM, respectively.
196 Samples for dissolved gasses were collected by filling 12 ml Exetainer vials from the
197 bottom up. Vials were allowed to overflow by several volumes before being preserved with 100
198 µL of saturated $ZnCl_2$. Exetainers were capped and stored underwater below collection
199 temperature until analysis for dissolved gasses (O_2, $^{29}N_2$, $^{30}N_2$, Ar) on a membrane inlet mass
200 spectrometer (MIMS) (Kana et al. 1994, An et al. 2001). An inline furnace was added to the
201 MIMS for $^{29}N_2$ and $^{30}N_2$ samples to increase precision and remove O_2 which can lead to
202 overestimation of denitrification (Lunstrum and Aoki 2016). Samples for DNRA ($^{15}NH_4^+$) were
203 filtered through a 0.45 µm Whatman polyethersulfone (PES) filter and frozen until analysis using
204 the OX/MIMS method (Yin et al. 2014). The OX/MIMS method uses hypobromite iodine
205 solution to oxidize $^{15}NH_4^+$ to $^{29}N_2$ or $^{30}N_2$, and concentrations of both isotopic species were

206 determined on the MIMS with an inline furnace (Yin et al. 2014). The production of $^{29}N_2$ and
207 $^{30}N_2$ associated with the hypobromite oxidation of $^{15}NH_4^+$ was calculated by difference between
208 paired oxidized and unoxidized samples, including those from the bypass line.
209 Fluxes were calculated as:

210 $$J = \left(\left[i_{outflow}\right] - \left[i_{inflow}\right]\right) \times \frac{F}{A}$$

211 where $[i_{outflow}]$ and $[i_{inflow}]$ are the concentrations (μM) of dissolved constituents leaving and
212 entering the core, respectively; F is the peristaltic pump flow rate (3 ml min^{-1}); and A is the
213 surface area of the core (m^2). A positive flux indicates release from the sediment to the water
214 column while a negative flux represents uptake from the water column by the sediment. Negative
215 O_2 fluxes are expressed as sediment oxygen demand (SOD). Denitrification of ambient $^{14}NO_3^-$
216 (D_{14}) and added $^{15}NO_3^-$ (D_{15}) were calculated using the isotope pairing technique equations
217 (Nielsen 1992):

218 $$D_{15} = p29 + 2p30$$

219 $$D_{14} = D_{15} \times (p29/2p30)$$

220 where p29 and p30 are production rates of $^{29}N_2$ and $^{30}N_2$, respectively. In an ecosystem such as
221 Smith Island Bay, where ambient water column NO_x^- is low (less than 1 μM) relative to the
222 experimentally-added ^{15}N-$NaNO_3$ (~100 μM), D_{15} is considered the potential denitrification rate
223 or the capacity of the sediments to denitrify when NO_x is provided in excess. Rates of
224 denitrification utilizing nitrate in the water (D_w) were calculated based on the concentration (μM)
225 of $^{15}NO_3^-$ relative to $^{14}NO_3^-$ in the inflow water, times D_{15} using the following equation:

226 $$D_w = ([^{14}NO_3^-] / [^{15}NO_3^-]) * D_{15}$$

227 Denitrification supported by nitrate produced through nitrification in the sediments (D_n) was
228 calculated by the difference between D_{14} and D_w (Nielsen 1992). We did not account for

incomplete denitrification resulting in N_2O production or N_2 production from anammox. Potential rates of DNRA ($DNRA_{15}$) were determined as $^{15}NH_4^+$ production (An and Gardner 2002). The concentration of $^{15}NH_4^+$ was determined from the total $^{15}N_2$ produced after hyprobromite oxidation as described by Yin et al. (2014). Ambient DNRA ($DNRA_{14}$) was calculated based on the assumption that the relative rates of DNRA utilizing $^{15}NO_3^-$ and $^{14}NO_3^-$ occur at the same ratio as denitrification (Risgaard-Petersen et al. 1995):

$$DNRA_{14} = DNRA_{15} * (D_{14}/D_{15})$$

DNRA, based on NO_3^- from the water column ($DNRA_w$), was calculated using the ratio of $^{14}NO_3$ to $^{15}NO_3$ concentration. Rates of DNRA coupled to sediment nitrification ($DNRA_n$) were estimated from $DNRA_w$ and the ratio between D_n and D_w (Risgaard-Petersen et al. 1995).

Nitrification rates were calculated as the sum of ambient denitrification (D_{14}), ambient DNRA ($DNRA_{14}$), and NO_x^- effluxes:

$$\text{Nitrification} = \text{Positive } NO_x \text{ flux} + D_{14} + DNRA_{14}$$

DNRA measurements are considered conservative because we did not extract NH_4^+ from the sediment and only calculated for ammonium fluxing to the water column (i.e. DNRA is in the sediment not the overlying water) (Bruesewitz et al. 2013). Since nitrification rates are calculated from DNRA, these are also conservative.

Statistical analysis

Statistical analyses were performed in R 2.13.1 (R Developement Core Team 2015). Mixed effects models were used to examine the interactive effects of site (oyster reef sediment, clam aquaculture sediment, bare sediment) and season for nutrient fluxes, D_{14}, D_{15}, $DNRA_{14}$, $DNRA_{15}$, and SOM. The mixed effects model (*lme* function from the 'nlme' package) consisted of a random effect of number of bivalves in sediment cores, to account for the fact that a random

number of clams were included in samples from the clam aquaculture site and no oysters were included in oyster reef samples, and fixed effect of site and season. Benthic chlorophyll *a*, pore-water DIN and H_2S, which were collected from the field, were also analyzed with a linear model but did not include number of clams (*gls* function from the 'nlme' package). Tukey HSD post-hoc analysis was used to compare means when an effect was significant. A mixed effects model was also used to compare ambient and potential rates (i.e. D_{15} to D_{14} as well as $DNRA_{15}$ to $DNRA_{14}$) measured from the same core in each season, followed by Tukey HSD post-hoc analysis. Linear regressions were used to assess the relationship between sediment oxygen demand (SOD) or sediment organic matter (SOM) and nitrate reduction rates, calculated nitrification and nutrient fluxes across all sites and seasons. Pearson correlation coefficients were also calculated. Assumptions of normality and homogeneity were tested using Shapiro-Wilkes and Levene's tests, respectively. Logarithmic or Box-Cox transformations were preformed if necessary to meet assumptions of analyses. All analyses were considered significant at the $p<0.05$ level.

RESULTS

Site Characteristics

Water temperatures, salinity and dissolved nutrient concentrations varied between seasons at Smith Island Bay (Table 2). Over the course of the study, water temperature ranged from 14ºC to 25ºC, with the highest temperature in summer and lowest in fall. Salinity was relatively consistent, varying by 3 units over the course of the study. NH_4^+ concentration ranged from 0.45 μM to 3.06 μM, with the highest concentration in the spring. NO_x concentration was less than NH_4^+ for all seasons and only detectable in the fall when concentration was 0.72 μM. Dissolved oxygen (DO), measured around mid-day, was always above 90% sat. and ranged from

6.36 mg/l (91.5% sat.) in the summer to 8.05 mg/l (98.1% sat.) in the spring for Smith Island Bay (Table 2).

Pore-water NO_x was similar across seasons (Table 3; p=0.424) and between sites (Table 3; p=0.419). There was no seasonal effect on pore-water NH_4^+ (Table 3; p=0.098) but there were site differences (Table 3; p=0.001). Pore-water NH_4^+ concentration was higher at the clam aquaculture site compared to bare sediment (Tukey HSD; p=0.001) and oyster reef sediment (Tukey HSD; p=0.002). Pore-water H_2S varied seasonally (Table 3; p=0.012), with the highest concentration in the summer but there were no differences between sites (p=0.092). SOM, varied between the sites (Table 3; p=0.005) and was significantly higher at the oyster reef and clam aquaculture site compared to the bare sediment (Tukey HSD; p=0.001, p=0.001, respectively). The interaction between site and season affected benthic chlorophyll *a* (p=<0.001). Sediment chlorophyll *a* was higher in the fall compared to the spring and summer for each site (Table 3). The sediment cores from the clam aquaculture site contained different numbers of clams, ranging from 0 to 4. There were 0 clams present in cores collected from the clam aquaculture site in the summer, 4 clams in each core collected during the fall, in the spring two cores contained 3 clams and one core contained 4 clams (Supplemental Table 1).

Nitrate Reduction Rates

Denitrification (D_{14}) rates showed a significant interaction between site and season (p<0.001). Oyster reef sediments had higher D_{14} in the summer compared to spring and fall, clam bed sediments showed no difference among seasons and the bare sediment exhibited a sequential decline from spring, summer, to fall (Figure 2). There were also differences between the sites within each season. During the summer, D_{14} was highest in sediments from oyster reef sediment and lowest at the bare sediment with clam aquaculture site having an intermediate rate. In the

298 fall, D_{14} at the clam aquaculture site was higher than the bare sediment and oyster reef sediment
299 but in the spring there were no differences between the sites.
300 In contrast to D_{14}, $DNRA_{14}$ showed a significant difference between sites ($p=0.045$) and
301 season ($p=0.026$; Figure 2), with no significant interaction ($p=0.830$). $DNRA_{14}$ was similar in the
302 spring and fall (Tukey HSD, $p=0.47$) but higher in the fall compared to the summer (Tukey
303 HSD, $p=0.021$). Overall, $DNRA_{14}$ was not significantly enhanced at the clam aquaculture site or
304 oyster reef sediment compared to the bare sediment (Tukey HSD, $p=0.219$, 0.601, respectively)
305 but $DNRA_{14}$ at the clam aquaculture site was significantly higher than the oyster reef sediment
306 (Tukey HSD, $p=0.038$). For all sites, D_{14} was at least 3 times greater than $DNRA_{14}$ (Fig 2a). The
307 majority of D_{14} and $DNRA_{14}$ was coupled to nitrification (Figure 2); D_n ranged from 94.7% of
308 total nitrate reduction at the oyster reef sediment in the fall to 99.8% of total nitrate reduction
309 from the bare sediment in the spring. DNRA followed a similar pattern, with nitrification
310 accounting for more than 95% of the nitrate used for DNRA.
311 Potential nitrate reduction rates (D_{15} plus $DNRA_{15}$) (i.e. when nitrate was experimentally
312 added) exhibited similar patterns to ambient nitrate reduction (D_{14} plus $DNRA_{14}$) rates. Potential
313 denitrification (D_{15}) showed a significant interaction between site and season ($p<0.001$, Figure
314 3). For the oyster reef sediment D_{15} was significantly higher in the summer than the spring or
315 fall. This was also true for the bare sediment. The clam aquaculture site, D_{15} was highest in the
316 summer but not significantly different compared to the other seasons. During the summer, the
317 oyster reef sediment had the highest D_{15} (75.23 ± 18.51 μmol N m^{-2} hr^{-1}). This was also the
318 overall highest D_{15}. In the other seasons, the clam aquaculture site had the highest D_{15}. D_{15} was
319 higher than $DNRA_{15}$ for all sites and seasons (Figure 3). $DNRA_{15}$ was affected by both site
320 ($p=0.042$) and season ($p=0.034$). $DNRA_{15}$ was highest in the fall and lowest in the summer, with

spring having an intermediate rate. Mean $DNRA_{15}$ was higher (3.8 times) at the clam aquaculture site compared to the oyster reef sediment and 3.7 times higher than bare sediment, but the increase was only significant for the clam aquaculture site compared to the oyster reef sediment (Tukey HSD, p=0.04). $DNRA_{15}$ was not different at the clam aquaculture site (Tukey HSD, p=0.191) or oyster reef sediment (Tukey HSD, p=0.640) compared to the bare sediment. The highest D_{15} did not correspond to the highest $DNRA_{15}$.

D_{15} was higher than D_{14} ($p<0.001$) and $DNRA_{15}$ was higher than $DNRA_{14}$ ($p<0.001$). This indicates nitrate reduction increased with water column nitrate; however, the magnitude of this increase was site dependent. D_{15} significantly increased compared to D_{14} at the oyster reef sediment (Tukey HSD, $p<0.001$) and clam aquaculture site (Tukey HSD; p=0.005), while D_{14} and D_{15} at the bare site were not significantly different from each other (Tukey HSD, p=0.173). This indicates that denitrification was limited by nitrate at the bivalve sites but not at the bare site. The response of DNRA to nitrate addition was less than denitrification, with the largest increase observed at the clam aquaculture site. The clam aquaculture site was the only site where $DNRA_{15}$ was significantly higher than $DNRA_{14}$ (Tukey HSD, $p<0.001$).

Nutrient Fluxes

The largest efflux of both NH_4^+ and NO_x was observed from the clam aquaculture site, in the spring and in the summer, respectively (Table 4). NO_x fluxes were affected by site (p=0.006) and season (p=0.001) and the interaction was not significant (p=0.050). All sites had a positive NO_x flux during the summer, resulting in summer fluxes being different from the spring (Tukey HSD; p=0.009) and fall (Tukey HSD; p=0.009). NO_x fluxes were higher at the clam aquaculture site compared to the bare sediment (Tukey HSD; p=0.005) but the NO_x flux from the oyster reef

343 sediment was not significantly different compared to the bare sediment (Tukey HSD; p=0.072)
344 or the clam aquaculture site (Tukey HSD; p=0.42).

345 The interaction between site and season was significant for NH_4^+ fluxes (p<0.001). The
346 clam aquaculture site was the only site to have an efflux of NH_4^+during each season and rates
347 were not significantly different between seasons (Tukey HSD; p=0.081 (fall-spring), p=0.795
348 (fall-summer), p=0.247 (spring-summer). Oyster reef sediments had the highest NH_4^+ flux in the
349 summer compared to the spring (Tukey HSD; p<0.001) and fall (Tukey HSD; p<0.001). At the
350 bare site, NH_4^+ fluxes were positive in the summer and spring and significantly different from
351 the negative fluxes observed in the fall (Tukey HSD; p<0.001 & p<0.001, respectively). During
352 the summer NH_4^+ fluxes were similar between all the sites but during the spring and fall, NH_4^+
353 flux from the clam aquaculture site was significantly higher than either the oyster reef sediment
354 (Tukey HSD; p<0.001 (spring), p=0.001 (fall)) or bare sediment (Tukey HSD; p=0.007 (spring),
355 p<0001 (fall)). Calculated nitrification rates ranged from 1.54 ± 0.42 μmol N m^{-2} hr^{-1} at the bare
356 site during the fall to 18.25 ± 3.68 μmol N m^{-2} hr^{-1} in the oyster reef during the summer (Table
357 4). The interaction between site and season was significant for nitrification rates (p=0.03).
358 Nitrification rates were similar across all seasons for the clam aquaculture site and bare sediment
359 but higher in the summer compared to the spring (Tukey HSD; p=0.004) or fall (Tukey HSD;
360 p=0.002) for the oyster reef sediment. Nitrification was similar between the sites in the spring
361 but the clam aquaculture site was higher than the bare sediment (Tukey HSD; p=0.019) and
362 slightly higher than the oyster reef (Tukey HSD; p=0.048) in the fall, while nitrification at the
363 oyster reef was slightly higher than the bare sediment (Tukey HSD; p=0.005) and similar to the
364 clam aquaculture site (Tukey HSD; p=0.130) in the summer.

There were both seasonal ($p<0.001$) and site ($p=0.020$) differences in SOD, but the interaction was not significant ($p=0.105$). The lowest SOD (324.75 ± 23.19 μmol O_2 m^{-2} hr^{-1}) was measured from the bare sediment during the fall. The clam aquaculture site had the highest SOD (1883.69 ± 368.06 μmol O_2 m^{-2} hr^{-1}), measured during the spring (Table 4). SOD was higher in the oyster reef sediment and clam aquaculture site compared to the bare sediments (Table 4), however the only significant increase was between the bare sediment and the clam aquaculture site (Tukey HSD; $p=0.021$). SOD across all of the sites was significantly lower in the fall compared to the spring (Tukey HSD; $p<0.001$) and summer (Tukey HSD; $p<0.001$).

Correlations

SOD was significantly and positivity related to total ambient nitrate reduction ($p=0.017$), calculated nitrification ($p=0.019$), and NH_4^+ flux ($p=0.015$). SOD explained 21% of the variance in total ambient nitrate reduction, calculated as the sum of $DNRA_{14}$ plus D_{14}, 17% of the variance in calculated nitrification and 31% of the variance in NH_4^+ flux (Figure 4). SOM also explained 10% of the variance in $DNRA_{14}$ and 20% of the variance in D_{14} (Figure 5). D_{14} was significantly and positively correlated with SOM ($p=0.018$). $DNRA_{14}$ was positively correlated with SOM but this relationship was not significant ($p=0.107$).

DISCUSSION

Clam aquaculture and restored oyster reefs enhanced N transformations via their effect on sediment carbon and nitrification. N cycling pathways were related to SOD (Figure 4) and SOM controlled denitrification (Figure 5). These relationships indicate the importance of carbon availability as a predictor of nitrate reduction in the oligotrophic Smith Island Bay ecosystem (Eyre et al. 2013). While carbon availability contributes to the differences between the sites, the high percent of nitrate reduction, coupled to nitrification (both $DNRA_n$ and D_n were greater than

388 95% of $DNRA_{14}$ and D_{14}), indicates the importance of nitrification as a source of nitrate for all
389 sites.

390 Previous studies have attributed increased rates of N cycling processes in sediments
391 associated with bivalves to high organic matter loading due to biodeposition, high surface area of
392 the oyster reef substrate compared to bare sediment, bioturbation activity and gut and shell
393 biofilm communities and, when present, excretion by the animals themselves (Newell et al.
394 2002, Smyth et al. 2013a, Kellogg et al. 2014, Welsh et al. 2015). Given the higher SOM at the
395 clam aquaculture site and oyster reef sediment compared to the subtidal bare sediment,
396 biodeposition of labile organic carbon likely contributed to the observed enhanced
397 denitrification. In addition, because denitrification rates increased with added NO_3^- at the bivalve
398 sites but not at the bare sediment, we conclude NO_3^- also limited denitrification at bivalve sites.
399 In contrast, denitrification rates were not primarily limited by NO_3^- at the bare sediment
400 locations, which may be more strongly driven by organic C, anoxic conditions, or microbial
401 community structure.

402 While oyster reef sediments and clam aquaculture sites increase denitrification compared
403 to the bare sediment, the effect varied seasonally and was dependent on bivalve species. The
404 largest bivalve effect on denitrification was observed at the oyster reef sediment in the summer.
405 During the spring and fall, denitrification was enhanced more at the clam aquaculture site than
406 the oyster reef sediment. In contrast, there were no seasonal differences within the clam
407 aquaculture site, suggesting farm management practices may override seasonal effects. The
408 seasonal differences possibly associated with variation in the bivalve response to food
409 availability (i.e. filtration rates). In warmer months when seston is high, oysters tend to increase
410 biodeposit production while clams decrease clearance rates (Langdon and Newell 1989, Hawkins

411 et al. 1998, Newell et al. 2005). However, there were no clams inside the sediment cores in the
412 summer. Since denitrification associated with the clam shells, gut and gills, which can be very
413 important (Welsh and Castadelli 2004) was unaccounted for during this time and these
414 measurements may be an underestimate.

415 When assessing the role of bivalves for enhanced N removal through denitrification, it is
416 important to consider the magnitude of N removal versus N recycling. Overall, denitrification
417 efficiency ($D_{14}/(D_{14}+NO_x+NH_4^+)$) was 15%, 47% and 66%, in the clam aquaculture site, bare
418 sediment and oyster reef sediment, respectively. The lower efficiency at the clam aquaculture
419 site is associated with increased ammonium flux. DNRA accounted for less than 10% of the total
420 NH_4^+ efflux and is unlikely responsible for the enhanced ammonium flux. Rather, the high
421 ammonium efflux at the clam aquaculture site may be due to microbial mineralization as well as
422 direct excretion of NH_4^+ by the clams. The difference between the oyster reef sediment and clam
423 aquaculture site is due in part to the fact that live oysters were not included in oyster reef
424 samples. Denitrification efficiency at the oyster reef would likely be lower and nitrogen
425 regeneration would be higher if live oysters were included with the sediment, as excretion from
426 live oysters is also source of NH_4^+ (Smyth et al. 2013a, Caffrey et al. 2016).

427 Bivalves can have a disproportionally large effect on nitrogen cycling relative to the area.
428 For example, oyster reefs only occupy 2.7% of the area in Bogue Sound, North Carolina yet
429 remove 4% of the estimated annual nitrogen load through enhanced denitrification (Smyth et al.
430 2013b). Restoration of oyster reefs in Maryland could remove about half of the external nitrogen
431 inputs (Kellogg et al. 2013) and 26% of N inputs into Ninigret Pond, Rhode Island could be
432 removed if 5% of the estuary was used for oyster aquaculture (Humphries et al. 2016). However,
433 nitrogen regeneration associated with bivalves can also be disproportionately high relative to the

aerial coverage of the bivalves. Clam aquaculture occupies 3% of the subtidal bottom in Cherrystone Inlet, Virginia but nitrogen regeneration is equal to about half of the watershed nitrogen load (Murphy et al. 2016b). Similarly, at a mussel farm in Sacca di Goro, Italy, mussels covered only 5% of the area but excretion accounted for 25% of total DIN regeneration in the system (Nizzoli et al. 2011). Density is one of the main factors that determines whether bivalves are nitrogen sources or sinks at the ecosystem scale. High densities of bivalves tend to decrease denitrification and increase nitrogen regeneration due to excretion and organic matter loading to the sediments (Newell 2004). The exact density of bivalves that increases nitrogen regeneration and decreases denitrification depends on hydrodynamics and sediment quality. To extrapolate rates from experiments to the ecosystem scale assumes that processes scale linearly with density. We did not capture denitrification directly related to the oyster microbiome itself or account for oyster excretion. Therefore, extrapolating rates from our experiment may underestimate the overall impact of bivalves on N cycling. More work is needed to determine ways to evaluate the effect of bivalves on processes at the ecosystem scale.

In addition to direct modification of N pools and fluxes, bivalves can affect O_2 availability. The clam aquaculture site had the highest SOD, which coincided with a larger ammonium efflux. The correlation between SOD and ammonium production for bare sediment and oyster reef sediment reflects demand of oxygen to support nitrification and aerobic decomposition. When live clams are present, SOD also incorporates clam respiration (Murphy et al. 2016a). Thus, clams are major sinks for oxygen and sources of N, in addition to being sites for nitrification and denitrification (Welsh et al. 2015, Benelli et al. 2017).

As observed in other oligotrophic systems with low nutrients, biodeposition enhances denitrification. Biodeposits are a carbon source and accumulation on the sediment decreases the

457 oxygen penetration depth and diffusive pathway for water column nitrate, leading to increased
458 fluxes (Caffrey et al. 1993, Smyth et al. 2015). However, there is likely a threshold above which
459 the addition of organic matter from bivalves increases SOD and no longer stimulates
460 denitrification (Newell et al. 2005, Hoellein and Zarnoch 2014, Humphries et al. 2016). This
461 threshold depends on competition for oxygen, which limits nitrification rates and impacts nitrate
462 availability as well as denitrifier abundance and sediment redox conditions. Higher SOD is
463 indicative of increased organic matter oxidation or higher nitrification (Caffrey et al. 1993,
464 Fulweiler et al. 2008); however, a high SOD is also related to sediment anoxia, as the abiotic
465 oxidation of reduced compounds such as sulfide consume oxygen. Because of the relationship
466 between SOD and nitrate reduction, we conclude that carbon deposition is driving nitrate
467 reduction, particularly at the bivalve sites. The relationship between total nitrate reduction and
468 SOD reflects the complicated relationship between nitrate availability, O_2 and carbon for
469 controlling nitrate reduction.
470 As expected based on the environmental conditions of Smith Island (i.e. low water
471 column nitrate concentrations, high flow and an oxygenated water column), the majority (>95%)
472 of nitrate reduction measured in this study was supported by NO_3^- from nitrification (D_n and
473 $DNRA_n$, Figure 2). The bivalve sites had higher rates of nitrification than the bare sediment,
474 possibly due to organic matter loading and/or ample NH_4^+ supply. Although nitrification is a
475 chemolithoautotrophic process, which does not rely on organic matter, small amounts of organic
476 matter loading increase nitrification (Caffrey et al. 1993). The mineralization of this organic
477 matter serves as a source of NH_4^+ to the nitrifying community. However, high SOM, and
478 subsequent increase in SOD, can lead to oxygen limitation for nitrification and the highest rates
479 of nitrification occur at intermediate concentrations of both ammonium and oxygen (Caffrey et

al. 1993, Sloth et al. 1995, Blackburn 1996). Alternatively, carbon can affect nitrification if ammonium-oxidizing archaea (AOA), which are mixotrophs that require a carbon source, rather than ammonium-oxidizing bacteria (AOB).are responsible for nitrification (Qin et al. 2014).

The experimental addition of NO_3^- significantly enhanced denitrification rates in both the clam aquaculture site and oyster reef sediments (i.e. $D_{15} > D_{14}$). The increase in denitrification with increased NO_3^- in the water column suggests denitrification was NO_3^- -limited rather than carbon limited at both bivalve sediments. The increase in denitrification at the bivalve sites supports the hypothesis that bivalve biodeposition supplies organic carbon for NO_3^- reduction. At the bare sediment, both DNRA and denitrification were ultimately limited by carbon availability or potentially, microbial metabolic capacity since the experimental NO_3^- addition had no effect on denitrification or DNRA

Higher water column NO_3^- can alleviate the competition between DNRA and denitrification and enhance both processes (Roberts et al. 2012) (Koop-Jakobsen and Giblin 2010). However, in our study, the stimulatory effect of NO_3^- addition on DNRA was minimal and only observed at the clam aquaculture site (Figure 3). The fact that NO_3^- stimulated DNRA at the clam aquaculture site may be due to differences in the microbial community structure compared to bare sediment. In a nearby tributary, clam aquaculture sediments had significantly higher abundances of DNRA communities compared with bare sediments based on the quantification of cytochrome C nitrite reductase genes (*nrfA*) (Murphy et al. 2016a). But, despite the increase in DNRA at the clam aquaculture site, the relative importance of DNRA to total NO_3^- reduction was unaffected by the added NO_3^-. A similar trend has been observed in salt marsh ecosystems (Koop-Jakobsen and Giblin 2010) where the addition of nitrate alleviated competition between DNRA and denitrification for nitrate from nitrification and allowed both

503 processes to increase. Yet, when water column nitrate increased, the ratio of carbon to NO_3^-
504 shifted because of increased NO_3^-, conditions which favor denitrification resulting in higher
505 denitrification than DNRA(Burgin and Hamilton 2007). The NO_3^- addition lowered the ratio of
506 organic carbon to NO_3^-, resulting in more energetically favorable conditions for denitrification
507 compared to DNRA (Burgin and Hamilton 2007).
508 Based on recent studies examining N cycling in clam aquaculture sites and oyster habitats in a
509 tributary of Chesapeake Bay that show that DNRA exceeds denitrification (Murphy et al. 2016a,
510 Lunstrum et al. 2017), it was expected that biodeposition would favor DNRA over
511 denitrification. However, this was not observed in our study; rather, we found denitrification
512 exceeded DNRA and rates of DNRA were lower than those reported previously. Moreover,
513 unlike other studies, DNRA was not affected by clam aquaculture or oysters although slightly
514 higher DNRA was observed in the clam aquaculture site compared to the oyster reef sediment.
515 The availability of nitrate from nitrification maintains conditions that favor denitrification
516 despite the higher SOM at the at clam aquaculture site and restored oyster reefs (Tiedje 1988,
517 Burgin and Hamilton 2007). Differences between our data and previous published studies
518 highlight the fact that the effects of bivalves on sediment biogeochemistry is site specific. More
519 studies are necessary to fully understand how environmental conditions drive the effect of
520 bivalves on N dynamics.
521 The inclusion of live bivalves in sediment cores can drastically alter benthic nitrogen
522 cycling (Kellogg et al. 2013, Smyth et al. 2013a, Turek and Hoellein 2015, Humphries et al.
523 2016, Murphy et al. 2016a). While there are advantages to including live bivalves in experiment
524 chambers, especially oysters, which build complex reef ecosystems, it is logistically challenging
525 (Humphries et al. 2016). Our study focused on how oyster reefs and clam aquaculture affect N

526 dynamics at the sediment-water interface. By sampling sediments from the oyster reef and
527 omitting the oyster reef itself, we may underestimate the effects of oyster reef ecosystems. The
528 random sampling of clam aquaculture sites resulted in an unequal number of clams in each core.
529 The clams were not removed to prevent altering the natural biogeochemical gradients in the
530 sediments, which are critical to maintain realistic conditions during the incubations. By chance
531 no clams were contained in the summer sediment cores. To account for this, we analyzed our
532 data using a mixed effects model, which allows for variation due to the number of bivavles
533 present. While the current study used sediments from clam aquaculture sites and restored oyster
534 reefs to investigate the effects of these bivalves on sediment N cycling, studies wishing to
535 compare bivalves would benefit from including a known number of bivalves in experimental
536 chambers.
537
538 **Conclusion**
539 Understanding how clam aquaculture and oyster reefs affect sediment N cycling is
540 important when assessing the use of bivalves as management tools for controlling eutrophication.
541 While oyster reefs and clam aquaculture can enhance N removal compared to reference
542 sediments, certain conditions may result in net N regeneration. Clam aquaculture, which utilize a
543 predator exclusion net in ecosystems with low water column nitrate and short residence time may
544 be a source of new N from mineralization and excretion (Murphy et al. 2016b). The high efflux
545 of NH_4^+ at the clam aquaculture site is due in part to the presence of live bivalves in the sediment
546 cores, indicating the direct control that infaunal bivalves and their associated microbiota have on
547 overall benthic flux and N cycle process rates. Conversion of bare sediment to oyster reefs,
548 would, likely have a similar effect, where oyster reefs enhance sediment denitrification relative

549 to bare sediment and where the shell can be colonized by denitrifying bacteria but the addition of
550 live oysters to the ecosystem may increase ammonium regeneration through excretion and N
551 regeneration (Kellogg et al. 2013, Smyth et al. 2013a). Factors such as location, hydrodynamics,
552 species, growing conditions and sediment redox condition likely affect whether the addition of
553 bivalves will remove or enhance N. As N loading and bivalve aquaculture and oyster reef
554 restoration continue to expand in shallow coastal ecosystems, understanding what conditions
555 yield removal of N compared to recycling is important for determining the efficacy of bivalves in
556 controlling N pollution.
557

558 **ACKNOWLEDGEMENTS**

559 We thank Edward Smith, Sean Fate, Hunter Walker, Martha Murphy, Ann Arfkin, Jennifer
560 Stanhope, Cay Thompson and PG Ross for field and laboratory assistance. This research was
561 financially supported by the National Science Foundation (NSF-OCE-1321373), Virginia Coast
562 Reserve Long Term Ecological Research (NSF-DEB-1237733, 0621014), and the David H.
563 Smith Conservation Research Postdoctoral Fellowship. Comments from Tim Hoellein and three
564 anonymous reviewers greatly improved this manuscript. This is contribution #XXXX from the
565 Virginia Institute of Marine Science.

References

An, S., and S. B. Joye. 2001. Enhancement of coupled nitrification-denitrification by benthic photosynthesis in shallow estuarine sediments. Limnology and Oceanography 46:62–74.

An, S., and W. S. Gardner. 2002. Dissimilatory nitrate reduction to ammonium (DNRA) as a nitrogen link, versus denitrification as a sink in a shallow estuary (Laguna Madre/Baffin Bay, Texas). Marine Ecology Progress Series 237:41–50.

An, S., W. S. Gardner, and T. Kana. 2001. Simultaneous Measurement of Denitrification and Nitrogen Fixation Using Isotope Pairing with Membrane Inlet Mass Spectrometry Analysis. Applied and Environmental Microbiology 67:1171–1178.

Beck, M. W., R. D. Brumbaugh, L. Airoldi, A. Carranza, L. D. Coen, C. Crawford, O. Defeo, G. J. Edgar, B. Hancock, M. C. Kay, H. S. Lenihan, M. W. Luckenbach, C. L. Toropova, G. Zhang, and X. Guo. 2011. Oyster Reefs at Risk and Recommendations for Conservation, Restoration, and Management. BioScience 61:107–116.

Benelli, S., M. Bartoli, E. Racchetti, P. C. Moraes, M. Zilius, I. Lubiene, and E. A. Fano. 2017. Rare but large bivalves alter benthic respiration and nutrient recycling in riverine sediments. Aquatic Ecology 51:1–16.

Blackburn, T. H. 1996. Nitrogen gas flux from sediments: insights from simulation modelling. Aquatic Microbial Ecology 10:209–211.

Bricker, S. B., K. C. Rice, and O. P. Bricker. 2014. From Headwaters to Coast: Influence of Human Activities on Water Quality of the Potomac River Estuary. Aquatic Geochemistry 20:291–323.

Bruesewitz, D. A., W. S. Gardner, R. F. Mooney, L. Pollard, and E. J. Buskey. 2013. Estuarine ecosystem function response to flood and drought in a shallow, semiarid estuary: Nitrogen cycling and ecosystem metabolism. Limnology and Oceanography 58:2293–2309.

Burgin, A. J., and S. K. Hamilton. 2007. Have we overemphasized the role of denitrification in aquatic ecosystems? A review of nitrate removal pathways. Frontiers in Ecology and the Environment 5:89–96.

Caffrey, J. M., J. T. Hollibaugh, and B. Mortazavi. 2016. Living oysters and their shells as sites of nitrification and denitrification. Marine Pollution Bulletin 112:86–90.

Caffrey, J. M., N. P. Sloth, H. F. Kaspar, and T. H. Blackburn. 1993. Effect of organic loading on nitrification and denitrification in a marine sediment microcosm. FEMS Microbiology Ecology 12:159–167.

Cline, J. D. 1969. Spectrophotometric Determination of Hydrogen Sulfide in Natural Waters. Limnology and Oceanography 14:454–458.

600 Dame, R. F., R. G. Zingmark, and E. Haskin. 1984. Oyster reefs as processors of estuarine
601 materials. Journal of Experimental Marine Biology and Ecology 83:239–247.

602 Diaz, R. J., and R. Rosenberg. 2008. Spreading Dead Zones and Consequences for Marine
603 Ecosystems. Science (New York, N.Y.) 321:926–929.

604 Emery, K. A. 2015. Coastal bivalve aquaculture carbon cycling, spatial distribution and resource
605 use in Virginia, USA and Baja California, Mexico.

606 Erler, D. V., D. T. Welsh, W. W. Bennet, T. Meziane, C. Hubas, D. Nizzoli, and A. J. P.
607 Ferguson. 2017. The impact of suspended oyster farming on nitrogen cycling and nitrous
608 oxide production in a sub-tropical Australian estuary. Estuarine, Coastal and Shelf Science
609 192:117–127.

610 Eyre, B. D., D. T. Maher, and P. Squire. 2013. Quantity and quality of organic matter (detritus)
611 drives N_2 effluxes (net denitrification) across seasons, benthic habitats, and estuaries. Global
612 Biogeochemical Cycles 27:1083–1095.

613 Fulweiler, R. W., S. W. Nixon, B. A. Buckley, and S. L. Granger. 2008. Net Sediment N_2 Fluxes
614 in a Coastal Marine System Experimental Manipulations and a Conceptual Model.
615 Ecosystems 11:1168–1180.

616 Gardner, W. S., and M. J. McCarthy. 2009. Nitrogen dynamics at the sediment water interface in
617 shallow, sub-tropical Florida Bay: why denitrification efficiency may decrease with
618 increased eutrophication. Biogeochemistry 95:185–198.

619 Hardison, A. K., C. K. Algar, A. E. Giblin, and J. J. Rich. 2015. Influence of organic carbon and
620 nitrate loading on partitioning between dissimilatory nitrate reduction to ammonium
621 (DNRA) and N_2 production. Geochimica et Cosmochimica Acta 164:146–160.

622 Hauxwell, J., J. Cebri n, C. Furlong, and I. Valiela. 2001. Macroalgal canopies contribute to
623 eelgrass (*Zostera marina*) decline in temperate estuarine ecosystems. Ecology 82:1007–
624 1022.

625 Hawkins, A., R. Smith, S. H. Tan, and Z. B. Yasin. 1998. Suspension-feeding behavior in
626 tropical bivalve molluscs: *Perna viridis, Crassostrea belcheri, Crassostrea iradelei*,
627 *Saccostrea cucculata* and *Pinctada margarifera*. Marine Ecology Progress … 166:173–185.

628 Hoellein, T. J., and C. B. Zarnoch. 2014. Effect of eastern oysters (*Crassostrea virginica*) on
629 sediment carbon and nitrogen dynamics in an urban estuary. Ecological Applications
630 24:271–286.

631 Humphries, A. T., S. G. Ayvazian, J. C. Carey, B. T. Hancock, S. Grabbert, D. Cobb, C. J.
632 Strobel, and R. W. Fulweiler. 2016. Directly Measured Denitrification Reveals Oyster
633 Aquaculture and Restored Oyster Reefs Remove Nitrogen at Comparable High Rates.
634 Frontiers in Marine Science 3:107–10.

635 Kana, T. M., C. Darkangelo, M. D. Hunt, and J. B. Oldham. 1994. Membrane inlet mass
636 spectrometer for rapid high-precision determination of N2, O2, and Ar in environmental
637 water samples. Analytical Chemistry 66:4166–4170.

638 Kellogg, M. L., A. R. Smyth, M. W. Luckenbach, R. H. Carmichael, B. L. Brown, J. C.
639 Cornwell, M. F. Piehler, M. S. Owens, D. J. Dalrymple, and C. B. Higgins. 2014. Use of
640 oysters to mitigate eutrophication in coastal waters. Estuarine, Coastal and Shelf Science
641 151:156–168.

642 Kellogg, M. L., J. C. Cornwell, M. S. Owens, and K. T. Paynter. 2013. Denitrification and
643 nutrient assimilation on a restored oyster reef. Marine Ecology Progress Series 480:1–19.

644 Koop-Jakobsen, K., and A. E. Giblin. 2010. The effect of increased nitrate loading on nitrate
645 reduction via denitrification and DNRA in salt marsh sediments. Limnology and
646 Oceanography 55:789–802.

647 Langdon, C. J., and R. Newell. 1989. Utilization of detritus and bacteria as food sources by two
648 bivalve suspension-feeders, the oyster *Crassostrea virginica* and the mussel *Geukensia*
649 *demissa*. Marine Ecology Progress Series 58:299–310.

650 Lorenzen, C. J. 1967. Determination of chlorophyll and pheo-pigments: spectrophotometric
651 equations. Limnology and Oceanography 12:343–346.

652 Lunstrum, A., and L. R. Aoki. 2016. Oxygen interference with membrane inlet mass
653 spectrometry may overestimate denitrification rates calculated with the isotope pairing
654 technique. Limnology and Oceanography: Methods 14:425–431.

655 Lunstrum, A., K. McGlathery, and A. Smyth. 2017. Oyster (*Crassostrea virginica*) aquaculture
656 shifts sediment nitrogen processes toward mineralization over denitrification. Estuaries and
657 Coasts. Accepted. 1–17.

658 Murphy, A. E., I. C. Anderson, A. R. Smyth, B. Song, and M. W. Luckenbach. 2016a. Microbial
659 nitrogen processing in hard clam (*Mercenaria mercenaria*) aquaculture sediments: the
660 relative importance of denitrification and dissimilatory nitrate reduction to ammonium
661 (DNRA). Limnology and Oceanography 61:1589–1604.

662 Murphy, A. E., I. C. Anderson, and M. W. Luckenbach. 2015. Enhanced nutrient regeneration at
663 commercial hard clam (*Mercenaria mercenaria*) beds and the role of macroalgae. Marine
664 Ecology Progress Series 530:135–151.

665 Murphy, A. E., K. A. Emery, I. C. Anderson, M. L. Pace, M. J. Brush, and J. E. Rheuban. 2016b.
666 Quantifying the Effects of Commercial Clam Aquaculture on C and N Cycling: an Integrated
667 Ecosystem Approach. Estuaries and Coasts 39:1746–1761.

668 Newell, R. 2004. Ecosystem influences of natural and cultivated populations of suspension-
669 feeding bivalve molluscs: a review. Journal of Shellfish Research 23:51–61.

Newell, R. I. E., J. C. Cornwell, and M. S. Owens. 2002. Influence of simulated bivalve biodeposition and microphytobenthos on sediment nitrogen dynamics: A laboratory study. Limnology and Oceanography 47:1367–1379.

Newell, R., T. R. Fisher, and R. R. Holyoke. 2005. Influence of eastern oysters on nitrogen and phosphorus regeneration in Chesapeake Bay, USA. *in* R. F. Dame and S. Olenin, editors. The Comparative Roles of Suspension-Feeders in Ecosystems.

Nielsen, L. 1992. Denitrification in sediment determined from nitrogen isotope pairing. FEMS Microbiology Letters 86:357–362.

Nizzoli, D., D. T. Welsh, and P. Viaroli. 2011. Seasonal nitrogen and phosphorus dynamics during benthic clam and suspended mussel cultivation. Marine Pollution Bulletin 62:1276–1287.

Nizzoli, D., D. T. Welsh, E. A. Fano, and P. Viaroli. 2006. Impact of clam and mussel farming on benthic metabolism and nitrogen cycling, with emphasis on nitrate reduction pathways. Marine Ecology Progress Series 315:151–165.

Paerl, H. W. 1997. Coastal eutrophication and harmful algal blooms: Importance of atmospheric deposition and groundwater as "new" nitrogen and other nutrient sources. Limnology and Oceanography 42:1154–1165.

Pelegrí, S. P., and T. H. Blackburn. 1994. Bioturbation effects of the amphipod *Corophium volutator* on microbial nitrogen transformations in marine sediments. Marine Biology 121:253–258.

Piehler, M. F., and A. R. Smyth. 2011. Habitat-specific distinctions in estuarine denitrification affect both ecosystem function and services. Ecosphere 2:art12–16.

Pinckney, J., R. Papa, and R. Zingmark. 1994. Comparison of high-performance liquid chromatographic, spectrophotometric, and fluorometric methods for determining chlorophyll a concentrations in estuarine sediments. Journal of Microbiological Methods 19:59–66.

Qin, W., S. A. Amin, W. Martens-Habbena, C. B. Walker, H. Urakawa, A. H. Devol, A. E. Ingalls, J. W. Moffett, E. V. Armbrust, and D. A. Stahl. 2014. Marine ammonia-oxidizing archaeal isolates display obligate mixotrophy and wide ecotypic variation. Proceedings of the National Academy of Sciences 111:12504–12509.

Reeburgh, W. S. 1969. Observations of Gases in Chesapeake Bay Sediments. Limnology and Oceanography 14:368–375.

Risgaard-Petersen, N., N. P. Revsbech, and S. Rysgaard. 1995. Combined Microdiffusion-Hypobromite Oxidation Method for Determining Nitrogen-15 Isotope in Ammonium. Soil Science Society of America Journal 59:1077–1080.

704 Roberts, K. L., V. M. Eate, B. D. Eyre, D. P. Holland, and P. L. M. Cook. 2012. Hypoxic events
705 stimulate nitrogen recycling in a shallow salt-wedge estuary: The Yarra River estuary,
706 Australia. Limnology and Oceanography 57:1427–1442.

707 Rose, J. M., S. B. Bricker, and J. G. Ferreira. 2014. Comparative analysis of modeled nitrogen
708 removal by shellfish farms. Marine Pollution Bulletin 91:1–6.

709 Safak, I., P. L. Wiberg, D. L. Richardson, and M. O. Kurum. 2015. Controls on residence time
710 and exchange in a system of shallow coastal bays. Continental Shelf Research 97:7–20.

711 Secrist, R. G. 2013. Food availability and utilization for cultured hard clams. Thesis. The College
712 of William and Mary, Gloucester Point, VA, USA

713 Sloth, N. P., H. Blackburn, L. S. Hansen, N. Risgaard-Petersen, and B. A. Lomstein. 1995.
714 Nitrogen cycling in sediments with different organic loading. Oceanographic Literature
715 Review 9:733.

716 Smyth, A. R., M. F. Piehler, and J. H. Grabowski. 2015. Habitat context influences nitrogen
717 removal by restored oyster reefs. Journal of Applied Ecology 52:716–725.

718 Smyth, A. R., N. R. Geraldi, and M. F. Piehler. 2013a. Oyster-mediated benthic-pelagic coupling
719 modifies nitrogen pools and processes. Marine Ecology Progress Series 493:23–30.

720 Smyth, A. R., N. R. Geraldi, S. P. Thompson, and M. F. Piehler. 2016. Biological activity
721 exceeds biogenic structure in influencing sediment nitrogen cycling in experimental oyster
722 reefs. Marine Ecology Progress Series 560:173–183.

723 Smyth, A. R., S. P. Thompson, K. N. Siporin, W. S. Gardner, M. J. McCarthy, and M. F. Piehler.
724 2013b. Assessing nitrogen dynamics throughout the estuarine landscape. Estuaries and
725 Coasts 36:44–55.

726 Stief, P. 2013. Stimulation of microbial nitrogen cycling in aquatic ecosystems by benthic
727 macrofauna: mechanisms and environmental implications. Biogeosciences 10:7829–7846.

728 Tiedje, J. M. 1988. Ecology of denitrification and dissimilatory nitrate reduction to ammonium.
729 Biology of anaerobic microorganisms.

730 Turek, K. A., and T. J. Hoellein. 2015. The invasive Asian clam (Corbicula fluminea) increases
731 sediment denitrification and ammonium flux in 2 streams in the midwestern USA.
732 Freshwater Science 34:472–484.

733 Welsh, D. T. 2003. It's a dirty job but someone has to do it: The role of marine benthic
734 macrofauna in organic matter turnover and nutrient recycling to the water column.
735 Chemistry and Ecology 19:321–342.

736 Welsh, D. T., and G. Castadelli. 2004. Bacterial nitrification activity directly associated with
737 isolated benthic marine animals. Marine Biology 144:1029–1037.

738 Welsh, D. T., D. Nizzoli, E. A. Fano, and P. Viaroli. 2015. Direct contribution of clams
739 (Ruditapes philippinarum) to benthic fluxes, nitrification, denitrification and nitrous oxide
740 emission in a farmed sediment. Estuarine, Coastal and Shelf Science 154:84–93.

741 Yin, G., L. Hou, M. Liu, Z. Liu, and W. S. Gardner. 2014. A Novel Membrane Inlet Mass
742 Spectrometer Method to Measure 15NH 4+for Isotope-Enrichment Experiments in Aquatic
743 Ecosystems. Environmental Science & Technology 48:9555–9562.

744

Figure Captions

Figure 1: Image of Smith Island Bay, Chesapeake Bay, US. Locations of clam beds (triangle), oyster reefs (square) and bare sediment (circle) are identified.

Figure 2: Actual denitrification (D_{14}) (A) and DNRA ($DNRA_{14}$) (B) rates for each site and season. Direct denitrification (D_w) or DNRA ($DNRA_w$) are in black and denitrification or DNRA coupled to nitrification (D_n, or $DNRA_n$) is presented in gray. Error bars are on standard error of the mean (n=3).

Figure 3: Potential denitrification (D_{15}) (A) and DNRA ($DNRA_{15}$) (B) rates for each site and season. Error bars are on standard error of the mean (n=3).

Figure 4: Regression of Total Ambient NO_3^- reduction (A), nitrification (B) and NH_4^+ flux (C) against SOD. Pearson correlation coefficient, regression equation, r^2, and p-value for the relationship, which includes all the data for each site and season, and are included on each graph. Samples from the clam bed are identified as triangles, oyster reef sediments are squares and bare sediment are circles.

Figure 5: Regression of D_{14} (A) and $DNRA_{14}$ (B) against sediment organic matter (SOM). Pearson correlation coefficient, regression equation, r^2, and p-value for the relationship, which includes all the data for each site and season, and are included on each graph. Samples from the clam bed are identified as triangles, oyster reef sediments are squares and bare sediment are circles.

Figure 1

Click here to download Figure Fig 1.jpg

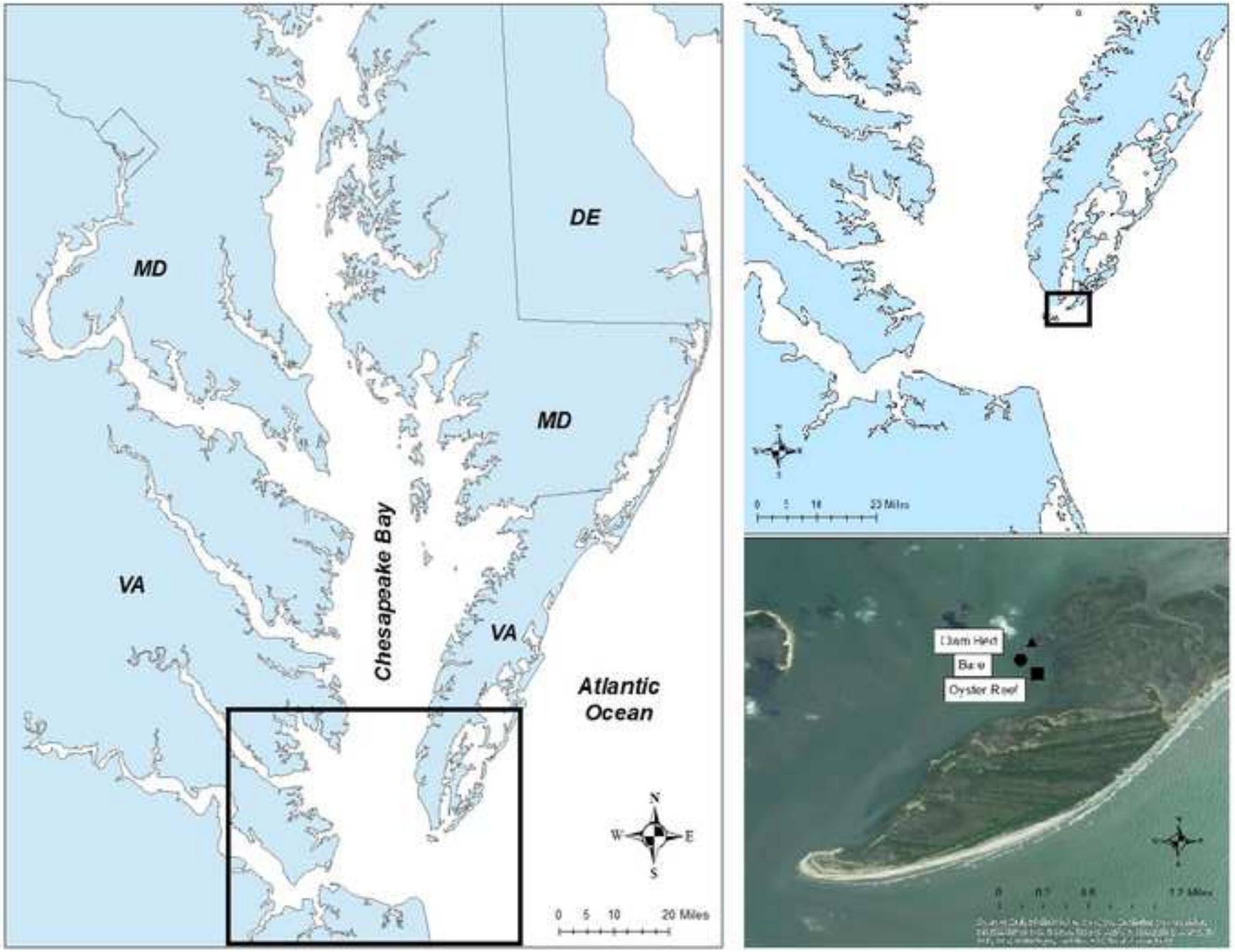


Figure 2

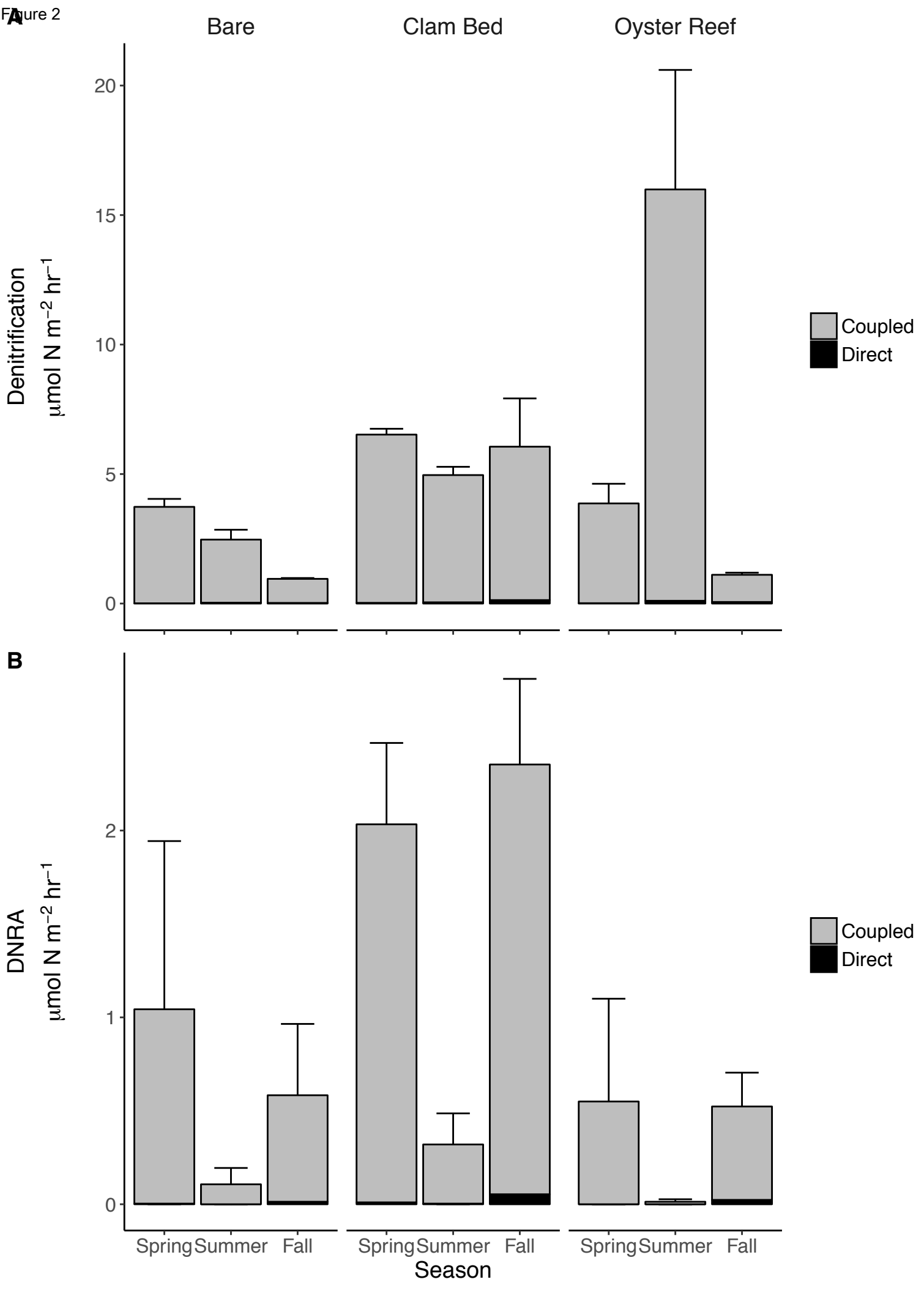

A
Bare
Clam Bed
Oyster Reef
Denitrification
µmol N m−2 hr−1
20
15
10
5
0
Coupled
Direct
B
DNRA
µmol N m−2 hr−1
2
1
0
Coupled
Direct
Spring
Summer
Fall
Season

Figure 3

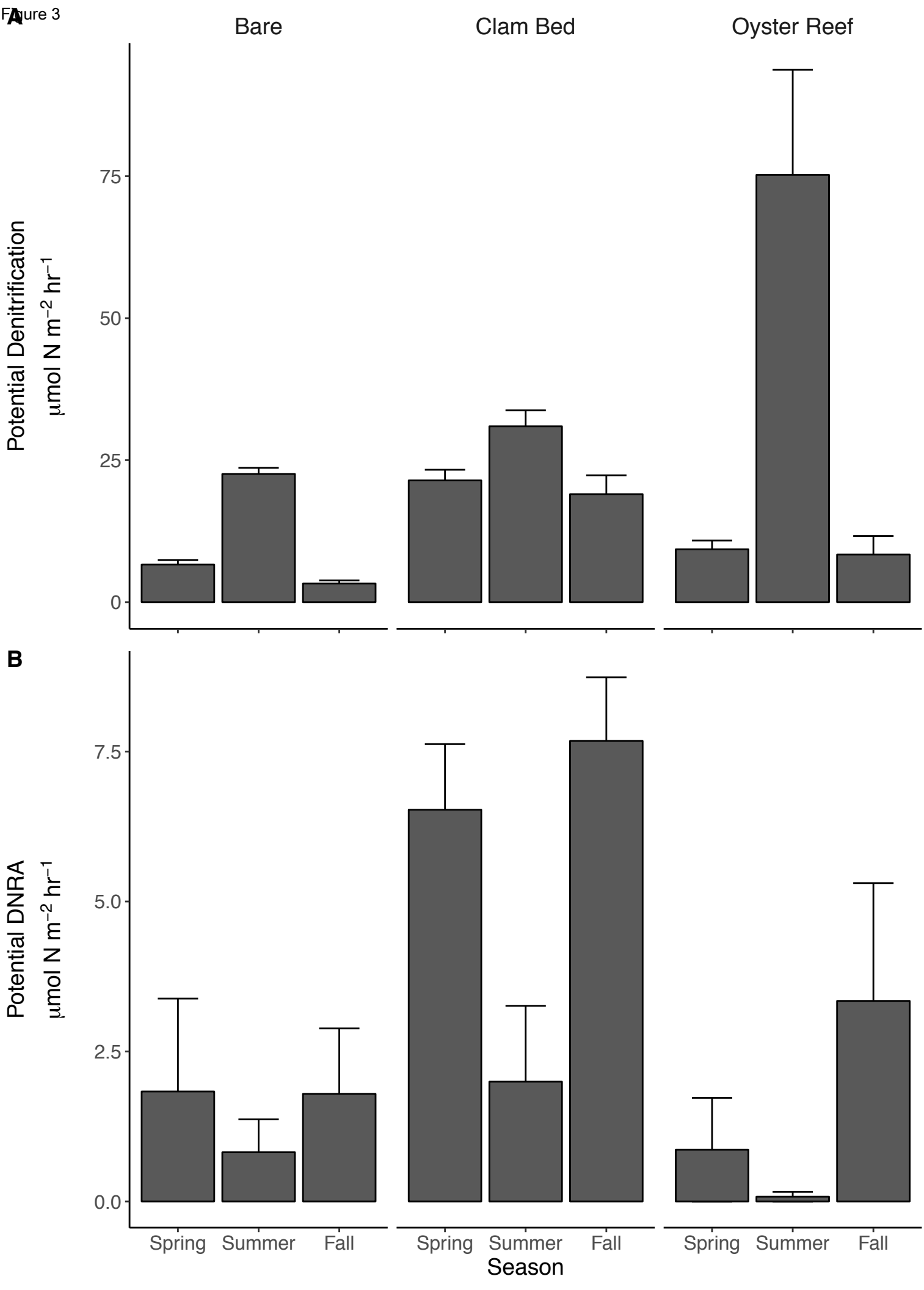

A
Bare
Clam Bed
Oyster Reef
Potential Denitrification
μmol N m−2 hr−1
75
50
25
0
B
Potential DNRA
μmol N m−2 hr−1
7.5
5.0
2.5
0.0
Spring
Summer
Fall
Season

Figure 4

Click here to download Figure Fig 4.jpg

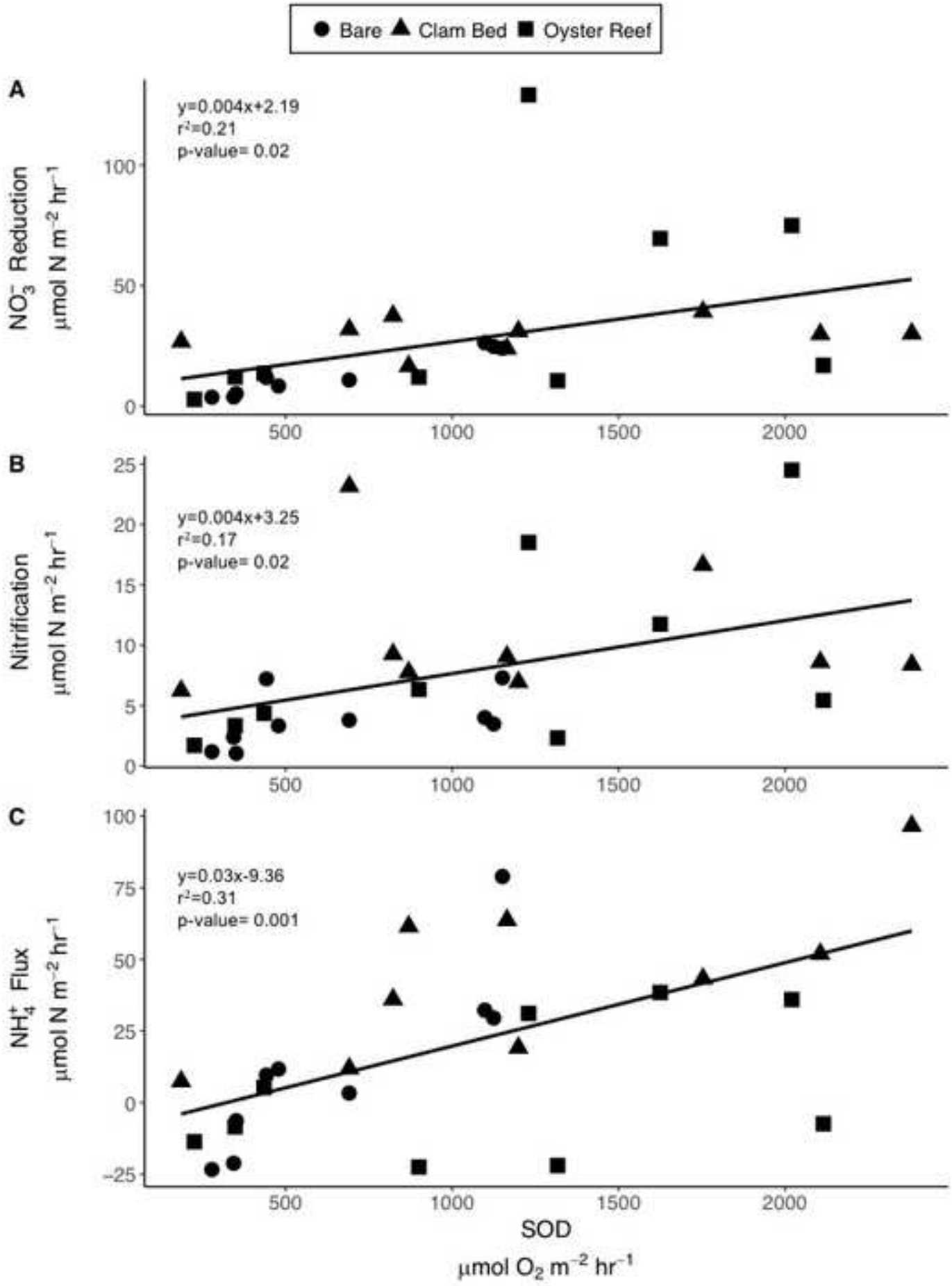
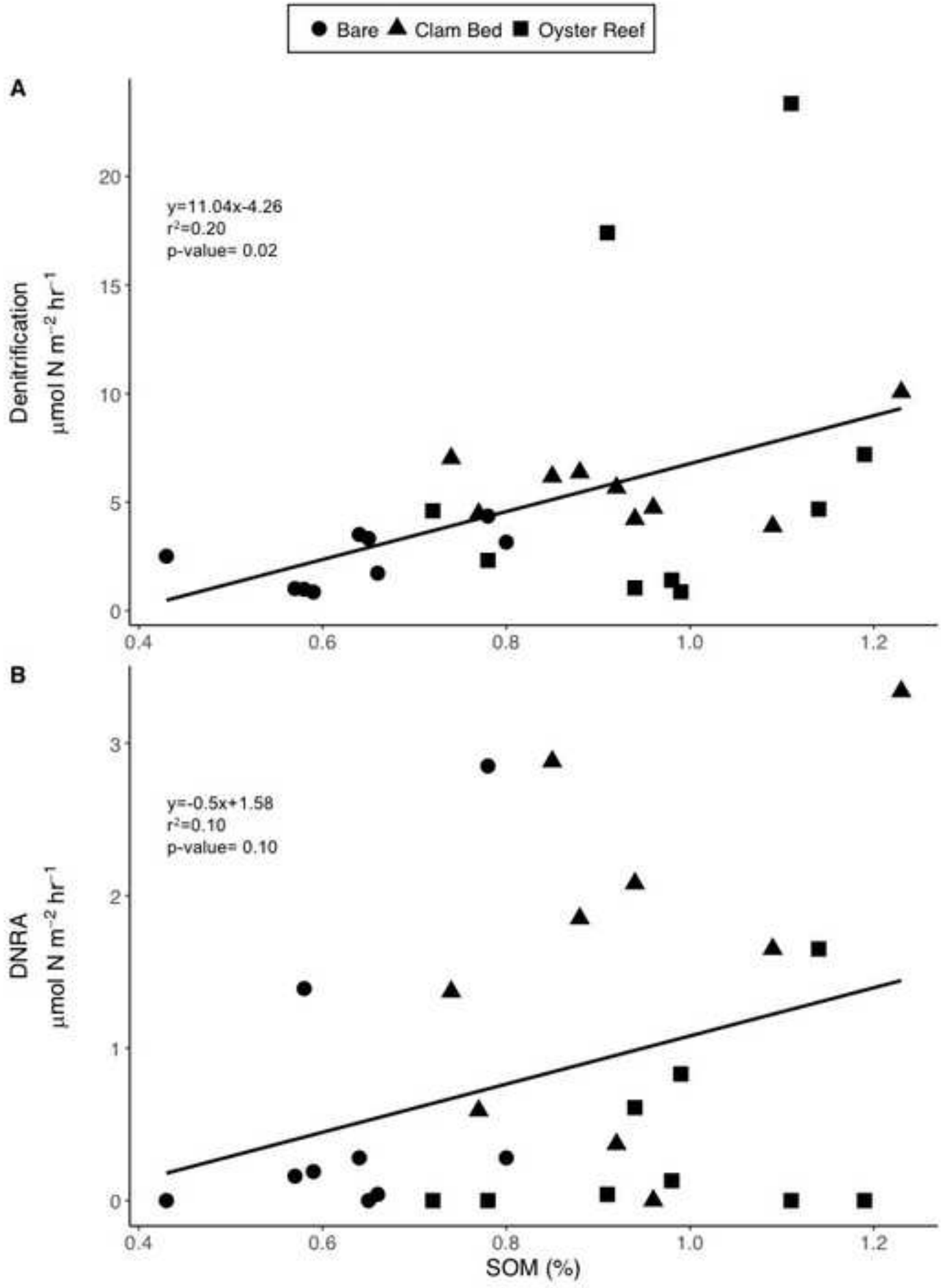


Figure 5

Click here to download Figure Fig 5.jpg



Tables

Table 1: Reported rates of denitrification and DNRA measured from the eastern oyster (*Crassostrea virginica)* or hard clam (*Mercenaria mercenaria)* ecosystems using either the N_2:Ar method or Isotope Pairing Technique (IPT).

Bivalve Species	Context	Denitrification (μmol m^{-2} hr^{-1})	DNRA (μmol m^{-2} hr^{-1})	Denitrification increased above Bare Sediment Site?	DNRA increased above Bare Sediment Site?	Location	Methods	Reference
Mercenaria mercenaria **(hard clam)**	Aquaculture	0.77 - 2.9	2.7 - 14.2	Only in the Fall	Yes, all seasons	Shallow polyhaline sediments (Cherrystone Inlet, VA)	Whole core batch incubation; isotope pairing technique	Murphy et al., 2016
Mercenaria mercenaria **(hard clam)**	Aquaculture	4.9 - 6.5	0.32 - 2.36	Yes	No	Shallow coastal bay (Smith Island Bay, VA)	Whole core continuous-flow incubation; isotope pairing technique	This Study
Crassostrea virginica **(eastern oyster)**	Aquaculture	0 - 65	---	No	---	Shallow mesohaline sediments below floats (St. Jerome Creek and Spencer Creek. VA)	Whole core continuous-flow incubation; N_2:Ar Technique	Higgins et al., 2013
Crassostrea virginica **(eastern oyster)**	Aquaculture	0 - 1097	---	Yes	---	Back barrier lagoon (Ninigret Pond, RI)	Whole chamber batch incubations; N_2:Ar Technique	Humphries et al., 2016
Crassostrea virginica **(eastern oyster)**	Aquaculture	-4.7 – 12.1	---	No	---	Shallow subtropical estuary below floats (Mobile Bay, AL)	Whole core continuous-flow incubation; N_2:Ar	Mortazavi et al. 2015
Crassostrea virginica **(eastern oyster)**	Aquaculture	<1—19.2	<1 to 40.3	Yes	Yes	Shallow polyhaline sediments (Cherrystone Inlet, VA)	Whole core batch incubation; isotope pairing technique	Lunstrum et al. 2017
Crassostrea virginica **(eastern oyster)**	Restored Reef	250 -1590	---	Yes	---	Mesohaline subtidal reef (Choptank River, MD)	Whole chamber batch incubations; N_2:Ar Technique	Kellogg et al., 2013

***Crassostrea virginica* (eastern oyster)**	Restored Reef	0 - 332	0.8 - 104	Yes	Yes, in the summer	Intertidal sediments adjacent to reef (Bogue Sound, NC)	Whole core continuous-flow incubation; N_2:Ar; then $^{15}NO_3^-$ for **potential** DNRA measurements	Smyth et al., 2013b
***Crassostrea virginica* (eastern oyster)**	Restored Reef	0 - 1803	---	Yes	---	Back barrier lagoon (Ninigret Pond, RI)	Whole chamber batch incubations; N_2:Ar Technique	Humphries et al. 2016
***Crassostrea virginica* (eastern oyster)**	Restored Reef	1.1 - 16.0	0.01 - 0.55	Yes	No	Shallow coastal bay (Smith Island Bay, VA)	Whole core continuous-flow incubation; isotope pairing technique	This Study
***Crassostrea virginica* (eastern oyster)**	Natural Reefs	10-30	---	Yes, enriched location only	---	Intertidal sediments adjacent to reef in an enriched and reference streams (Great Bay Estuary, NH)	Whole core continuous-flow incubation with $^{15}NO_3^-$ addition	Hoellein et al. 2015

Table 2: In situ water properties at each sampling date. Mean and standard error (n=3) are presented for water column nutrients. BD=Below Detection.

Season	Date	Temp (°C)	Salinity	Dissolved Oxygen (mg/l) [O_2%]	NO_x (µM)	NH_4^+ (µM)
Spring	24-Apr-14	15.6	33	8.05 [98.1%]	BD	3.06 ±0.04
Summer	23-Jun-14	25	30.5	6.36 [91.5%]	BD	0.46±0.01
Fall	5-Nov-15	14	31.7	8.28 [96.8%]	0.72 ±0.01	1.76±0.04

Table 3: Seasonal sediment characteristics for each site and sampling date for all parameters. H_2S in the clam bed in spring, bare sediment in summer and oyster reef in fall only had one sample above the detection limit. Data are mean ± standard error (SE) for all parameters. NA for SE indicates that only one sample was above the detection limit. Letters indicate significant differences between the means.

		SOM (%)	Benthic Chl ($\mu g/cm^2$)	H_2S (μM)	Porewater NO_X (μM)	Porewater NH_4^+ (μM)
Spring	Bare	0.69 ± 0.05	0.41 ± 0.03	25.74 ± 5.55	0.62 ± 0.18	70.73 ± 12.67
	Clam Bed	0.82 ± 0.04	0.80 ± 0.08	2.20 ± NA	0.39 ± 0.07	187.89 ± 25.82
	Oyster Reef	0.88 ± 0.13	0.74 ± 0.07	33.97 ± 11.04	0.36 ± 0.02	24.34 ± 10.12
Summer	Bare	0.63 ± 0.11	2.11 ± 0.19	10.89 ± NA	0.56 ± 0.04	51.40 ± 11.29
	Clam Bed	0.88 ± 0.06	1.47 ± 0.09	265.81 ± 99.27	0.92 ± 0.22	184.76 ± 28.30
	Oyster Reef	1.07 ± 0.08	1.30 ± 0.52	228.20 ± 115.04	0.65 ± 0.06	121.59 ± 67.31
Fall	Bare	0.58 ± 0.01	5.54 ± 0.45	11.11 ± 3.59	0.49 ± 0.09	52.34 ± 14.48
	Clam Bed	1.09 ± 0.09	7.37 ± 0.43	53.28 ± 17.68	0.96 ± 0.60	102.98 ± 12.98
	Oyster Reef	0.97 ± 0.02	3.69 ± 0.19	4.72 ± NA	0.50 ± 0.03	42.63 ± 13.96
Mean	Bare	0.63 ± 0.04	2.69 ± 0.77	15.95 ± 3.90	0.56 ± 0.06	58.16 ± 7.17
	Clam Bed	0.93 ± 0.05	3.22 ± 1.05	137.07 ± 59.76	0.76 ± 0.21	158.5 ± 18.15
	Oyster Reef	0.97 ± 0.05	1.91 ± 0.48	93.84 ± 56.34	0.52 ± 0.05	67.67 ± 26.13

Table 4: Seasonal mean (n=3) fluxes of ammonium (NH_4^+), nitrate+nitrite (NO_x^-), calculated nitrification and sediment oxygen demand (SOD). Data are mean ± SE for all parameters.

		NH_4^+ Flux (μmol N m^{-2} hr^{-1})			**NO_x Flux (μmol N m^{-2} hr^{-1})**			**Nitrification (μmol N m^{-2} hr^{-1})**			**SOD (μmol O_2 m^{-2} hr^{-1})**		
Spring	Bare	8.24	±	2.52	-0.58	±	0.03	4.78	±	1.22	537.19	±	77.49
	Clam Bed	70.75	±	13.38	-0.19	±	0.29	8.69	±	0.20	1883.69	±	368.06
	Oyster Reef	-17.26	±	4.95	0.08	±	0.41	4.70	±	1.21	1443.94	±	356.35
Summer	Bare	46.92	±	16.04	2.35	±	0.84	4.93	±	1.20	1124.74	±	15.30
	Clam Bed	32.84	±	7.15	5.67	±	3.26	10.95	±	2.93	1258.32	±	270.33
	Oyster Reef	35.20	±	2.15	2.25	±	1.15	18.25	±	3.68	1624.84	±	228.21
Fall	Bare	-16.93	±	5.35	-3.42	±	0.43	1.54	±	0.42	324.75	±	23.19
	Clam Bed	26.97	±	17.33	3.97	±	2.90	12.38	±	5.41	582.43	±	204.68
	Oyster Reef	-5.58	±	5.65	1.15	±	1.14	3.13	±	0.77	336.56	±	60.77
Mean	Bare	12.74	±	10.51	-0.55	±	0.88	3.74	±	0.75	662.23	±	121.96
	Clam Bed	43.52	±	9.55	3.15	±	1.53	10.65	±	1.85	1241.48	±	236.99
	Oyster Reef	4.12	±	8.26	1.16	±	0.57	8.69	±	2.66	1135.11	±	236.15

Supplemental Table 1: Number of clams per core by season.

Season	*Core ID*	*Number of Clams*
Spring	Clam Bed-1	4
Spring	Clam Bed-2	3
Spring	Clam Bed-3	4
Summer	Clam Bed-1	0
Summer	Clam Bed-2	0
Summer	Clam Bed-3	0
Fall	Clam Bed-1	4
Fall	Clam Bed-2	4
Fall	Clam Bed-3	4