**Title:** Spatial dynamics of animal-mediated nutrients in temperate waters

**Running title:** Animals drive nutrient variability across scales

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EGL: Conceptualization (equal); formal analysis; investigation (lead); methodology (lead); visualization; writing – original draft preparation. CMA: Conceptualization (supporting); investigation (equal); methodology (equal). KDC: Conceptualization (supporting); funding acquisition (supporting); investigation (equal); methodology (equal); project administration (equal). JMS: Funding acquisition (supporting); investigation (supporting); project administration (supporting). KRK: Investigation (supporting). EJL: Investigation (supporting). BM: Investigation (supporting); methodology (supporting). ALB: Investigation (supporting). FJ: Funding acquisition (equal); project administration (supporting). IMC: Conceptualization (equal); funding acquisition (equal); methodology (equal); project administration (equal); supervision. All authors contributed to writing – review & editing.

**Abstract**

Consumer-mediated nutrient dynamics (CND), through which animals’ metabolic waste products fertilize primary producers, drive variability in nutrient availability in tropical waters. This variability influences primary productivity and community functioning. Yet, examinations of CND as a driver of nutrient variability in temperate marine ecosystems are limited. Therefore, we assessed the existence and drivers of variation in CND in temperate waters at meso, small, and fine spatial scales. To do so, we quantified the occurrence of 48 fish and 92 macroinvertebrate taxa and measured in situ ammonium at 27 northeast Pacific rocky reefs for three years and 17 kelp forests of varying density for one year. Ammonium concentrations ranged from 0.01 to 2.5 uM across rocky reefs separated by tens of km. The relationship between animal abundance and ammonium among sites was mediated by water flow, where flood tides seemed to “wash away” the effect of nutrient regeneration by animals, although enrichment was possible on ebb tides. Ammonium concentration was significantly greater within than outside of kelp forests, a difference that increased with kelp biomass, tidal exchange, and to a lesser degree animal biomass. Caging experiments revealed that fine-scale (~2 m) ammonium variability and nutrient enrichment were only possible under low-flow conditions. Our results suggest that CND drives nutrient variability at scales ranging from two meters to over 20 km, acting on a finer scale than allochthonous nitrogen sources such as upwelling. Therefore, consumer-mediated nutrient dynamics are implicated as a previously overlooked driver of spatial variation in primary productivity in temperate marine systems.

**Introduction**

Variation in resource availability across spatial and temporal scales can drive substantial heterogeneity in the growth, biomass and composition of primary producers (Tilman 1984; Leibold 1991; Dayton et al. 1999; McInturf et al. 2019). In many marine ecosystems, community structure is regulated through bottom-up control, i.e. structure depends on factors that generate variability in the resources available to lower trophic levels (Gruner et al. 2008). Although marine ecologists have historically focused on external, abiotic sources of nutrients (e.g., upwelling) as drivers of variability in nutrient availability, there is emerging evidence that consumers also contribute to bottom-up effects (Allgeier et al. 2017). Metabolic waste products (i.e., excretion and egestion) of animals fertilize primary producers via a process termed consumer-mediated nutrient dynamics (CND; Vanni, 2002). Consumers excrete metabolic waste in the form of ammonium (NH₄⁺), which is preferentially taken up by primary producers over other forms of nitrogen like nitrate and nitrite (Lobban and Harrison 1994; Phillips and Hurd 2004). However, the ecological importance of consumer-regenerated nutrients at varied spatial scales remains unclear. Therefore, identifying the extent to which biologically relevant variation in nutrient availability contributes to heterogeneity in primary productivity remains an active area of research (Allgeier et al. 2017).

Heterogeneity in consumer habitat use greatly influences spatial and temporal variation in nutrients supplied by animal waste (Uthicke 2001; Roman and McCarthy 2010; Benkwitt et al. 2019). For example, tropical coral reefs provide habitat, shelter, and food sources that attract dense aggregations of vertebrate and invertebrate consumers which regenerate nutrients (Archer et al. 2015; Shantz et al. 2015). On a meso-scale (i.e., 10 to 100 km; Broitman et al., 2001), productivity increases with proximity to reefs with high densities of fishes (Layman et al. 2016), while on a fine scale, sheltering schools of fish increase nitrogen concentrations around individual heads of corals relative to neighboring uninhabited corals (Holbrook et al. 2008). Ammonium supply can also vary with not only the body size, but also the diversity of the consumers (Allgeier et al. 2014). Diurnal migrations are another source of temporal and spatial variation in consumer-regenerated nutrients, as some fishes travel away from reefs to forage at night, then return to excrete waste around their hiding spots during the day (Meyer and Schultz 1985; Francis and Côté 2018). At an even larger-scale, variation can arise from the migration of megafauna; for instance, whales transport and deposit nutrients across thousands of kilometers as they travel from their feeding to breeding grounds (Doughty et al. 2016). However, the current understanding of animal-driven spatio-temporal variability of nitrogen is drawn substantially from tropical ecosystems (Meyer et al. 1983; Holbrook et al. 2008; Allgeier et al. 2013), often overlooking productive temperate marine ecosystems.

In temperate oceans, external sources of nutrients, such as upwelling and freshwater runoff, are generally considered the dominant drivers of nitrogen variability (Dayton et al. 1999; Lønborg et al. 2021). Due to the open nature of many nearshore environments, fast water flow from currents, tides, and wave action are believed to limit small-scale (1 to 100 m2) nutrient variation (Probyn and Chapman 1983). Therefore, research on intertidal and shallow subtidal ecosystems has traditionally focused on top-down trophic interactions as the drivers of community composition at small scales, while limiting considerations of resource limitation to large regional or continental scales (Paine 1986; Menge 1992). However, evidence suggests meso-scale variation in allochthonous nitrogen via upwelling and internal waves may contribute to bottom-up control of marine communities (Menge et al. 1997; Nielsen and Navarrete 2004; Leichter et al. 2023) and even weaken top-down control (Sellers et al. 2020). Consumer-mediated nutrient dynamics may also contribute to smaller scales of nutrient variability than previously assumed. For instance, the abundance of intertidal mussel beds has been linked to variation in nitrogen concentrations along entire coastlines (Pfister et al. 2014), across 10s of meters (Aquilino et al. 2009), and among tidepools (fine-scale microhabitats; Bracken, 2004). Therefore, regenerated nitrogen may contribute substantially to meso-, small-, and fine-scale variation in nutrient availability, even in fast-flow, upwelling-dominated nearshore coastal ecosystems.

Shallow subtidal rocky reefs and kelp forests are temperate nearshore habitats that attract dense aggregations of fishes and invertebrates, many of which are economically, ecologically, and culturally important (Steneck et al. 2002). Elevated NH₄⁺ excretion from the concentrated diversity and biomass of these communities may also contribute to nutrient hotspots on small to meso-scales and exceed NH₄⁺ delivery from other sources (Shrestha et al. 2024). Fast-growing canopy kelps, which form expansive underwater forests, may benefit from these excretions directly as a source of nitrogen (in the form of NH₄⁺), especially during low upwelling periods (Brzezinksi et al. 2013; Lees et al. 2024). These kelps, which comprise giant kelp (*Macrocystis pyrifera*) and bull kelp (*Nereocystis leutkeana*) in the northeast Pacific, also influence the hydrodynamics and hydrochemistry of seawater, both slowing water flow within the forests and generating gradients of carbon content, pH, alkalinity, and oxygen (Jackson and Winant 1983; Gaylord et al. 2007; Pfister et al. 2019). These modifications of the surrounding fluid environment by kelp forests could affect the productivity and community composition of other primary producers and contribute to small-scale spatial nutrient heterogeneity.

We aimed to quantify the contribution of animal-regenerated nitrogen to spatial variability of nutrients in a temperate, wave-swept upwelling region (Barkley Sound, British Columbia, Canada). This region is located on the traditional territories of the Huu-ay-aht Nation and comprises an archipelago of islands dotted with rocky reefs and kelp forests of heterogeneous structure. We hypothesized that animal-regenerated nutrients contribute to variability in resource availability across three distinct spatial scales. Specifically, we predicted that NH₄⁺ variation would be detectable at the meso-scale due to variation in animal abundance among sites. We also expected to observe variation in NH₄⁺ concentrations at small scales (within natural sites) and fine scales (between experimental cages), but only under conditions that allow for local enrichment (e.g., low tidal exchange). To test these predictions, we measured variation in NH₄⁺ concentrations among rocky reef sites (meso scale of ~10s of km), in and out of kelp forest sites (small scale of 5 m), and near experimentally caged consumers (fine scale of < 2 m, Fig. 1a–c). We quantified the abundance and diversity of fishes and invertebrates at each rocky reef and kelp forest site and measured kelp forest metrics and abiotic variables to explore potential drivers of variation in NH₄⁺ concentrations. By characterizing the scale at which animal-driven nutrients vary, we hope to uncover the extent to which consumers in temperate regions structure communities not only from the top down, but also the bottom up through CND.

**Methods**

*Site description*

Barkley Sound is located in an upwelling region on the west coast of Vancouver Island, Canada. Upwelling supplies nitrates in the spring and early summer, while storms flush riverine inputs into the nearshore in the winter and spring (Pawlowicz 2017). Due to the proximity of the Bamfield Marine Sciences Centre (BMSC), this region has been a long-term focal area for studies seeking to document the response of kelps to marine heatwaves, establish ecological baselines, and unravel ecosystem dynamics (Tanasichuk 1998; Starko et al. 2022, 2024; Attridge et al. 2024). Subtidal fish communities in this region comprise at least 18 families, including gobies, surfperches, rockfishes, greenlings, and sculpins (E.G. Lim, unpubl.). Macroinvertebrate assemblages, which are made up of over 49 families, are dominated by sea urchins, turban snails, sea stars, sea cucumbers, and abalone (E.G. Lim, unpubl.).

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**Figure 1.** Scales of study and site locations in Barkley Sound, British Columbia, Canada.(a) Meso-scale (among-site), (b) small-scale (within-site), and (c) fine-scale (microhabitat) schematics of the three spatial scales of variability investigated. (d) Rocky reefs (circles) surveyed for meso-scale ammonium variation, and kelp forests surveyed (triangles) for small-scale ammonium variation. Site colour indicates mean ammonium concentration found at each site across all three years, with darker points having greater concentrations of ammonium. (★) Indicates the location of the fine-scale sea cucumber caging experiment and (\*) denotes location of the crab caging experiment.

*Surveys of meso-scale (among-site) variation*

To explore meso-scale variation in animal-regenerated nutrients among rocky reefs, we measured ammonium (NH₄⁺) concentrations and surveyed fish and invertebrate communities at 27 subtidal sites ranging from 0.07 – 24 km apart in Barkley Sound (Fig. 1d). We used a globally standardized method (i.e., Reef Life Survey, RLS) at each site to estimate fish and invertebrate abundance and collected subtidal NH4+ samples during each survey. We conducted our surveys in the spring (April-May) for three years (2021-2023), with all annual surveys occurring within two weeks of each other (Supplemental Material; Table S1.01). A full explanation of the Reef Life Survey method is available online (http://www.reeflifesurvey.com/methods) and provided by Edgar and Stuart-Smith (2009) and Edgar et al. (2020). Briefly, at each rocky reef site, a pair of RLS-trained SCUBA divers assessed fish and invertebrate abundance and diversity along each side of a 50 m transect line. First, fishes in the water column were counted and sized (total length, in various size class categories) within 5 m on either side of the transect line (500 m2), and then benthic cryptic fishes (also sized) and large mobile invertebrates (> 2.5 cm) were counted within 1 m on either side of the transect line (100 m2).

Immediately following each RLS survey, we collected three 60 mL subtidal seawater samples at 0, 25, and 50 m along the transect, 0 – 2 m above the substrate, and stored the syringes in sealed plastic bags upon collection to prevent contamination. Seawater samples were filtered into opaque amber bottles in the field and frozen for a maximum of two weeks before NH4+ analysis. We confirmed that freezing samples for this duration did not affect NH4+ concentration (E. G. Lim unpubl.). In 2021 and 2022, we followed the fluorometric method using 40 mL seawater samples (Holmes et al. 1999), and in 2023, we followed the fluorometric standard-additions protocol II (Taylor et al. 2007). These methods produce similar results, although the Holmes single spike method is associated with larger variability in measurements (Taylor et al. 2007). The three NH₄⁺ samples collected during each survey were averaged to determine the mean NH₄⁺ concentration for each site.

*Surveys of small-scale (within-site) variation*

To investigate small-scale variation of animal-regenerated nutrients, we measured NH4+ concentrations inside and outside kelp forests and surveyed the resident biological communities as potential moderators of this variation. Our 16 sites comprised forests of varying densities dominated by giant kelp or bull kelp, and two no-kelp control sites. We conducted surveys from July to September 2022 (Table S1.02). First, divers conducted RLS surveys (as above) along 50 m transect lines along the edge of the kelp forest to quantify the abundance and biodiversity of fish and invertebrate communities associated with each kelp forest (Fig. S1.01). We were unable to conduct RLS transects entirely within the kelp forests due to visual obstruction and concern with diver entanglement by the kelp. Next, divers ran four 5 m-long transects perpendicular to the RLS transect (5 m apart) into the kelp forest to assess kelp density, canopy height, and kelp biomass (Fig. S1.01). Divers then counted the number of canopy kelp individuals (bull or giant kelp) within 0.5 m on either side of each kelp transect to measure kelp density. To estimate canopy height, we measured the length of five randomly selected kelp individuals per species per kelp transect; for bull kelp we measured the total length from holdfast to pneumatocyst in situ, but for giant kelp, we collected five random individuals to measure the length from holdfast to the tip of the apical meristem on dry land. To quantify bull kelp biomass, we measured the sub-bulb circumference (15 cm below the bottom of the bulb) of the same five bull kelps per transect in situ and calculated individual biomass using a quadratic diameter to biomass formula for Barkley Sound (C. M. Attridge unpubl.). For giant kelp biomass, we weighed (to the nearest 1 g or less) the same five individuals per transect that were collected for total length measurements. We multiplied the mean biomass estimate for each kelp species by the species density to calculate a biomass/m2 estimate for each kelp transect, which we then averaged over the four transects per forest to estimate overall mean forest biomass/m2. We estimated total forest area by swimming around the perimeter of the forest on the surface with a Garmin GPSMAP 78SC, which we used to calculate total forest biomass (kg).

Finally, to compare NH₄⁺ concentrations inside vs outside each kelp forest, we collected paired 60 mL syringes of seawater immediately outside the kelp forest within 0 – 2 m above the substrate, and 5 m into the kelp forest at the same depth. We collected three paired NH₄⁺ samples from each site, which were spaced 5 m apart, by matching them with the first three kelp transects (Fig. S1.01). At our two control, no-kelp sites, we followed the same procedure in terms of number and spacing of samples, with the ‘inside’ samples taken ~ 5 m closer to shore than the ‘outside’ samples. Outside each kelp forest and at each control site, we also collected seawater to create a standard curve, following the standard-additions protocol II for the fluorometric detection of NH₄⁺ in 40 mL samples (Taylor et al. 2007). Samples and standards were filtered into amber bottles in the field and stored on ice for transportation back to the laboratory, at which point we measured NH4+ concentration in each sample bottle following the protocol above (Taylor et al. 2007). For each paired inside and outside kelp forest NH₄⁺ sample, we calculated ∆NH₄⁺ = inside NH₄⁺ - outside NH₄⁺.

*Surveys of fine-scale (microhabitat) variation*

To quantify the ability of animals to affect the NH₄⁺ concentration in their immediate vicinity we conducted two caging experiments in situ near Bamfield, BC. We used California sea cucumbers (*Apostichopus californicus*) in the first caging experiment because this species is a large, abundant invertebrate with a high excretion rate (Bray et al. 1988). The first experiment occurred May 27 – 28, 2021 at Scott’s Bay (48°50'05.2"N, 125°08'49.3"W), a wide, exposed bay that opens into Trevor Channel (Fig. 1d). We constructed 18 wire cages (26 x 26 x 26 cm), which we covered in 2 mm plastic mesh. These cages were spaced 3 m apart along two weighted lines (9 cages per line) and deployed at 3 to 5.8 m depth. We collected adult California sea cucumbers from the site via SCUBA, measured contracted sea cucumber length and girth, and immediately placed them into the cages in randomly assigned densities of 0, 1, or 2 sea cucumbers (*n* = 6 replicates per density). After 24 hours, we returned to collect water samples from each cage in situ. While underwater, we minimized water movement by reducing our fin and hand movements while opening the mesh lids, which were secured with wire and just wide enough to collect a 60 mL syringe of seawater. Once at the surface, we filtered 40 mL of each sample into amber bottles and transported them on ice to the lab, where we measured NH4+ using the fluorometric method (Holmes et al. 1999).

We used red rock crabs (*Cancer productus*) in the second caging experiment to see if a species with an even larger individual-level excretion rate could produce fine-scale nutrient variation. The second experiment occurred over nine days from June 10 – 19, 2023 in Bamfield Inlet (48°49'53"N 125°08'11"W), a narrow, sheltered inlet (Fig. 1d). We replicated this experiment from June 19 – 28, 2023 following the same methodology. We collected red rock crabs from the site using crab traps and kept them at BMSC in flow-through sea tables for 2 – 10 days. Crabs were fed salmon every 2 – 4 days, and all crabs were fed the night before each experiment started. We constructed 12 cages from clear plastic (40 x 28 x 17 cm), with two 15 x 9 cm windows covered in a dual layer of 10 mm plastic mesh and 1 mm mesh to allow for water flow. The cages were randomly distributed every 2 m along a lead line anchored with cement blocks 0.8 m below chart datum. Each cage contained either one large crab (carapace 15.0 – 15.9 cm), one medium crab (11.6 – 14.4 cm), or a control (i.e., a small rock, scraped clean, so weight was similar across all cages) (*n* = 4 replicates per experiment). During both experiments, we replaced the crabs after 4 days with freshly fed, similar-sized crabs, at this point, we re-randomized the order of the cages along the line to minimize any effect of cage location. We measured seawater NH₄⁺ concentration via snorkel at low tide at the beginning, middle, and end of each nine-day experiment. A fixed narrow rubber tube that began in the centre of the cage and extended several inches outside the mesh window allowed us to draw water samples using a 60 mL syringe without disturbing the cages. We filtered 40 mL of each sample into amber bottles, which were stored on ice before NH₄⁺ analysis via fluorometric standard-additions protocol II (Taylor et al. 2007).

*Statistical analyses*

All statistical analysis were conducted in R (v4.1.2, R Core Team, 2019) using RStudio (v1.3.1093, RStudio Team, 2016). We used tidyverse packages for data manipulation and visualization (Wickham et al. 2019), ‘glmmTMB’ for all modelling (Brooks et al. 2017), and DHARMa to check model fit (Hartig 2022). We ensured all models met assumptions by inspecting residuals using the DHARMa simulateResiduals function and checked for collinearity between variables using the vif function from the car package with a cutoff value of 2.

For each Reef Life Survey conducted, we calculated fish biomass from fish length following the formula:

W is fish weight, L is the fish length, a and b are species-specific constants from FishBase (Froese et al. 2014). All mobile invertebrates were counted, but only sunflower sea stars (*Pycnopodia helianthoides*) and economically important species (abalone [*Haliotis kamtschatkana*] and scallops [*Crassadoma gigantea*]) were sized. We used published length–weight relationships to calculate wet weight for these three species. For all other invertebrates, we used published wet weights to estimate biomass for each taxon. We used shell-free wet weight for species with large shells, such as hermit crabs and snails. When biomass information was unavailable for a species, we used estimates from the closest relative or most similarly sized species available (Table S1.03). Animal abundance per m2 was calculated as the total number of fishes and invertebrates counted on each survey, and animal biomass per m2 was calculated as the total wet weight of all animals on each survey (divided by 500 m2 for pelagic fishes and by 100 m2 for cryptic fishes and macroinvertebrates). We used the ‘vegan’ package to calculate Shannon and Simpson diversity indices (Oksanen et al. 2022). We calculated tidal exchange by computing the percent change of the tide height every minute, averaged over the hour-long survey.

Meso-scale (among-site) variation

To explore meso-scale variation in NH₄⁺, we constructed generalized linear mixed-effect models (GLMMs) with NH₄⁺ concentration as the response variable, and animal abundance, tidal exchange, an interaction between abundance and tide, Shannon diversity, and survey depth as predictors, and random effects of site and year. All predictors were scaled and centered around the mean using the scale function. We used a gamma distribution (link = ‘log’). To test the robustness of our modelling approach, we considered animal biomass/m2 as a predictor instead of abundance/m2, and Simpson’s diversity instead of Shannon diversity; alternative models including these predictors were not better supported by AIC (Table S1.04).

Small-scale (within-site) variation

To determine whether NH₄⁺ concentration differed inside and outside of kelp forests, we used a linear mixed-effects model (LMM) with ∆NH₄⁺ as the response variable (*n* = 3 estimates per site), and kelp species, mean forest kelp biomass (per m2), tidal exchange, animal biomass, survey depth, Shannon diversity, and interactions between kelp biomass and tidal exchange, kelp biomass and animal biomass, and animal biomass and tide exchange as fixed effects. All continuous predictors were scaled and centered around the mean as above. We included site as a random effect (1|site) as each site contributed three estimates to the analysis and used a Gaussian distribution. As above, we chose our final set of predictors upon comparing AIC values of models with alternative predictors (Table S1.05).

Fine-scale (microhabitat) variation

We constructed separate linear models for each caging experiment to quantify the impact of caged animals on adjacent NH₄⁺ concentration. For the sea cucumber experiment, we regressed cage NH₄⁺ concentration against the treatment (i.e., sea cucumber density: 0, 1, or 2 sea cucumbers) and cage depth (centered) using a Gaussian distribution. For the red rock crab experiment, we constructed a generalized linear mixed-effects model (GLMM) with cage NH₄⁺ concentration as the response variable and treatment (no crab, medium crab, or large crab) as the predictor variable with a gamma distribution (link = ‘log’). We included a random effect of sampling day because we measured NH₄⁺ three times per experiment, and a random effect of experimental week, because we replicated the whole experiment twice. We also calculated the respective NH₄⁺ supply rates by sea cucumbers and red rock crabs using a previously generated size-excretion relationship for each sea cucumber (Table S1.06), and a carapace-excretion relationship for each crab (Table S1.07).

**Results**

We found evidence of meso-scale variation in ammonium (NH₄⁺) concentrations, which ranged from 0.07 μM – 2.06 μM among rocky reefs in Barkley Sound (Fig. 1d). Overall, we found no evidence that NH₄⁺ concentration was correlated with animal abundance (GLMM, *p* = 0.57), tidal exchange (*p* = 0.99), Shannon diversity (*p* = 0.41), or survey depth (*p* = 0.61; Fig. 2a; Table S1.08). However, we did find a significantly negative interaction between animal abundance and tidal exchange (*p* = 0.01; Fig. 2b), revealing a weakly positive effect of total animal abundance per m2 on NH₄⁺ concentration, but only at ebb tide. **A comparison of different colored lines

Description automatically generated with medium confidenceFigure 2.** Ecological drivers of seawater ammonium concentration observed across 27 rocky reef sites (meso-scale) in Barkley Sound, British Columbia, Canada. (a)Model coefficients with 95% confidence intervals, and (b) model-generated predictions with shaded 95% confidence intervals of the effect of the interaction between animal abundance and tidal exchange on among-site variation in ammonium concentration. The coefficients were generated from a generalized linear mixed-effects model with a gamma distribution (link = ‘log’), so coefficients are presented in log space. Continuous predictors were centred and scaled to compare effect sizes between predictors with varying units.

We also documented evidence of significant small-scale, within-site variation of NH₄⁺ (Fig. 3; Table S1.09).; concentrations were 1.3x greater inside giant kelp forests and 1.6x greater inside bull kelp forests than outside (LMM, *p* < 0.001; Fig. 3b). The ‘excess’ NH₄⁺ concentration inside kelp forests increased with kelp biomass (*p* < 0.001; Fig. 3c), and tidal exchange (*p* = 0.02; Fig. 3a). We found limited evidence for an effect of animal biomass (*p* = 0.10; Fig. 3a), and no evidence of an effect of survey depth (*p* = 0.19; Fig. 3a), or Shannon diversity on ∆NH₄⁺ (*p* = 0.23; Fig. 3a). There was a positive interaction between kelp forest biomass and tidal exchange, whereby the positive effect of kelp biomass on ∆NH₄⁺ increased with tidal exchange (*p* < 0.001; Fig. 3a,c). We also identified a negative interaction between kelp biomass and animal biomass (*p* = 0.04; Fig. 3a,d), and a negative interaction between tidal exchange and animal biomass (*p* = 0.001; Fig. 3a,e). The change in NH₄⁺ was negative between samples taken 5 m apart at the no-kelp control sites (*p* = 0.004; Fig. 3b).

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**Figure 3.** Drivers of variation in ammonium concentration inside vs outside kelp forests across 16 sites (small-scale) in Barkley Sound, British Columbia, Canada. (a) Model coefficients with 95% confidence intervals and (b-e) model-generated predictions with shaded 95% confidence intervals of the effects of significant drivers of within-site variation in ammonium concentration. Continuous variables were scaled and centered to facilitate comparisons between variables measured in different units. Nereo = *Nereocystis luetkeana*, Macro = *Macrocystis pyrifera*.

We found mixed evidence for animal-related fine-scale variability in NH₄⁺ concentration. For sea cucumbers, we found no effect of sea cucumber density on cage NH₄⁺ concentration despite a supply rate of 14 μM/h and 28 μM/h for the low and high treatments, respectively. Overall, the mean NH₄⁺ concentration was 0.92 ± 0.04 μM across all cages (LM, *p* > 0.75 for both treatments; Fig. 4a; Table S1.10). However, we saw a positive effect of cage depth, whereby NH₄⁺ increased by 0.38 ± 0.05 μM per m increase in depth (*p* < 0.001). For red rock crabs, both medium and large crabs significantly increased the cage NH₄⁺ concentration relative to control cages, by 8.7x and 12.1x respectively (GLMM, *p* < 0.001 for all; Fig. 4b; Table S1.11). Medium crabs excreted on average 88 μM/h while large crabs excreted 150 μM/h.

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**Figure 4.** Fine scale effect of California sea cucumbers (*Apostichopus californicus*) and red rock crabs (*Cancer productus*) on ammonium concentration. Mean ammonium concentration in experimental cages containing (a) zero (control), one, or two California sea cucumbers (*n* = 6), and (b) zero (control), one medium, or one large red rock crab (*n* = 8 for control and medium, *n* = 7 for large). Error bars indicate 95% confidence intervals.

**Discussion**

We found evidence of variability of animal-regenerated nutrients from the largest to the smallest scale examined, although the strength of the signal varied. Ammonium (NH₄⁺) varied by up to 16x between rocky reef sites within a year, 1.9x inside vs outside kelp forests, and 40x between cages with and without crabs. Water flow (i.e., tidal exchange and wave exposure) mediated the capacity for animals to saturate the water column with nutrients. Among sites a flooding tide seemed to “wash away” the impact of animals on NH₄⁺ concentrations; in contrast, within sites, moving water made kelps’ ability to slow flow and trap animal-regenerated nutrients more pronounced. In the fine-scale caging experiments we only detected an effect when the nutrient providers were crabs – an effect that we attribute mainly to the low water flow in the protected inlet rather than to the taxa. Nevertheless, across all three scales, there was animal-mediated spatial heterogeneity in nutrient availability, which may contribute to bottom-up effects.

*Meso-scale (among-site) variation*

In rocky reef habitats, we detected a 16-fold difference in NH₄⁺ among sites with the lowest and highest concentrations. This difference is substantially greater than previous measurements of among-site variation in nitrate (3.7x and 6.5x) and ammonium (0.4x and 0.8x) from the same region (Druehl et al. 1989; Hurd et al. 2000). It is also larger than among-site NH₄⁺ differences measured in nearby Washington State (1.1x, Pfister et al., 2014). We had predicted that variation in NH₄⁺ concentration among sites would be driven primarily by animal abundance. However, the only significant predictor of among-site differences in NH₄⁺ was a negative interaction between tidal exchange and animal abundance, whereby animal excretions may enrich the seawater when the tide is ebbing, but the effect of animal abundance is washed away when the tide comes in. Although marine species diversity sometimes covaries with animal abundance or biomass (Yee and Juliano 2007; Müller et al. 2018), we found no relationship between Shannon diversity and NH₄⁺. We did not quantify intertidal animals or microbial regeneration, which are additional sources of NH₄⁺ (Aquilino et al. 2009; Lowman et al. 2023), but these sources may be more important in shallower waters and soft-sediment areas than on the subtidal rocky reefs we studied. We conclude that CND likely contributes to meso-scale variation in NH₄⁺ in an unexpected, dynamic, tide-associated manner, which could drive among-site variation in primary productivity and thus bottom-up control.

*Small-scale (within-site) variation*

We found evidence of kelp-mediated nutrient variation on a smaller scale (5 m) than previously established. Although greater NH₄⁺ concentrations inside high-density kelp forests have been documented (e.g., Pfister et al., 2019), these studies compared nutrient samples taken from the middle of very large kelp forests to sites more than 50 m away from the forest edges (Stewart et al. 2009; Pfister et al. 2019; Traiger et al. 2022). By sampling inside and outside forests across a gradient of kelp densities we further demonstrate a positive relationship between kelp biomass and NH₄⁺ retention. The retention of NH₄⁺ observed is likely due to the dampening of flow within the kelp forest bed and subsequent flow acceleration around the edges (Gaylord et al. 2007; Rosman et al. 2007). Indeed, as predicted, we found the effect of kelp biomass on NH₄⁺ retention was more pronounced when the tide was rising (flood tide). Unfortunately, we never sampled on ebbing tides, and did not quantify water motion due to waves or currents, so we could only contrast slack and flooding tides. Nevertheless, it seems that water flow due to tidal exchange enhances, rather than masks, NH₄⁺ variability within kelp forests.

We uncovered additional drivers of differences in NH₄⁺ concentration inside and outside kelp forests, namely kelp species and animal biomass. We found greater ∆NH₄⁺ (NH₄⁺ inside - NH₄⁺ outside) in bull kelp forests compared to giant kelp forests, which may be due to their different allocations of biomass in the water column and thus different alterations of water flow. Indeed, Traiger et al., (2022) found the effect of giant kelp forests on water chemistry was smaller than that of bull kelp forests, previously described by Murie and Bourdeau (2020). Species-specific NH₄⁺ uptake rates may also impact NH₄⁺ retention, as giant kelp may have higher NH₄⁺ uptake rates than bull kelp (Ahn et al. 1998; Lees et al. 2024). Somewhat surprisingly, NH₄⁺ was lower in the “inside” samples than in the “outside” samples at the no-kelp sites. Despite our efforts to maintain a consistent sampling depth, the “inside” samples were taken at slightly shallower depths since they were closer to shore. This result is therefore in line with the observation that NH₄⁺ tends to *increase* with depth (Brzezinksi et al. 2013). The fact that we found the opposite trend at our kelp sites increases our confidence that kelp help to retain NH₄⁺ within forests.

Even though kelp forests attract dense aggregations of fishes and invertebrates, the positive effect of animal biomass on ∆NH₄⁺ was weak and mediated by water flow and kelp biomass. Shannon diversity was positively associated with both animal and kelp biomass (Fig. S1.02) but had no effect on ∆NH₄⁺. The negative interactions between animal biomass and both tide and kelp biomass suggest a potentially saturating relationship among these variables. When animal biomass was low, increased kelp biomass or tidal exchange increased ∆NH₄⁺, whereas at high animal biomass, increasing kelp or flow had no effect. There may therefore be a threshold for how much animals can saturate and therefore increase ∆NH₄⁺ in kelp forest ecosystems. Beyond this point, increased water motion or kelp biomass no longer enhances NH₄⁺ inside kelp forests relative to the forest edge.

*Fine-scale (microhabitat) variation*

At the smallest scale of variability tested, we found evidence of variation in NH₄⁺ only in our cage experiments in the sheltered inlet, which may suggest that water motion mediates variation at this scale as well. Under laboratory conditions we confirmed that NH₄⁺ enrichment by animals declines with increasing flow rates (Fig. S1.03). Alternative or complementary explanations include a taxonomic effect and/or an experimental effect. The crabs in the sheltered cages excreted NH₄⁺ at a rate roughly 6x greater than the sea cucumbers caged in the more exposed location. This difference in NH₄⁺ production could have given us more scope to detect differences among treatments in the protected inlet. In addition, the crab cages were constructed with only two mesh windows, in contrast to the fully meshed cages of sea cucumbers, which could have promoted nutrient retention in the former. The experimental duration was also shorter for the sea cucumber experiment, but NH₄⁺ concentrations in the crab cages were consistent across the three sampling days (day one, four, and nine) so cumulative enrichment is unlikely.

Fine-scale NH₄⁺ enrichment by animals is nevertheless possible in wave-exposed conditions; for example, seawater above mussel beds had ~16x greater NH₄⁺ compared to neighbouring rock without mussels on the northern California coast (Aquilino et al. 2009). Our cage experiments were not designed to test why variation was found in one experiment and not the other, but rather to see whether fine-scale variation might arise at all. Therefore, we simply conclude that at least in sheltered conditions, variation on the scale of meters driven by animal biomass is possible.

*Implications for primary productivity*

Heterogeneity in primary productivity arises from variation in resource supply. Increased primary productivity has been seen with orders of variation (i.e., 1.3 – 9x) in NH₄⁺ (Uthicke and Klumpp 1998; Arzul 2001; Vinther and Holmer 2008; West et al. 2009), which is within the range of variation we observed at all three spatial scales considered here. Regardless of nitrate availability, NH₄⁺ is less costly for primary producers to use, and can even inhibit nitrate uptake (MacIsaac and Dugdale 1969, 1972). In nitrate-replete upwelling ecosystems of the west coast of North America, a 15.8x increase in NH₄⁺ (from 0.08 uM to 1.26 uM) was linked to increased tissue nitrogen and percent cover of an intertidal seaweed (Aquilino et al. 2009), while a 1.8x increase in NH₄⁺ increased growth in subtidal seaweeds (Druehl et al. 1989). Furthermore, the mismatch of timing between upwelling and the growing season at temperate latitudes leads to an increased reliance on NH₄⁺ over time as nitrate becomes depleted (Druehl et al. 1989; Brzezinksi et al. 2013). For example, oceanographic processes may only provide half of the nitrogen needed to sustain giant kelp growth between late summer and early fall in southern California (Fram et al. 2008).

Heterogeneity in NH₄⁺ availability likely influences primary productivity in Barkley Sound at each spatial scale studied. Although nitrate varies among sites at the meso-scale (Druehl et al. 1989; Hurd et al. 2000), external nutrient supply (upwelling) may not control primary productivity as strongly as local factors (Pawlowicz 2017). The meso-scale NH₄⁺ variability we documented may be one such local factor. At a small scale, nitrate from external sources (e.g., upwelling and run-off) becomes depleted or unchanged as it flows through a kelp forest (Pfister et al., 2019; Stewart et al., 2009), in contrast to NH4+ which is continuously regenerated by animals within and around the forest and seemingly retained. As such, kelp forests effectively concentrate their preferred form of nitrogen. This process likely facilitates not only the growth of these canopy kelps but also understory seaweeds and phytoplankton in the water column. As urchin overgrazing and climate change continue to degrade kelp forests, this important ecosystem function may be lost. At fine scales, invertebrates and fishes living in close contact with primary producers can enrich kelps directly. Many seaweeds are capable of surge uptake of NH₄⁺ (Cedeno et al. 2021), allowing them to maximize the benefit from animal excretion directly on their thalli. Indeed, given turnover rates of water and NH₄⁺ uptake rates, seaweeds in New Zealand were estimated to derive up to 79% of their needed nitrogen from direct epifauna excretion (Taylor and Rees 1998).

*Conclusion*

Despite the mixing forces of currents, tides and waves, spatial heterogeneity in NH₄⁺ concentration was detectable at meso, small, and fine spatial scales. Given the annual depletion of nitrates each summer (Druehl et al. 1989), primary producers’ preference for NH₄⁺ over nitrate (Phillips and Hurd 2004), and capacity for surge uptake of NH₄⁺ (Cedeno et al. 2021), it seems likely that the animal-driven variation in NH₄⁺ we observed could be contributing to heterogeneity in primary productivity. Our results disrupt the dominant paradigm that bottom-up effects in temperate waters are primarily driven by external sources of nutrients acting on large scales, while animals contribute to smaller-scale variation mainly through top-down, consumptive effects. Instead, animals in temperate waters likely drive bottom-up effects across multiple spatial scales while also contributing to top-down effects. Animal-driven spatio-temporal variability of nitrogen is known to drive bottom-up effects in the tropics, and our results suggest animal-regenerated nutrients also play a previously unappreciated role in shaping nutrient availability in temperate regions as well.

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**Data Availability Statement**

All datasets generated during the current study and code are available in the GitHub repository: <https://github.com/em-lim13/Ch2_Spatial_pee>.

**Conflict of Interest Statement**

The authors declare no conflict of interest.

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**Supplemental Material for:**

**Title:** Spatial dynamics of animal-mediated nutrients in temperate waters

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**Supplemental Material Section 1.**

**A green hill with yellow dots and a black text

Description automatically generated**

**Figure S1.01.** Schematic of methods used to survey biological communities adjacent to a kelp forest, kelp forest density, and NH₄⁺ inside vs outside the forest. We first ran a 50 m Reef Life Survey transect parallel to the kelp forest (green shaded area) and counted fishes in the water column within 5 m on either side of the transect (light blue shaded areas), and cryptic fishes and macroinvertebrates within 1 m on either side of the transect (darker blue shaded area). Next, we ran four 5 m long transects into the kelp forest, 5 m apart from each other, to assess kelp density and biomass within 0.5 m on either side of the transect (four perpendicular black lines). Finally, we took NH₄⁺ samples at the beginning and end of the first three kelp transects (yellow circles) to compare NH₄⁺ inside vs outside kelp forests.

A comparison of different types of mass

Description automatically generated

**Figure S1.02**. Relationship between Shannon diversity and (a) kelp forest biomass and (b) animal biomass in kelp forests across 16 sites (small-scale) in Barkley Sound, British Columbia, Canada. Macro = *Macrocystis pyrifera*, Nereo = *Nereocystis luetkeana*, None = no kelp control.

**A diagram of a flow rate

Description automatically generated**

**Figure S1.03.** Change in ammonium in containers containing zero or four California sea cucumbers (*Apostichopus californicus*) relative to initial ammonium concentration after 24 hours in mesocosms with varying flow rates. Shaded areas indicate 95% confidence intervals, and raw data are plotted as points. While NH₄⁺ concentration remained the same across flow rates in the control mesocosms, sea cucumbers enriched NH₄⁺ concentration when flow was low. This enrichment declined as flow rate increased.

**Table S1.01.** Rocky reef sites sampled using Reef Life Survey methods, with the associated coordinates and years each site was surveyed.

|  |  |  |  |
| --- | --- | --- | --- |
| **Site code** | **Site name** | **Coordinates** | **Years sampled** |
| BMSC1 | Dodger Channel | 48.82894897, -125.1975708 | 2021, 2022, 2023 |
| BMSC2 | Kirby | 48.85039902, -125.1987686 | 2021, 2023 |
| BMSC3 | Ohiat | 48.85558319, -125.1837997 | 2021, 2022, 2023 |
| BMSC4 | Kii xin | 48.81511688, -125.1753311 | 2021, 2023 |
| BMSC5 | Taylor Rock | 48.82733154, -125.1966019 | 2021, 2022, 2023 |
| BMSC6 | Baeria Rocks South Island | 48.95023346, -125.1555481 | 2021, 2022, 2023 |
| BMSC7 | Baeria Rocks N Island Southside | 48.95464325, -125.1539917 | 2021 |
| BMSC8 | Baeria Rocks N Island Northside | 48.95508194, -125.1533737 | 2021, 2022, 2023 |
| BMSC9 | Eagle Bay | 48.83478928, -125.1470261 | 2021, 2022, 2023 |
| BMSC10 | Ross Islets Slug Island | 48.87051773, -125.160347 | 2021, 2022, 2023 |
| BMSC11 | Wizard Island South | 48.85746765, -125.1582336 | 2021, 2022, 2023 |
| BMSC12 | Wizard Island North | 48.858284, -125.1609192 | 2021, 2022, 2023 |
| BMSC13 | Effingham West | 48.8650322, -125.3137207 | 2021, 2022 |
| BMSC14 | Effingham Archipelago | 48.87908173, -125.2974014 | 2021, 2022 |
| BMSC15 | Raymond Kelp Rock | 48.88028336, -125.3128815 | 2021, 2022 |
| BMSC16 | Faber Islets | 48.89070129, -125.300499 | 2021, 2022 |
| BMSC17 | Wouwer Channel | 48.86548233, -125.3614807 | 2021, 2022 |
| BMSC18 | Eussen Rock | 48.91161728, -125.2670364 | 2021, 2022 |
| BMSC19 | Ed King SW Pyramid | 48.82860184, -125.2212982 | 2021, 2022, 2023 |
| BMSC20 | Ed King East | 48.83566666, -125.214798 | 2021, 2022, 2023 |
| BMSC21 | Dixon SW | 48.85205078, -125.1235657 | 2021, 2022, 2023 |
| BMSC22 | Dixon Inside | 48.85426712, -125.1170349 | 2021, 2022, 2023 |
| BMSC23 | Aguilar Point | 48.837589, -125.144145 | 2022, 2023 |
| BMSC24 | Swiss Boy | 48.916073, -125.131174 | 2023 |
| BMSC25 | Goby Town | 48.838595, -125.135015 | 2023 |
| BMSC26 | Hosie South | 48.9071, -125.037017 | 2023 |
| BMSC27 | San Jose North Island | 48.901183, -125.060433 | 2023 |

**Table S1.02.** Kelp forest site names, coordinates, survey dates and dominant kelp forest species. Macro = giant kelp (*Macrocystis pyrifera*), Nereo = bull kelp (*Nereocystis luetkeana*).

|  |  |  |  |  |
| --- | --- | --- | --- | --- |
| **Site code** | **Site name** | **Coordinates** | **Date** | **Kelp** |
| KCCA1 | Ross Islet Slug Island | 48.87039, -125.1599 | 2022-07-04 | Macro |
| KCCA2 | Between Scott & Brady | 48.83287, -125.1493 | 2022-07-05 | Macro |
| KCCA3 | Dodger Channel 1 | 48.83072, -125.19439 | 2022-07-06 | Macro |
| KCCA4 | Flemming 112 | 48.87868, -125.1434 | 2022-07-07 | Macro |
| KCCA6 | Less Dangerous Bay | 48.87535, -125.0915 | 2022-07-24 | None |
| KCCA7 | Ed King East Inside | 48.83608, -125.2131 | 2022-07-25 | Macro |
| KCCA9 | Wizard Islet South | 48.85728, -125.1595 | 2022-07-27 | Macro |
| KCCA12 | North Helby Rock | 48.85831, -125.1649 | 2022-08-03 | Macro |
| KCCA14 | Danvers Danger Rock | 48.877, -125.0923 | 2022-08-06 | Macro |
| KCCA15 | Cable Beach | 48.82484, -125.16067 | 2022-08-07 | Nereo |
| KCCA16 | Tzartus 116 | 48.90084, -125.0811 | 2022-08-18 | Macro |
| KCCA17 | Turf Island 2 | 48.884864, -125.146937 | 2022-08-20 | Macro |
| KCCA18 | Second Beach | 48.815969, -125.174 | 2022-08-21 | Nereo |
| KCCA19 | Wizard Islet North | 48.85916, -125.15908 | 2022-08-22 | None |
| KCCA21 | Bordelais Island | 48.81822, -125.2294516 | 2022-09-01 | Nereo |
| KCCA22 | Taylor Rock | 48.82721, -125.19717 | 2022-09-05 | Macro |

**Table S1.03**. Wet weight estimates for each invertebrate species used to calculate total biomass for Reef Life Survey data. We used shell-free wet weight for species with large shells (e.g., hermit crabs, snails). When weight information was unavailable for a species, we used estimates from the closest relative or most similarly sized species available. For the three species we sized in situ (*Pycnopodia helianthoides*, *Crassadoma gigantea*, and *Haliotis kamtschatkana*), we used published length-weight relationships to calculate wet weight from size.

|  |  |  |
| --- | --- | --- |
| **Species** | **Weight (g)** | **Source, proxy species if applicable** |
| *Cancer productus* | 200 | E.G. Lim, unpubl. |
| *Glebocarcinus oregonensis* | 3 | Hines 1982, small crabs |
| *Romaleon antennarium* | 3 | Hines 1982, small crabs |
| *Chorilia longipes* | 1.235 | Hines 1982, *Pugettia richii* |
| *Pugettia foliata* | 1.235 | Hines 1982, *Pugettia richii* |
| *Pugettia gracilis* | 1.235 | Hines 1982, *Pugettia richii* |
| *Pugettia producta* | 46 | Hines 1982 |
| *Pugettia richii* | 1.235 | Hines 1982 |
| *Scyra acutifrons* | 2 | Hines 1982 |
| *Scyra* spp*.* | 1.235 | Hines 1982 |
| *Cryptolithodes sitchensis* | 3 | Hines 1982, small crabs |
| *Cryptolithodes typicus* | 3 | Hines 1982, small crabs |
| *Hapalogaster mertensii* | 65 | Stewart et al. 2015, *Phyllolithodes papillosus* |
| *Lopholithodes mandtii* | 65 | Stewart et al. 2015, *Phyllolithodes papillosus* |
| *Phyllolithodes papillosus* | 65 | Stewart et al. 2015 |
| *Oregonia gracilis* | 3 | Hines 1982, small crabs |
| *Paguroidea* spp*.* | 0.43 | McKinney et al. 2004, Paguroidea |
| *Pagurus beringanus* | 0.43 | McKinney et al. 2004, Paguroidea |
| *Pagurus hemphilli* | 0.43 | McKinney et al. 2004, Paguroidea |
| *Pandalus danae* | 0.11 | McKinney et al. 2004, *Palaemonetes pugio* |
| *Pandalus gurneyi* | 0.11 | McKinney et al. 2004, *Palaemonetes pugio* |
| *Pandalus* spp*.* | 0.11 | McKinney et al. 2004, *Palaemonetes pugio* |
| *Pandulus spp.* | 0.11 | McKinney et al. 2004, *Palaemonetes pugio* |
| *Lophopanopeus bellus* | 3 | Hines 1982, small crabs |
| *Pachycheles pubescens* | 4.25 | Stillman and Somero 1996, *Petrolisthes* spp*.* |
| *Petrolisthes eriomerus* | 4.25 | Stillman and Somero 1996, *Petrolisthes* spp*.* |
| *Heptacarpus stylus* | 0.11 | McKinney et al. 2004, *Palaemonetes pugio* |
| Brachyura spp. | 3 | Hines 1982, small crabs |
| Unidentified shrimp | 0.11 | McKinney et al. 2004, *Palaemonetes pugio* |
| *Polyorchis penicillatus* | 0.01 | Båmstedt and Martinussen 2015, *Bolinopsis infundibulum* |
| *Mitrocoma cellularia* | 0.01 | Båmstedt and Martinussen 2015, *Bolinopsis infundibulum* |
| *Pleurobrachia bachei* | 0.01 | Båmstedt and Martinussen 2015, *Bolinopsis infundibulum* |
| *Bolinopsis infundibulum* | 0.01 | Båmstedt and Martinussen 2015 |
| *Evasterias troschelii* | 66.5 | O’Clair and Rice 1985 |
| *Leptasterias hexactis* | 5.5 | Menge 1975, *Leptasterias* spp*.* |
| *Leptasterias* spp*.* | 5.5 | Menge 1975, *Leptasterias* spp*.* |
| *Orthasterias koehleri* | 66.5 | O’Clair and Rice 1985, *Evasterias troschelii* |
| *Pisaster brevispinus* | 146.18 | Peters et al. 2019, *Pisaster giganteus* |
| *Pisaster ochraceus* | 128 | Sanford 2002 |
| *Pycnopodia helianthoides* | 0.018\*size^3.13 | Lee et al. 2016 |
| *Stylasterias forreri* | 66.5 | O’Clair and Rice 1985, *Evasterias troschelii* |
| *Patiria miniata* | 26.97 | Peters et al. 2019 |
| *Henricia pumila* | 10 | Menge 1975, *Henricia* spp. |
| *Henricia* spp*.* | 10 | Menge 1975 |
| *Dermasterias imbricata* | 92 | Montgomery 2014 |
| *Mediaster aequalis* | 10 | Menge 1975, *Henricia spp*. |
| *Solaster dawsoni* | 486 | Montgomery 2014, *Solaster stimpsoni* |
| *Solaster stimpsoni* | 486 | Montgomery 2014 |
| *Pteraster tesselatus* | 10 | Menge 1975, *Henricia* spp*.* |
| *Mesocentrotus franciscanus* | 29.51 | Schuster and Bates 2023 |
| *Strongylocentrotus droebachiensis* | 20 | Stewart et al. 2015, *Strongylocentrotus polyacanthus* |
| *Strongylocentrotus purpuratus* | 20 | Stewart et al. 2015, *Strongylocentrotus polyacanthus* |
| *Apostichopus californicus* | 319.31 | Peters et al. 2019, *Apostichopus parvimensis* |
| *Chlamys hastata* | 2.5 | MacDonald et al. 1991, *Chlamys* spp*.* |
| *Crassadoma gigantea* | 0.038\*size^2.39 | MacDonald et al. 1991 |
| *Enteroctopus dofleini* | 137.5 | Osborn 1995, *Octopus rubescens* |
| *Octopus rubescens* | 80 | Osborn 1995 |
| *Opalia wroblewskyi* | 0.91 | Palmer 1982, *Nucella* spp*.* |
| *Diodora aspera* | 0.91 | Palmer 1982, *Nucella* spp*.* |
| *Megathura crenulata* | 0.91 | Palmer 1982, *Nucella* spp*.* |
| *Haliotis kamtschatkana* | 0.00058\*size^3.2 | Zhang et al. 2007 |
| *Neverita lewisii* | 0.91 | Palmer 1982, *Nucella* spp*.* |
| *Ceratostoma foliatum* | 0.91 | Palmer 1982, *Nucella* spp*.* |
| *Nucella lamellosa* | 0.91 | Palmer 1982, *Nucella* spp*.* |
| *Armina californica* | 0.54 | McKinney et al. 2004, gastropods |
| *Cadlina luteomarginata* | 0.54 | McKinney et al. 2004, gastropods |
| *Cadlina modesta* | 0.54 | McKinney et al. 2004, gastropods |
| *Cadlina sylviaearleae* | 0.54 | McKinney et al. 2004, gastropods |
| *Coryphella verrucosa* | 0.54 | McKinney et al. 2004, gastropods |
| *Dendronotus iris* | 0.54 | McKinney et al. 2004, gastropods |
| *Dirona albolineata* | 0.54 | McKinney et al. 2004, gastropods |
| *Dirona pellucida* | 0.54 | McKinney et al. 2004, gastropods |
| *Diaulula odonoghuei* | 0.54 | McKinney et al. 2004, gastropods |
| *Diaulula sandiegensis* | 0.54 | McKinney et al. 2004, gastropods |
| *Peltodoris nobilis* | 0.54 | McKinney et al. 2004, gastropods |
| *Doris montereyensis* | 0.54 | McKinney et al. 2004, gastropods |
| *Doris odhneri* | 0.54 | McKinney et al. 2004, gastropods |
| *Antiopella fusca* | 0.54 | McKinney et al. 2004, gastropods |
| *Hermissenda crassicornis* | 0.54 | McKinney et al. 2004, gastropods |
| *Acanthodoris hudsoni* | 0.54 | McKinney et al. 2004, gastropods |
| *Acanthodoris nanaimoensis* | 0.54 | McKinney et al. 2004, gastropods |
| *Onchidoris bilamellata* | 0.54 | McKinney et al. 2004, gastropods |
| *Limacia cockerelli* | 0.54 | McKinney et al. 2004, gastropods |
| *Polycera tricolor* | 0.54 | McKinney et al. 2004, gastropods |
| *Triopha catalinae* | 0.54 | McKinney et al. 2004, gastropods |
| *Triopha modesta* | 0.54 | McKinney et al. 2004, gastropods |
| *Triopha* spp*.* | 0.54 | McKinney et al. 2004, gastropods |
| *Melibe leonina* | 0.54 | McKinney et al. 2004, gastropods |
| *Tritonia festiva* | 0.54 | McKinney et al. 2004, gastropods |
| *Acmaea mitra* | 0.91 | Palmer 1982, *Nucella* spp*.* |
| *Lottia scutum* | 0.91 | Palmer 1982, *Nucella* spp*.* |
| *Berthella chacei* | 0.91 | Palmer 1982, *Nucella* spp*.* |
| *Calliostoma ligatum* | 0.91 | Palmer 1982, *Nucella* spp*.* |
| *Tegula funebralis* | 0.91 | Palmer 1982, *Nucella* spp*.* |
| *Pomaulax gibberosus* | 31 | Schuster and Bates 2023 |
| *Eurylepta leoparda* | 0.54 | McKinney et al. 2004, gastropods |

**Table S1.04**. Akaike’s Information Criterion (AIC) values calculated for each model of ammonium concentration in relation to animal abundance (AA) or animal biomass (AB), Shannon diversity (SHD) or Simpson diversity (SID), tidal exchange rate (T), depth (D), and an interaction term. RE = random effect of both site and year. k is the number of parameters in the model. The model with the lowest AIC score is the “best” model; ΔAIC is the difference in AIC score between a given model and the “best” model; AIC weight represents the probability that a model is the best model, given the data and the set of candidate models.

|  |  |  |  |  |
| --- | --- | --- | --- | --- |
| **Predictors** | **k** | **AIC** | **ΔAIC** | **AIC weight** |
| AA + SHD + T + D + AA:T + RE | 9 | 45.60 | 0.00 | 0.50 |
| AA + SID + T + D + AA:T + RE | 9 | 46.09 | 0.50 | 0.39 |
| AB + SHD + T + D + AB:T + RE | 9 | 49.70 | 4.10 | 0.06 |
| AB + SID + T + D + AB:T + RE | 9 | 49.98 | 4.38 | 0.06 |

**Table S1.05**. Akaike’s Information Criterion (AIC) values calculated for each model of delta ammonium concentration in vs outside kelp forests in relation to animal abundance (AA) or animal biomass (AB), Shannon diversity (SHD) or Simpson diversity (SID), kelp species (KS), kelp biomass (KB), tidal exchange rate (T), depth (D), and three interaction terms. RE = random effect of site. k is the number of parameters in the model. The model with the lowest AIC score is the “best” model; ΔAIC is the difference in AIC score between a given model and the “best” model; AIC weight represents the probability that a model is the best model, given the data and the set of candidate models.

|  |  |  |  |  |
| --- | --- | --- | --- | --- |
| **Predictors** | **k** | **AIC** | **ΔAIC** | **AIC weight** |
| AB + SHD + KS + KB + T + D + AB :T + AB :KB + KB:T + RE | 13 | -33.16 | 0.00 | 0.30 |
| AA + SHD + KS + KB + T + D + AA:T + AA:KB + KB:T + RE | 13 | -32.89 | 0.27 | 0.26 |
| AA + SID + KS + KB + T + D + AA:T + AA:KB + KB:T + RE | 13 | -32.80 | 0.36 | 0.25 |
| AB + SID + KS + KB + T + D + AB:T + AB:KB + KB:T + RE | 13 | -32.38 | 0.78 | 0.20 |

**Table S1.06.** Excretion rate model to determine log transformed NH₄⁺ excretion rate (uM/hour/L) for California sea cucumbers (*Apostichopus californicus*) based on size index: sqrt(length\*girth). Adjusted R-squared for this model is 0.39.

|  |  |  |  |  |
| --- | --- | --- | --- | --- |
|  | **Estimate** | **Std. error** | **t value** | **p value** |
| Intercept | 1.40 | 0.22 | 6.23 | < 0.001 |
| Size index | 0.05 | 0.009 | 5.52 | < 0.001 |

**Table S1.07.** Excretion rate model to determine log transformed NH₄⁺ excretion rate (uM/hour/L) for red rock crabs (*Cancer productus*) based on carapace width (mm). Adjusted R-squared for this model is 0.82.

|  |  |  |  |  |
| --- | --- | --- | --- | --- |
|  | **Estimate** | **Std. error** | **t value** | **p value** |
| Intercept | 1.22 | 0.32 | 3.76 | 0.002 |
| Carapace | 0.02 | 0.003 | 8.73 | < 0.001 |

**Table S1.08.** Meso-scale linear mixed-effect model describing drivers of among-site variability in ammonium concentration. The model was constructed with a gamma distribution (link = ‘log’), so coefficients are presented in log space. Continuous predictors were centred and scaled to compare effect sizes between predictors with varying units.

|  |  |  |  |  |
| --- | --- | --- | --- | --- |
|  | **Estimate** | **Std. error** | **z value** | **p value** |
| Intercept | -0.61 | 0.25 | -2.48 | 0.01 |
| Abundance | -0.07 | 0.12 | -0.55 | 0.58 |
| Tidal exchange | 0.00 | 0.08 | 0.00 | 1.00 |
| Biodiversity | -0.10 | 0.12 | -0.80 | 0.42 |
| Depth | 0.05 | 0.09 | 0.51 | 0.61 |
| Abundance:tide | -0.25 | 0.10 | -2.54 | 0.01 |

**Table S1.09**. Small-scale linear mixed effect model describing drivers of ammonium concentration inside – outside kelp forests. Continuous predictors were centred and scaled. Kelp species is a categorical predictor with three levels: macro = *Macrocystis pyrifera* (intercept level), nereo = *Nereocystis luetkeana*, none = no kelp control.

|  |  |  |  |  |
| --- | --- | --- | --- | --- |
|  | **Estimate** | **Std. error** | **z value** | **p value** |
| Intercept | 0.16 | 0.03 | 5.91 | < 0.001 |
| Kelp nereo | 0.24 | 0.07 | 3.64 | < 0.001 |
| Kelp none | -0.40 | 0.10 | -4.13 | < 0.001 |
| Kelp biomass | 0.29 | 0.05 | 5.95 | < 0.001 |
| Tidal exchange | 0.06 | 0.02 | 2.25 | 0.02 |
| Animal biomass | 0.05 | 0.03 | 1.64 | 0.10 |
| Biodiversity | -0.03 | 0.02 | -1.20 | 0.23 |
| Depth | 0.04 | 0.03 | 1.31 | 0.19 |
| Kelp:tide | 0.29 | 0.08 | 3.85 | < 0.001 |
| Kelp:animals | -0.05 | 0.02 | -2.08 | 0.04 |
| Tide:animals | -0.13 | 0.04 | -3.19 | 0.001 |

**Table S1.10**. Fine-scale linear model describing ammonium variation between cages with 0, 1, or 2 California sea cucumbers (*Apostichopus californicus)*.

|  |  |  |  |  |
| --- | --- | --- | --- | --- |
|  | **Estimate** | **Std. error** | **z value** | **p value** |
| Intercept | 0.91 | 0.07 | 12.43 | < 0.001 |
| Cukes - one | 0.03 | 0.10 | 0.31 | 0.76 |
| Cukes - two | 0.00 | 0.10 | 0.01 | 0.99 |
| Depth | 0.38 | 0.05 | 8.07 | < 0.001 |

**Table S1.11.** Fine-scale model describing ammonium variation between cages with zero, medium, or large red rock crabs (*Cancer productus*). The model was constructed with a gamma distribution (link = ‘log’), so coefficients are presented in log space.

|  |  |  |  |  |
| --- | --- | --- | --- | --- |
|  | **Estimate** | **Std. error** | **z value** | **p value** |
| Intercept | -1.78 | 0.25 | -7.25 | < 0.001 |
| Crabs - medium | 2.20 | 0.26 | 8.48 | < 0.001 |
| Crabs - large | 2.61 | 0.27 | 9.83 | < 0.001 |

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