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Blue Nitrogen: Global Rates and Economic Importance

9 **Ziyan Wang^{1*}, Benoit Thibodeau^{1,2*}**

10 ¹The School of Life Sciences, The Chinese University of Hong Kong, New Territories, Shatin,
11 Hong Kong SAR, China

12 ²Department of Earth and Environmental Sciences, The Chinese University of Hong Kong, New
13 Territories, Shatin, Hong Kong SAR, China

14

15 Corresponding authors: Ziyan Wang (1155186101@link.cuhk.edu.hk); Benoit Thibodeau
16 (benoit.thibodeau@cuhk.edu.hk)

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18 **Key Points:**

- 19 • Global mangroves currently remove 870 Gg N annually, with the highest rates occurring
20 in deltaic mangrove systems.
- 21 • Removal of nitrogen by mangroves is highly vulnerable to global environmental changes.
- 22 • “Blue Nitrogen” credits offer a transformative opportunity to advance mangrove
23 conservation and improve coastal water quality.

24 **Abstract**

25 Nitrogen (N) pollution drives widespread coastal ecosystem collapse. In this paper, we argue
26 that mangrove forests represent an undervalued natural mitigation solution for N pollution. By
27 performing a comprehensive global meta-analysis, we reveal that mangroves remove 870 Gg N
28 annually, which represents an economic value reaching \$8.7 billion via N credit-based
29 valuation. This value is more than 12 times higher than the annual carbon credits value derived
30 from mangrove carbon sequestration. Despite the substantial economic importance of
31 mangroves' N-removal capacity, it remains largely overlooked in current conservation
32 frameworks. Moreover, we argue that N removal by mangroves has the potential of removing
33 over 5 million tonnes of N each year under optimal conditions. We also highlight critical
34 environmental thresholds demonstrating that this service is vulnerable: extreme
35 eutrophication, rising mean annual temperature above 22°C, and hypersalinity all suppress
36 mangrove N removal capacity. Our findings provide the scientific foundation to integrate “Blue
37 Nitrogen” credits in a larger N-credit framework, offering a transformative financing mechanism
38 for coastal water quality management that could reshape both mangrove conservation and N
39 pollution mitigation strategies, worldwide.

40 **Plain Language Summary**

41 While coastal ecosystems are stressed by excessive nitrogen (N) loads, key habitats within
42 them, such as mangroves, provide a powerful nature-based solution. Our study establishes the
43 first global benchmarks for mangrove N removal, revealing that mangroves remove 870 Gg N
44 annually, representing an economic value of \$8.7 billion per year based on N credit valuation.
45 This figure could be substantially higher under optimal conditions, highlighting the immense
46 untapped potential of mangroves in N mitigation. These findings provide a compelling case for
47 the establishment of a new "Blue Nitrogen" conservation market, which could revolutionize
48 both mangrove conservation and N pollution mitigation strategies worldwide by offering a
49 transformative financing mechanism for coastal water quality management.

50 **1 Introduction**

51 Nitrogen (N) release from human activities has increased by up to 50-fold since the early 1900s
52 (Battye et al., 2017; Galloway et al., 2004) and may double in China between 2010 and 2050
53 under a business-as-usual scenario (Gu et al., 2015). Nearly all of this anthropogenic N is
54 reactive—biologically available and actively participating in environmental and biogeochemical
55 processes—altering ecosystems worldwide (Galloway & Cowling, 2002). This reactive N triggers
56 a cascade of well-documented ecological crises in coastal and estuarine ecosystems.

57 Eutrophication—the over-enrichment of waters with nutrients, leading to excessive plant and
58 algal growth—fuels harmful algal blooms and, as this organic matter decomposes, depletes
59 oxygen to create hypoxic "dead zones." These processes cause substantial economic losses in
60 fisheries and aquaculture (Breitburg et al., 2009; Galloway et al., 2004; Jessen et al., 2015). The
61 severe economic damage of eutrophication is globally recognized, with annual losses in the
62 billions of dollars consistently reported across diverse watersheds in regions such as Europe
63 (HELCOM, 2009), the United States of America (Dodds et al., 2009), and China (Gu et al., 2015;
64 Strokal et al., 2020). Furthermore, elevated nitrate (NO_3^-) levels in water sources pose risks to
65 human health, including "blue baby syndrome" and thyroid disorder (World Health
66 Organization, 2022). Consequently, these widespread impacts on both ecosystem integrity and
67 public health have prompted nations worldwide to implement costly coastal nutrient reduction
68 programs (Boesch, 2002; Greening & Elfring, 2002) or investigate the potential of other
69 mitigation alternatives (Chen et al., 2025; Gu et al., 2023).

70 Conventional engineered solutions to N pollution, such as wastewater treatment, impose a
71 substantial financial burden. Examples from the United States of America show a growing \$81
72 billion annual funding gap for water infrastructure (American Society of Civil Engineers, 2021),
73 alongside a doubling of residential wastewater bills in the past decade (American Society of
74 Civil Engineers, 2025). This funding gap highlights the urgent need for innovative and cost-
75 effective alternatives to avoid further environmental damages.

76 Amidst this global challenge, mangrove forests offer a powerful nature-based solution. Their
77 unique, mostly anaerobic sediment fosters microbial processes like denitrification and

78 anaerobic ammonium oxidation (anammox) that transform reactive N back to N_2 (Adame &
79 Lovelock, 2011; Fernandes et al., 2016). Other processes, such as assimilation into biomass and
80 burial in sediments, also remove reactive N from coastal waters (e.g., Rivera-Monroy & Twilley,
81 1996). However, these pathways do not completely remove the reactive N from the ecosystem;
82 buried or assimilated N may eventually be remineralised and re-released into the environment.
83 Similarly, NO_3^- can also be reduced by dissimilatory nitrate reduction to ammonium. However in
84 this case ammonium (NH_4^+) is retained in the ecosystem and can be assimilated by other
85 organisms (Burgin & Hamilton, 2007). In contrast, denitrification and anammox convert reactive
86 N to its least bioavailable form (N_2), effectively removing it from the biologically active pool on
87 timescales relevant to coastal management.

88 Denitrification, the anaerobic reduction of NO_3^- to nitrous oxide (N_2O) and N_2 is generally the
89 primary process responsible for N removal in mangroves (e.g., Reis et al., 2017). This process is
90 thought to be fueled by the amount of available NO_3^- and organic carbon (OC) (Kraft et al.,
91 2014; Rivera-Monroy & Twilley, 1996). Anammox can also contribute to N removal by
92 converting NH_4^+ and nitrite (NO_2^-) directly into N_2 , partly decoupled from the availability of OC
93 (Fernandes et al., 2012; Meyer et al., 2005). However, despite the increasing recognition of
94 mangroves for their capacity to buffer coastal waters from N pollution (Adame et al., 2019; Lee
95 et al., 2009; Wu et al., 2008), a robust global assessment is lacking.

96 Barriers to such an assessment are partly methodological and stem from the conflation of two
97 distinct measurement approaches —actual and potential rate assays. Actual rate
98 measurements (e.g., intact sediment cores, benthic chambers), capture realized N
99 transformation rates under real-world constraints, including redox gradients, substrate
100 limitation, and diffusional barriers (Alongi et al., 2000; Trimmer et al., 2006; Whigham et al.,
101 2009). Conversely, potential rate assays use homogenized slurries under optimal conditions
102 (e.g., strict anoxia, abundant substrate, and thorough mixing), thereby revealing the maximum
103 functional capacity of microbial communities (Amano et al., 2011; Dalsgaard et al., 2005).
104 However, the distinction between these two methods makes it difficult to identify the specific
105 factors that govern net N-removal rate versus potential rates (Alongi, 2020; Reis et al., 2017).

106 An additional barrier is the lack of region-specific rate estimation. While biogeomorphic settings
107 of mangroves have successfully been used to refine global estimates of services like C
108 sequestration (Breithaupt & Steinmuller, 2022), a similar systematic analysis for mangrove N
109 removal has yet to be realized. It remains largely unknown how N removal varies across distinct
110 coastal settings, primarily deltaic, estuarine, open-coast and lagoonal (Worthington et al.,
111 2020). This lack of a spatially explicit framework hinders rigorous upscaling estimates of the
112 global N-removal service.

113 Accurately quantifying global N removal is a critical step toward unlocking new conservation
114 opportunities for mangroves. As demonstrated by “Blue Carbon” science, valuing ecosystem
115 services can powerfully drive conservation efforts (Costanza et al., 1997; Griscom et al., 2017;
116 Mcleod et al., 2011; Zhang et al., 2025). Likewise, when properly quantified, N removal by
117 coastal wetlands represents a valuable ecosystem service that should be incorporated into
118 conservation decision-making and policy frameworks (Daily et al., 2009). This approach can help
119 coastal regions advance toward the goal of N neutrality — a state of zero net reactive N release
120 to the environment (Chen et al., 2025; Leip et al., 2014).

121 Our objective is therefore to provide a clear assessment of mangrove N removal through a
122 global compilation and meta-analysis of actual and potential rates, and to identify their
123 respective drivers. Moreover, we provide a basic framework for the economic valuation of N
124 removal by mangroves based on the established economic benchmarks for N reduction.

125 **2 Methods**

126 **2.1 Meta-analysis**

127 We conducted a bibliographic search based on the Web of Science database using the search
128 terms “mangrove” AND (“denitrification” OR “anammox” OR “nitrate reduction” OR “nitrogen
129 removal” OR “nitrogen loss”), for articles published up to April 2025. This search yielded 924
130 papers, which were manually screened to retain only those studies reporting empirical N
131 removal rates from fresh mangrove sediment and ambient conditions. Reference lists were also
132 checked for additional studies. The final dataset for our meta-analysis comprised 51 published

133 studies (see Figure S1 for screening procedure) and was augmented with an unpublished
134 dataset from our own ongoing research (Wang and Thibodeau, 2026). We note that our dataset
135 is comprised of 28 mangrove forests in Asia, 6 in the North and Central America/Caribbean
136 region, 4 in Oceania, 3 in Africa, and 1 in South America. While Africa and South America
137 together represent 35% of global mangrove areas (Jia et al., 2023), they contribute only 8% of
138 the compiled sites in our study. Therefore, their importance is diminished in this meta-analysis
139 and thus more data from these parts of the world would improve the reliability of our analysis.

140 We created a dataset distinguishing between two fundamental metrics: (1) actual rates (n = 54),
141 measured in intact sediment cores using ^{15}N labeling, $\text{N}_2:\text{Ar}$ ratio, or acetylene inhibition
142 techniques; and (2) potential rates (n = 99), measured by in homogenized sediment slurries
143 using ^{15}N labeling or acetylene inhibition techniques. Our data compilation revealed that
144 studies using the N_2 gas flux technique report systematically higher and more variable actual
145 rates than other methods (Figure S2). In line with previous work identifying this as a
146 methodological artifact (Robert Hamersley & Howes, 2005), these studies were therefore
147 excluded to avoid overestimating true N removal. Throughout this study, total N removal is
148 defined as the sum of denitrification and anammox rates. This meta-analysis therefore
149 establishes a lower bound for total mangrove N removal, as it incorporates historical methods
150 (e.g., early ^{15}N labeling, acetylene inhibition techniques) that did not detect anammox
151 inherently and thus reflect denitrification alone. We explicitly address and quantify the
152 potential magnitude of this underestimation in our Results and Discussion (Section 3.1).

153 We standardized actual rates to areal units ($\mu\text{mol N m}^{-2} \text{h}^{-1}$, where h represents hour) and
154 potential rates to volumetric units ($\text{mmol N m}^{-3} \text{h}^{-1}$), using either the reported sediment density
155 or a representative value of 1.88 g cm^{-3} to convert mass-based rates when necessary (Hou et al.,
156 2015). Furthermore, to enable a direct comparison, we calculated a conservative areal flux from
157 volumetric potential rates by normalizing them over a 1-cm active sediment depth. The top 1-
158 cm depth reflects the typical shallow suboxic zone in organic-rich mangrove sediments where
159 denitrification is most active (Fernandes et al., 2012; Meyer et al., 2005).

160 We extracted a suite of key environmental variables and classified sites by coastal
161 environmental setting, ambient trophic status, and sediment type. We sourced the parameters
162 from the text, tables, and supplementary materials of each publication, with graphical data
163 extracted using WebPlotDigitizer. A complete description of all variables is provided in Text S1
164 and Table S1.

165 To upscale our N-removal rates globally, we used setting-specific mean rates to the published
166 area estimates for four coastal environmental settings: deltaic (54,972 km²), estuarine (37,411
167 km²), open coast (28,493 km²), and lagoonal (14,993 km²), summing to a total mangrove area of
168 135,869 km² (Worthington et al., 2020). We acknowledge that our calculations do not quantify
169 all uncertainties and methodological bias, as our approach averages seasonal and vertical scales
170 (see Text S3 for a detailed analysis). Nevertheless, it provides a robust first estimate of the
171 mean annual N-removal service provided by the world's mangroves.

172

173 **2.2 Statistical analysis**

174 All statistical analyses and data visualizations were performed in R (v. 4.3.0). As the raw data
175 did not meet normality assumptions (Shapiro-Wilk test, $p < 0.05$), a \log_{10} transformation was
176 applied. The transformed means and 95% confidence intervals were calculated and are
177 presented as back-transformed values, referred here to as “adjusted means” (Breithaupt &
178 Steinmuller, 2022). For comparison, medians and arithmetic means from the untransformed
179 data were also reported. To compare N-removal rates between groups, we used appropriate
180 parametric (one-way ANOVA with Tukey's post-hoc tests) or non-parametric (Kruskal-Wallis
181 with Dunn's post-hoc tests) tests.

182 Linear mixed effects models were used to evaluate relationships between N removal and
183 environmental parameters (Figure S3-S5). To further identify the key environmental drivers of
184 potential N removal and capture non-linear relationships, we developed a random forest model
185 using the R package randomForest (Liaw & Wiener, 2002). The model's predictive performance
186 was validated using 10-fold cross-validation. We assessed predictor importance based on the

187 percent increase in mean squared error (%IncMSE) and visualized the marginal effects of the
188 most influential variables using partial dependence plots (PDPs) via the pdp package. Full
189 details of the model parameters are provided in Text S2. Given the derived nature of the C:N
190 ratio and heterogeneous measurement protocols across studies, a traditional collinearity
191 assessment (e.g., variance inflation factor analysis, VIF) on the full predictor set was not
192 feasible. A reduced VIF analysis confirmed no severe multicollinearity among the core variables
193 (Table S4).

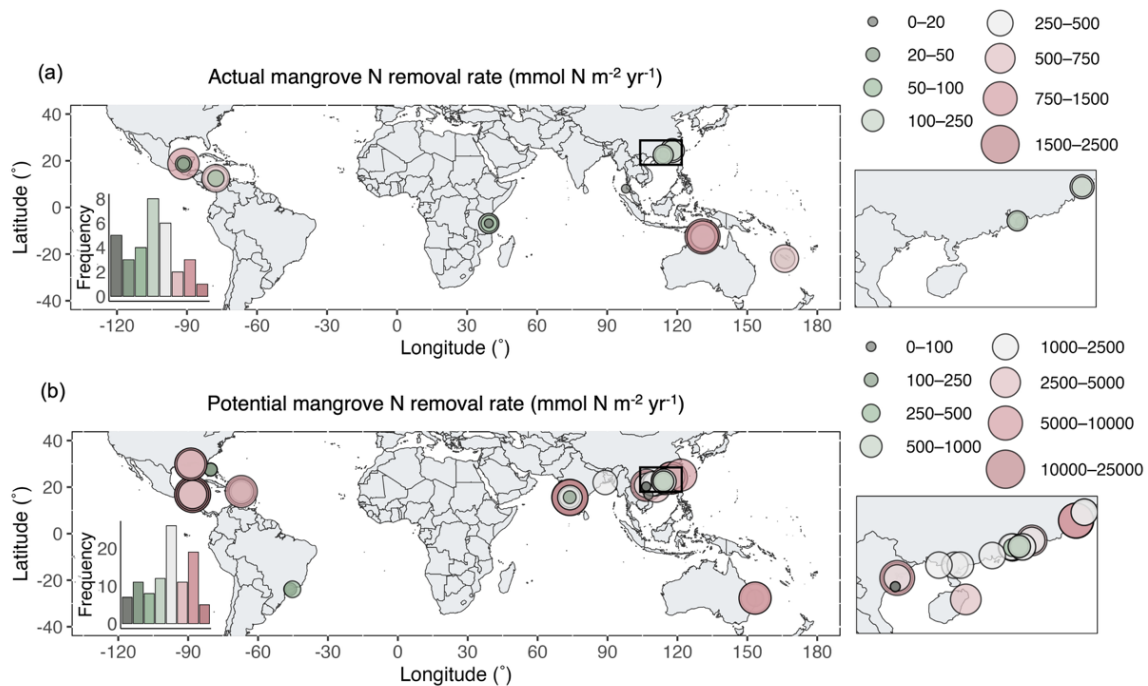
194 **2.3 Ecosystem service valuation**

195 We estimated the economic value of mangrove N removal using a market-based credit
196 approach. We adopted a value of \$ 10,053 per ton of N (in 2022 USD), derived from the
197 successful Connecticut Nitrogen Credit program (Dykes, 2022). This specific value was selected
198 because it represents a conservative median from our review of representative valuation
199 methods (Table S2) and is also grounded in a mature, policy-relevant market, ensuring its
200 applicability.

201 To provide a direct and methodologically consistent comparison for our annual N-removal
202 valuation, we developed a tailored estimate for the annual service of mangrove carbon
203 sequestration that aligns with the specific coastal geomorphic settings of our analysis – a
204 refinement absent from most previous “blue carbon” valuation (Pendleton et al., 2012; Zeng et
205 al., 2021; Zhang et al., 2025). Our valuation is based on biogeomorphology-specific C
206 sequestration rates (Breithaupt & Steinmuller, 2022), a CO₂ equivalence conversion factor of
207 3.67 (Zhang et al., 2025), and an average voluntary carbon market price of \$ 6.30 per ton of
208 CO₂e from Forest Trends’ Ecosystem Marketplace reports in 2020-2024
209 (<https://ecosystemmarketplace.com>).

210 **3 Results and Discussion**

211 **3.1 Global compilation of mangrove N removal**



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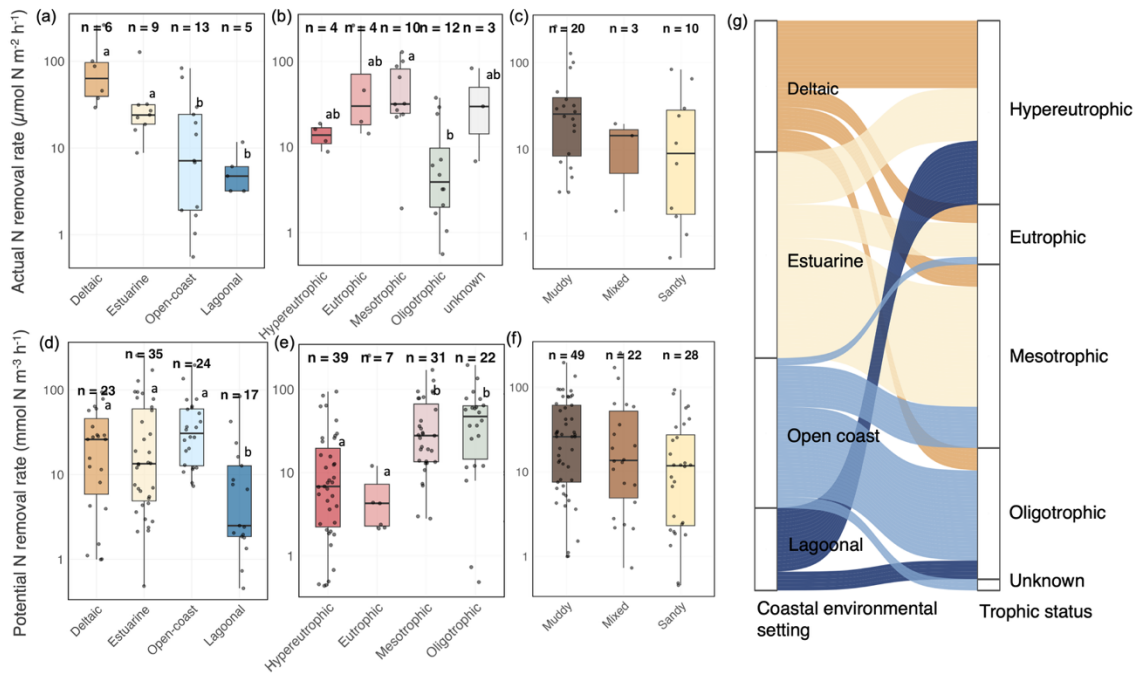
213 **Figure 1.** Global distribution and magnitude of mangrove N removal. Maps show (a) site-specific
214 actual rates, (b) site-specific potential rates. Inset histograms illustrate the frequency
215 distribution of the sites across calculated value categories.

216 Our global meta-analysis provides a new and robust baseline for mangrove N removal based on
217 data from 33 actual and 99 potential rate studies (Figure 1). The adjusted mean of actual N-
218 removal rates was 14.80 $\mu\text{mol N m}^{-2} \text{h}^{-1}$ (95% CI: 8.97–24.44), derived from log-transformed
219 values to account for data skewness (Breithaupt & Steinmuller, 2022). This value is more
220 conservative than the median of 27.67 $\mu\text{mol N m}^{-2} \text{h}^{-1}$ and the arithmetic mean of 35.09 $\mu\text{mol N}$
221 $\text{m}^{-2} \text{h}^{-1}$, which were influenced by a few extremely high measurements. As expected, potential
222 N-removal rates are higher, with an adjusted mean of 140.78 $\mu\text{mol N m}^{-2} \text{h}^{-1}$ (95% CI: 103.96–
223 190.65) that is similar to the median (156.04 $\mu\text{mol N m}^{-2} \text{h}^{-1}$) but much lower than the
224 arithmetic mean (340.59 $\mu\text{mol N m}^{-2} \text{h}^{-1}$).

225 Our findings support the idea that mangroves are N-removal hotspots at the land-sea interface,
226 as their representative actual N-removal rate ($14.8 \mu\text{mol N m}^{-2} \text{h}^{-1}$) is nearly double the global
227 average for terrestrial soils ($\sim 7.9 \mu\text{mol N m}^{-2} \text{h}^{-1}$; Battye et al., 2017) and higher than the mean
228 rate observed in marine sediments ($\sim 8.7 \mu\text{mol N m}^{-2} \text{h}^{-1}$; Chang et al., 2024). Based on paired
229 measurements of denitrification and anammox, our compilation also revealed that
230 denitrification is the dominant N-removal pathway in global mangrove sediments, while
231 anammox accounts on average for 23.2% of actual and 11.7% of potential N removal. Applying
232 these proportions as a preliminary correction factor to historical studies that only measured
233 denitrification rates, we found that the total N removal may have been systematically
234 underestimated by 8% (actual) or 6% (potential).

235 Globally, we estimated that mangroves provide an annual N removal of 867 Gg N yr^{-1} (95% CI:
236 234-1503), based on an area-weighted upscaling that accounts for coastal environmental
237 settings. Notably, these actual rates represent only about 15% of the ecosystem's full latent
238 capacity. We estimate that the mangrove N-removal potential is $\sim 5670 \text{ Gg N yr}^{-1}$, a theoretical
239 capacity that could offset 1.8-3.1% of total anthropogenic reactive N created annually
240 (Galloway et al., 2021), highlighting their biogeochemical importance while occupying less than
241 0.1% of the Earth's land surface. This should be considered as a first-order estimate as we
242 acknowledge the methodological uncertainties of our dataset (Text S3). While complete
243 denitrification results in the production of N_2 , incomplete denitrification can result in the
244 formation of the potent greenhouse gas N_2O (Fernandes et al., 2010). Mangroves have been
245 considered to be mostly characterized by complete denitrification releasing N_2 (e.g., Alongi,
246 2020) which is supported by recent evidence showing that mangroves may be net N_2O sinks in
247 pristine, oligotrophic systems (Adame et al., 2025). However, in eutrophic conditions mangrove
248 could turn into a net N_2O source (e.g., Cabral et al., 2025). Data on N_2O fluxes from mangrove
249 are scarce and thus elucidating this question will be crucial in the future to fully understand the
250 potential trade-off from N-removal.

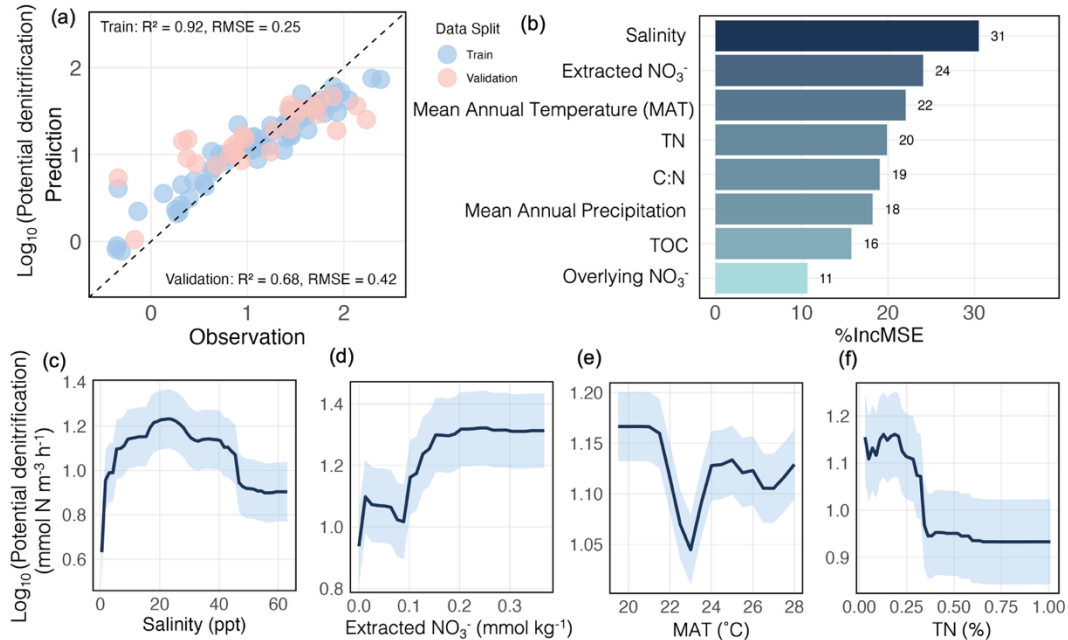
251 **3.2 Decoupled controls on N removal**



252 **Figure 2.** Actual and potential N-removal rates in global mangroves. Grouped by coastal
 253 environmental settings (a, d); surrounding water trophic status (b, e); sediment types within
 254 mangrove forests (c, f). Boxplots display the median (line), interquartile range (box), and
 255 individual data points. Different letters denote statistically significant differences among groups
 256 (Kruskal-Wallis with Dunn's post-hoc tests, $p < 0.05$). (g) Distribution of coastal environmental
 257 settings across trophic status categories.
 258

259 By comparing the different types of mangroves, we observed that actual N-removal rates were
 260 highest in deltaic and estuarine systems and in eutrophic waters (Figure 2a,b), reflecting that
 261 greater nutrient concentration generally supports higher N-removal activity (Rivera-Monroy &
 262 Twilley, 1996; Statham, 2012). This linkage was illustrated by the very strong positive
 263 correlation observed between actual N-removal rates and overlying water NO_3^- concentrations
 264 across our dataset ($R = 0.94$, $p < 0.001$; Figure S3b). However, mangroves in open-coast and
 265 oligotrophic settings demonstrated a removal potential that was comparable to or even
 266 superior to their nutrient-rich counterparts (Figure 2d & e, Table S3). We also found a moderate
 267 but significant negative correlation between potential N-removal capacity and overlying NO_3^-
 268 levels throughout the dataset ($R = -0.42$, $p < 0.05$; Fig. S3e). Therefore, while actual rates are

269 strongly governed by overlying water NO_3^- concentration, potential rates appear to be
 270 determined by other factors, as NO_3^- is not limiting in these experiments.



271
 272 **Figure 3.** Predictive modeling of potential denitrification rates using a random forest model. (a)
 273 Performance metrics of the model, showing the relationship between the training and
 274 validation sets. (b) The importance (increase in mean squared error, %IncMSE) of predictors of
 275 potential denitrification rates. (c)-(f) partial dependence plots showing the marginal effect of
 276 the four most influential variables on the potential denitrification prediction. The solid line
 277 represents the mean model prediction, and the shaded area indicates the standard deviation.

278 To understand the mechanisms of N-removal potential, we focused on potential denitrification,
 279 which serves as a robust proxy for total N removal. This is supported by its dominance as the
 280 primary N-removal pathway and its strong correlation with anammox ($R = 0.7$, $p < 0.001$, Fig.
 281 S4b), a coupling consistent with broader patterns in aquatic ecosystems (Xu et al., 2025).

282 Accordingly, we developed a random forest model to predict our potential denitrification rates
 283 from a suite of abiotic factors. The model demonstrated strong predictive power, with a ten-
 284 fold cross-validation capturing 64% of the variation in global rates (cross-validated $R^2 = 0.64$).

285 Therefore, the model provides a reliable tool for predicting rates and identifying their key
286 environmental controls. The resulting variable importance analysis revealed that local
287 biogeochemical factors, especially salinity and sediment-extracted NO_3^- concentration, are the
288 most important drivers (Figure 3b).

289 The importance of salinity as a key determinant suggests a direct link to the high N-removal
290 potential observed in open-coast mangroves. We tested this link directly by examining the
291 relationship between salinity and potential N removal within each coastal environmental
292 setting (Figure S5) and confirmed a significant positive correlation in open-coast mangroves
293 ($R=0.52$, $p<0.01$). This strong salinity dependence can be interpreted through two possible
294 mechanisms. First, high salinity acts as a powerful ecological filter, potentially selecting for a
295 more specialized and efficient denitrifying microbial community (Torregrosa-Crespo et al.,
296 2023). Additionally, these high-salinity, organic-rich environments can provide a source of
297 sulfide with the constant supply of sulfate from seawater, offering a potent alternative electron
298 donor for denitrification and increasing the system's overall N removal (Cojean et al., 2020;
299 Plummer et al., 2015; Wang et al., 2024).

300 The high potential in oligotrophic systems could also be a great example of the “feast and
301 famine” ecological concept (Koch, 2001; Poindexter, 1981). Though never invited to a feast,
302 nutrient-starved communities evolve a high capacity for nutrient uptake, as a fitness strategy to
303 catch any resource pulse (Zhu & Dai, 2024). Concurrently, our model identified sediment NO_3^-
304 as a key predictor of potential rates, likely reflecting the baseline capacity of the resident
305 denitrifying community. The potential rate assay stimulates a “feast” by supplying abundant
306 NO_3^- . The resulting explosive response is therefore driven by both the high affinity and inherent
307 capacity of these famine-adapted communities. In contrast, communities in eutrophic systems
308 that have already adapted to a constant feast show a less dramatic response.

309 **3.3 Implications for coastal resilience**

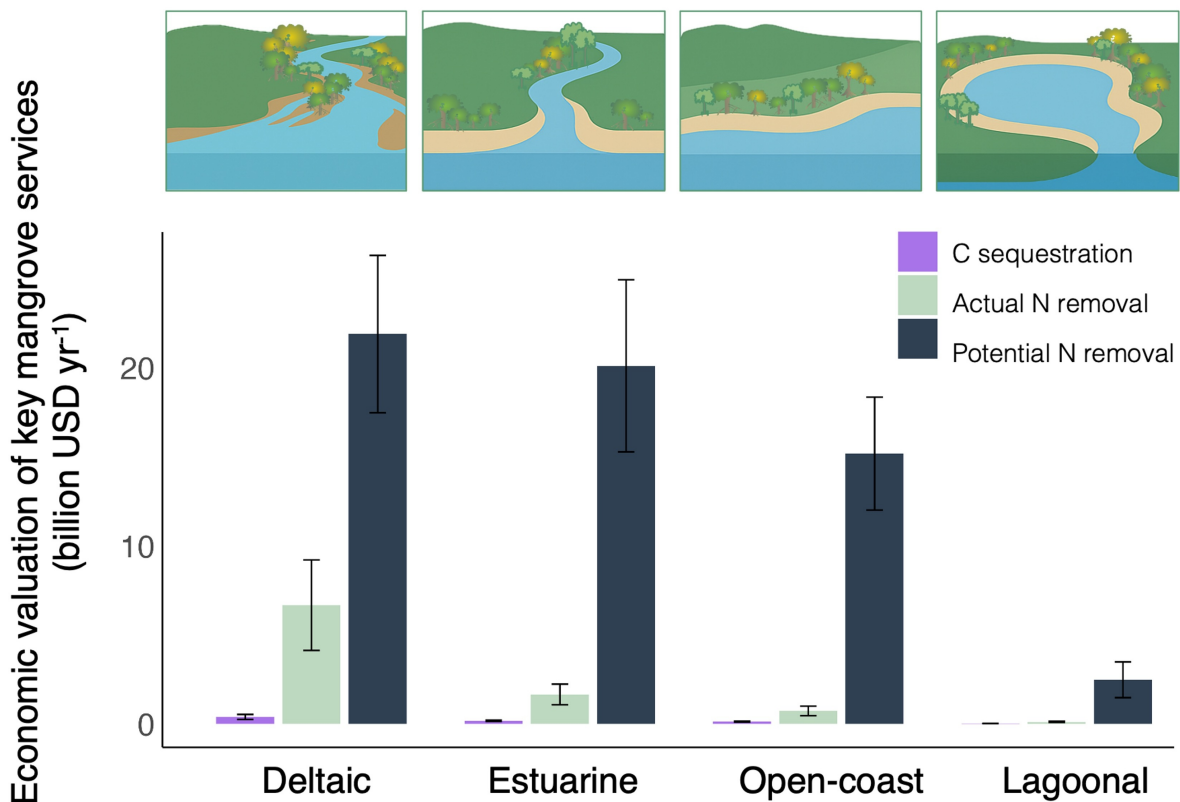
310 Our analysis reveals that N-removal response to environmental drivers is often non-linear, with
311 critical thresholds that have profound implications for mangrove functioning under future

312 global change scenarios. For instance, our observational data show rates of both actual and
313 potential N removal peak in mesotrophic waters before declining in hypereutrophic waters
314 (Figure 2b & 2e), suggesting suppression by extreme nutrient loading. This finding warns that as
315 coastal eutrophication worsens, this vital purification service by mangroves could fail precisely
316 where it is most needed.

317 Similarly, a climatic threshold is evident from actual rates being significantly lower in regions
318 with higher mean annual temperatures (MAT) (Figure S6b). Our random forest model
319 independently identified that potential denitrification could decline above a temperature of
320 around 22°C (Figure 3e). Contrary to assumptions based on the relationship between
321 temperature and metabolic rates, our model suggests that the long-term increases in MAT
322 projected by the IPCC (2023) may systematically suppress, and not enhance, the N-removal
323 capacity of mangroves, likely due to indirect effects like reduced soil moisture and smaller
324 anaerobic areas (Chen et al., 2023). This is similar to the decrease in denitrification rate and the
325 loss of microbial functions observed in seagrass meadows under marine heat waves (Bass et al.,
326 2025).

327 Further, while the effect of salinity on N removal is broadly positive, the effect at high
328 concentrations (Figure 3d, >40 ppt) suggests that hypersaline conditions may impose osmotic
329 stress that constrains the actual and potential microbial activity (Li et al., 2024), common in
330 lagoons (Figure 2a & 2d, Figure S7c) with restricted exchange (Mudge et al., 2008). This implies
331 that sea-level rise and saltwater intrusion will have complex and diverging effects, potentially
332 boosting N removal in historically fresher coastal systems while pushing already saline lagoons
333 beyond a critical stress point.

334 **3.4 Economic analysis and outlook for N neutrality**



335 **Figure 4.** Economic credit valuation of annual N removal and C sequestration services in deltaic,
 336 estuarine, open-coast, and lagoonal mangroves. Bars represent the arithmetic mean of each
 337 valuation and error bars represent the 95% confidence interval.

338 Quantifying the economic value of N removal via denitrification and anammox in mangroves
 339 reveals a significant and previously unrecognized benefit of these ecosystems. We performed a
 340 market-based valuation using a N credit price of \$10,053 per ton N in 2022 USD (Dykes, 2022)
 341 and estimated the value of the actual service at \$8.7 billion annually (95% CI: 2.4-15.1). This
 342 global value is highly concentrated in deltaic mangroves (\$7.92 billion yr⁻¹) due to their large
 343 area and the highest N-removal rates. Moreover, the value of the total potential N-removal
 344 capacity reaches around \$57 billion annually, highlighting the importance of improving our
 345 understanding of the drivers of N-removal in these ecosystems. This is significant as the societal
 346 cost of N pollution by agriculture has been estimated from €35 to €230 billion per year in
 347 Europe (Gu et al., 2021).

348 In systems like deltaic and estuarine mangroves, where N removal and C sequestration are both
349 highly efficient (Figure 4), the annual economic value of N removal can be an order of
350 magnitude greater than that of carbon sequestration. Furthermore, our analysis reveals that
351 open-coast systems, which may be undervalued in a purely carbon-focused framework due to
352 lower C accumulation (Breithaupt & Steinmuller, 2022), harbour a much higher value of N
353 removal. Therefore, integrating the value of N-removal into the conservation framework may
354 contribute to the preservation of these key ecosystems, as integrated C and N managements
355 have been suggested as a potentially effective solution for China (Xu et al., 2025).

356 Our valuation provides a clear economic case for integrating N cycling into ecosystem service
357 frameworks. The merit of a robust mangrove “Blue Nitrogen” market becomes evident, offering
358 a cost-efficient pathway for coastal zones to meet water quality goals compared to engineered
359 solutions alone. By properly valuing this service, we can unlock new streams of conservation
360 finance and create powerful incentives for strategic restoration and smarter investments into N
361 neutrality and sustainable coastal development.

362 **4 Conclusions**

363 This global meta-analysis establishes the first comprehensive benchmarks of N removal in
364 mangrove sediments, quantifying actual removal at 870 Gg N yr⁻¹ and revealing a latent
365 potential up to 5,670 Gg N yr⁻¹. Our study uncovers a decoupling between the drivers of actual
366 and potential N removal. Specifically, actual N-removal rates are primarily driven by nutrient
367 loading, whereas the potential capacity is shaped by seawater chemistry and microbial
368 strategies. We also identified critical environmental thresholds for nutrients, temperature, and
369 salinity, clearly demonstrating that mangrove N-removal services are vulnerable to ongoing
370 global change. Using an N-credit valuation approach, we estimate the currently unrecognized
371 economic value of this “Blue Nitrogen” service at \$8.7 billion yr⁻¹, more than 12 times the
372 annual carbon credits value of mangrove carbon sequestration. Together, these findings lay the
373 groundwork for a “Blue Nitrogen” framework, creating a new avenue for conservation finance
374 and policy aimed at safeguarding water quality and the resilience of our coastlines.

375 **Supplementary Information**

376 Additional supporting materials are available in the Supplementary Information of this article.

377 **Open Research**

378 The data on which this article is based are available in Wang and Thibodeau (2025).

379 **Acknowledgments**

380 This work was partially supported by the Areas of Excellence Scheme; Research Grants Council
381 of Hong Kong Special Administrative Region, China (Project Reference No. AoE/P-601-23 N,
382 Earth-HK) awarded to BT. ZW was supported by a postgraduate studentship provided by The
383 Chinese University of Hong Kong. We sincerely thank Simone Schuster and one anonymous
384 reviewer for their constructive comments and Sean Crowe for his comments on an earlier
385 version of the manuscript.

386 **Conflict of Interest Disclosure**

387 The authors declare there are no conflicts of interest for this manuscript.

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