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Monitoring nitrogen pollution using isotopic composition of macrophytes in oligotrophic tropical coastal areas: A short review

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ABSTRACT

This review explores the application of nitrogen stable isotopes measurements as a tool for monitoring nitrogen inputs in coastal zones, focusing on oligotrophic tropical ecosystems. Pulsed nutrient enrichment leads to the proliferation of macroalgae, compromising the health of coral reefs and seagrass beds. Through the analysis of 52 studies, we compile isotopic signatures measured in macroalgae and seagrass, while proposing reference thresholds to identify nitrogen pollution sources in tropical areas. By standardizing these isotopic compositions, our review aims to provide useful benchmarks for researchers and managers of coastal ecosystems. The main pollution sources identified include wastewater, groundwater, aquaculture, and fertilizers, each characterized by distinct $\delta^{15}\text{N}$ ranges. Among these, wastewater and sewage show the highest isotopic values ($>10\text{‰}$), while inorganic fertilizers exhibit the lowest (down to -4.5‰). Macroalgae, particularly *Dictyota* and Rhodophyta species (e.g. *Laurencia*), were most frequently used as bioindicators due to their wide distribution and responsiveness to nutrient variation. *Thalassia testudinum* emerged as the most commonly used seagrass species across studies. However, seagrasses overall were less frequently employed than macroalgae, and typically displayed less sensitivity to nitrogen enrichment in their isotopic signatures. The review highlights practical aspects of isotopic monitoring, including macrophyte selection, caging techniques, sample preparation, and environmental variables affecting interpretation such as seasonality, depth, and hydrodynamic conditions. Caging, though under-used in seagrass studies, proved essential in areas lacking native bioindicators. Finally, we provide recommendations for future studies to enhance consistency and standardization allowing temporal and intersites comparisons. This approach offers a cost-effective and sensitive tool for detecting nutrient inputs in vulnerable tropical ecosystems.

1. Introduction

1.1. Background: Nutrient enrichment in tropical waters

The phenomenon of eutrophication is defined as an increase in organic matter in an ecosystem, mainly due to excessive phytoplankton production, and induced by anthropogenic (man-made) inputs of inorganic nutrients (Malone and Newton, 2020). Over the past century, nutrient-rich terrigenous effluents have become a major factor of the degradation of global coastal ecosystems (Malone and Newton, 2020; Andrello et al., 2021). The origins of coastal pollution have multiplied with global population growth, as 2.15 billion people now live in the

near-coastal zone and almost 900 million inhabit low-elevation coastal areas (Reimann et al., 2023). Nutrients, particularly nitrogen derived from wastewater (animal wastes, septic systems, and sewage treatment plants), agricultural activities, industrial activities, and contaminated groundwater, are the most common and significant sources of pollution in coastal waters (Howarth et al., 2000). This pollution has proven to be threat to public health by degrading water quality and creating favorable conditions for the proliferation of pathogens (e.g. *Enterococcus* spp., *Vibrio* spp.) and coral diseases such as white syndrome disease (Redding et al., 2013; Yoshioka et al., 2016; Okeyo et al., 2018; Häder et al., 2020; Santiago et al., 2024). Excessive nitrogen input (generally $>1\text{--}3\ \mu\text{M}$ DIN, compared to $<0.3\ \mu\text{M}$ in pristine reefs) may be detrimental to the growth

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and immunity of hermatypic corals by disrupting their symbiosis with zooxanthellae, reducing calcification rates, and increasing their susceptibility to diseases (McConnaughey et al., 2000; Fabricius, 2005; D'Angelo and Wiedenmann, 2014). But the effects strongly depend on the chemical form of nitrogen (e.g. NH_4^+ vs NO_3^-), the exposure duration (pulse vs chronic), and the environmental context (light, temperature, nutrient stoichiometry) (Ezzat et al., 2015; Burkepile et al., 2020; Zhao et al., 2021). It also promotes the growth of harmful algal blooms and macroalgae on weakened reefs and seagrass beds (Rogers et al., 2014; van Tussenbroek et al., 2014; Willette et al., 2014; Lapointe et al., 2017). The accumulation of nutrients in seagrass beds can lead to an increase in the load of epiphytes and blooms of ephemeral macroalgae (Nelson, 2017; Bittick et al., 2018). This process reduces light availability and water exchange, while also increasing microbial respiration through the decomposition of detritus (Burkholder et al., 2007; Kelaher et al., 2013). Additionally, it decreases the abundance of grazers, as the proliferation of epiphytes and macroalgae reduces their access to preferred food sources, degrades habitat conditions, and increases competition and exposure to predators (Park et al., 2016; Wang et al., 2021; Bass and Falkenberg, 2023). At the same time, eutrophication alters biogeochemical cycles, leading to the accumulation of toxic compounds like hydrogen sulfide (H_2S) within the anoxic sediments of seagrass beds (McGlathery et al., 2007; Mishra and Farooq, 2025). Reductions in the surface area covered by coral reefs and seagrass beds have been in sharp decline since the 1970s, of which climate change and nutrient enrichment of the oceans are the main causes (Hughes, 1994; Gardner et al., 2003; Jackson et al., 2014). The global net anthropogenic

nitrogen input (NANI) estimation has exceeded the natural rate of N fixation of the ocean, rose from 378 to 1227 $\text{kg N.km}^{-2}.\text{yr}^{-1}$ over the period 1961–2009 (Han et al., 2020).

Nitrogen exists in two naturally occurring atomic forms: nitrogen 14 (^{14}N), which contains seven protons and seven neutrons, and nitrogen 15 (^{15}N), a heavier form with an extra neutron (Fry, 2006). These two isotopes enable us to study the sources and transformations of nitrogen in the environment, by measuring nitrogen isotopic compositions of living organisms, based on ratios measurements of heavy and light isotopes (Robinson, 2001). When related to an international standard, these measurements are called “isotopic signatures”, noted “ $\delta^{15}\text{N}$ ” (expressed in ‰), and calculated as follow:

$$\delta^{15}\text{N} (\text{‰}) = \left(R_{\text{sample}} / R_{\text{standard}} - 1 \right) \times 1000$$

with R as the atomic ratio $^{15}\text{N}/^{14}\text{N}$. Atmospheric N_2 is used as a standard for $\delta^{15}\text{N}$ analyses as it provides a baseline value of 0 ‰ (Owens, 1988). Its value is based on the relative abundance of nitrogen isotopes in the atmosphere. Since ^{14}N is much more abundant than ^{15}N , the ratio is close to 0 ‰.

The different sources of nitrogen pollution in coastal ecosystems have distinct isotopic signatures, which allows their origin to be identified. This difference is explained by the specific biogeochemical processes involved in their cycle carried out by bacteria (Fig. 1). In animal waste or wastewater, nitrogen is mainly excreted in the form of urea ($\text{CO}(\text{NH}_2)_2$). During hydrolysis, the temporary increase in pH promotes the conversion of ammonium (NH_4^+) to ammonia (NH_3), a gaseous form that is easily volatilized. This process fractionates the molecules of nitrogen,

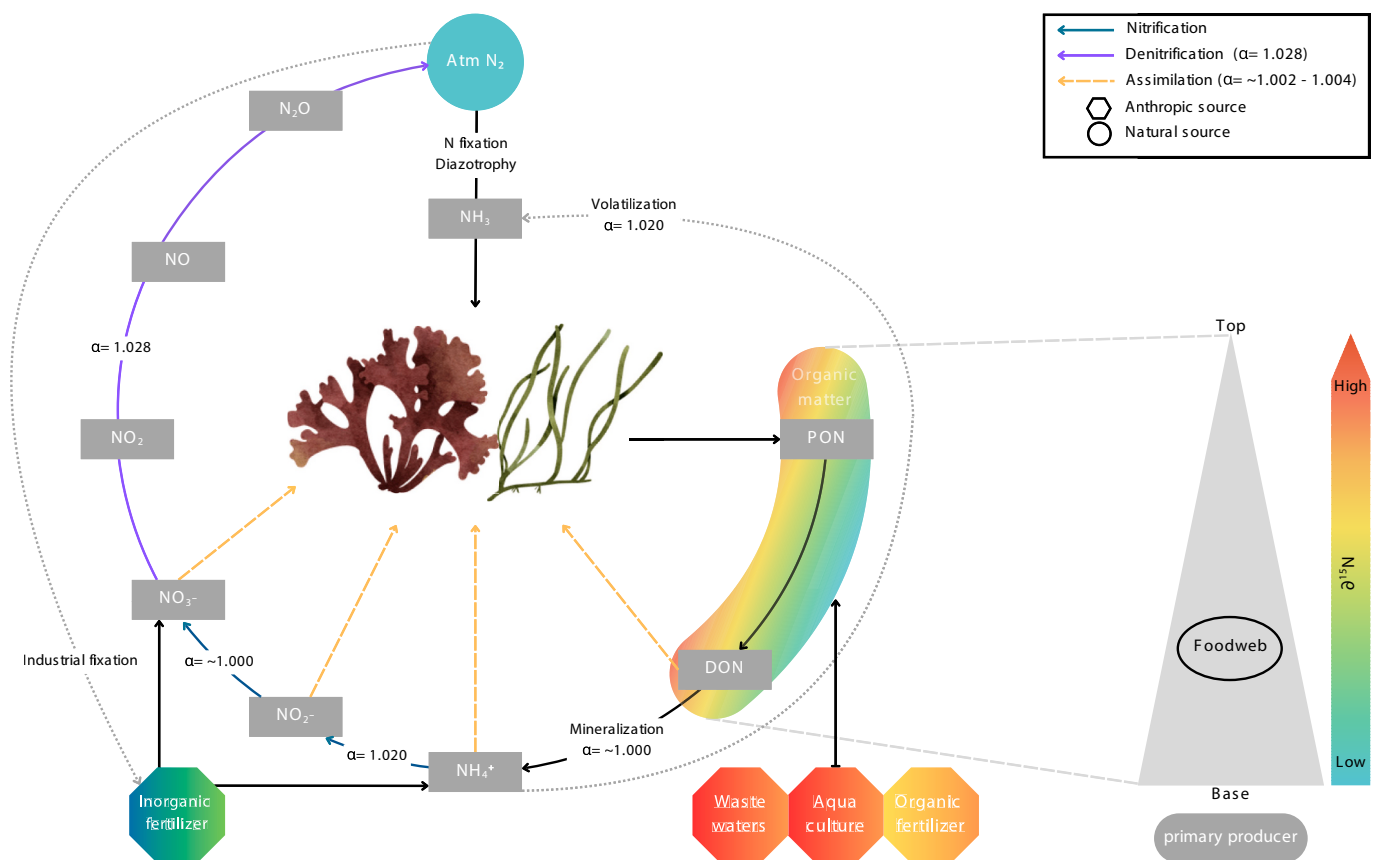


Fig. 1. Key biogeochemical processes of the nitrogen cycle in aquatic environments. The diagram illustrates nitrogen transformations and assimilation by algae and seagrass. Associated isotopic fractionation factor noted as (α) are represented. If $\alpha = 1.000$, no isotopic fractionation. If $\alpha = 1.020$ then the isotopic difference between two compounds is 20 ‰. Nitrogen circulates through the trophic chain as herbivores consume primary producers, with further transfer to higher trophic levels, where $\delta^{15}\text{N}$ increases due to isotopic fractionation. A portion of nitrogen follows detritic pathways, becoming trapped in organic matter and sediments, where it can be slowly recycled or buried over the long term (modified from Lepoint et al., 2004; Vonk et al., 2008; Swart and Evans, 2011; Hurd et al., 2014; Walton et al., 2016).

by depleting ammonia in ^{15}N , while the remaining ammonium becomes enriched in ^{15}N . This enriched ammonium is then nitrified, producing nitrates (NO_3^-) also enriched in ^{15}N , which can be easily leached and dispersed in water (Heaton, 1986). In addition, denitrification, which occurs under anaerobic conditions, plays a crucial role in reducing nitrates to gaseous nitrogen. This process tends to enrich the residual nitrogen reservoirs further in ^{15}N , accentuating the observed isotopic variations (Heaton, 1986).

Nitrogen is a fundamental nutrient for the proper growth of macrophytes, found in amino acids and enzymes (Engeland et al., 2011). Marine macrophytes are able to absorb and assimilate various forms of nitrogen from the water column into their tissues, with minimal fractionation. As a result, macrophytes isotopic compositions are similar to those of their N sources (Savage, 2005; Dudley et al., 2010), except under certain conditions where the nitrogen concentration is excessive (Denk et al., 2017). Also, the isotopic signature of N sources may change between the time they are emitted into the environment and the time they are absorbed by the macrophyte. Such as agricultural fertilizers which undergo denitrification as they pass through an anoxic marsh zone before being discharged into the sea. Therefore, the $\delta^{15}\text{N}$ value of the source at the time of absorption by the macrophytes will be higher than at the time of its release into the environment (Diebel and Zanden, 2009). The greater the number of processing steps, the more nitrogen is consumed, which ultimately results in a higher isotopic signature.

The $\delta^{15}\text{N}$ values of algae and seagrass are used worldwide to identify anthropogenic and natural sources of nitrogen in the natural environment, and to map the presence of anthropogenic nitrogen in various coastal environments (Riera et al., 2000; Umezawa et al., 2002; Fourqurean et al., 2005; Barile and Lapointe, 2005). Macroalgae and seagrass are commonly used for monitoring, due to their lower nitrogen fractionation during uptake and assimilation, except in the presence of high nitrogen concentrations (approximately 500 μM), which can induce significant isotopic shifts (Swart et al., 2014; Gröcke et al., 2017), as well as their ubiquity and ease of sampling (Costanzo et al., 2001; Lapointe et al., 2011; Swart et al., 2014; Gaubert-Boussarie et al., 2021; Page et al., 2023). Their sessile nature enables them to assimilate surrounding nitrogen, available in the environment at a precise location, to perform spatial-temporal monitoring, and also, to witness changes over time (Costanzo et al., 2005; Lemesle et al., 2016). Moreover, unlike measurements in water samples, which provide information at a given point in time (Zhang et al., 2019), macrophytes provide data on the quantity and type of nitrogen integrated over a long period of time from a few days to several weeks (Fourqurean et al., 1997; Gartner et al., 2002; Deutsch and Voss, 2006). It's an efficient and accessible tool for monitoring coastal pollution, in use since the 1990s in the tropics (Lapointe, 1997; France et al., 1998). However, its accessibility remains limited in many regions, where local management agencies often lack laboratory capacity and must rely on partnerships with research institutions to conduct isotopic analyses.

1.2. Knowledge gaps and need for threshold identification

The oligotrophic waters of tropical environments provide an interesting context regarding the nitrogen fixation by marine plants. Despite regenerated nitrogen inputs from coastal sources, its concentration remains low due to rapid biological uptake by highly productive reef ecosystems (Knapp et al., 2016; Benavides et al., 2018). This low nitrogen availability, often near the detection limits of commonly used analytical instruments (e.g. auto-analyzers, continuous-flow analyzers, or segmented-flow instruments), makes flux measurements particularly challenging in these regions. Also, their naturally low nutrient levels make them extremely sensitive to anthropogenic inputs, leading to rapid and sometimes irreversible changes in the structure of benthic communities. This particular vulnerability justifies the need to establish monitoring systems and provide the tools to interpret these changes. Moreover, tropical waters are rich in atmospheric nitrogen-fixing

organisms known as diazotrophs. These bacteria possess an enzyme capable of converting dissolved N_2 into NH_4^+ , which they then incorporate into their biomass. The nitrogen contained in the diazotrophic biomass is transformed into more bioavailable forms, such as nitrate (NO_3^-) (Knapp, 2012). These organisms play a crucial role in supplying new nitrogen to nutrient-poor ocean systems, and may cause algal blooms (Montoya et al., 2004; Montoya et al., 2007; Léger-Pigout et al., 2024).

Studies conducted on stable nitrogen isotopes in eutrophic environments are more common than those conducted in oligotrophic systems. Often focused on identifying or monitoring nutrient inputs in coastal environments, previous studies have estimated reference $\delta^{15}\text{N}$ thresholds to distinguish major pollution sources (Titlyanov et al., 2011; Dailer et al., 2012; Abaya et al., 2018b). For example, values $>10\%$ typically associated with sewage effluents, intermediate values around 4–8 ‰ linked to aquaculture inputs, and lower values (–2 to +3 ‰) reflecting atmospheric deposition or synthetic fertilizers. However, these thresholds have been derived mostly from temperate or eutrophic systems, and their applicability to oligotrophic tropical environments, where baseline $\delta^{15}\text{N}$ values and nitrogen cycling processes differ markedly, remains uncertain. There is a lack of work listing nitrogen isotopic values according to input origin, including geographical discrimination. Existing reference thresholds for identifying nitrogen input sources are not differentiated between temperate and tropical environments. Furthermore, no particular attention is paid to the choice of macrophyte species to be prioritized for monitoring in a tropical context. Depending on the species, isotopic signatures may be more sensitive due to ecophysiological aspects related to nutrient uptake and metabolism in macrophytes, including isotopic fractionation, bathymetric level, light availability, and different parts of the thallus (Teichberg et al., 2010; Jona-Lasinio et al., 2015).

The aim of this review is to provide a structured synthesis of nitrogen-isotope values reported for macrophytes (macroalgae and seagrass) in tropical coastal environments, and to identify coherent reference ranges associated anthropogenic nitrogen inputs. Rather than proposing a universal predictive model, an unrealistic goal given the context-specific nature of $\delta^{15}\text{N}$ signals, our objective is to consolidate the scattered literature into a usable framework that clarifies what is already known, highlights consistent patterns across tropical systems, and identifies where uncertainties remain. This synthesis is intended to benefit both researchers and environmental managers, including water authorities and marine-protected-area practitioners, by providing a clearer basis for interpreting $\delta^{15}\text{N}$, practical guidance on sample selection and monitoring protocols. Despite the increasing importance of anthropogenic pressures in tropical regions, no recent synthesis has systematically compiled and compared reference thresholds from a database as extensive as the one used here, filling a clear gap in the literature.

2. Methodology

2.1. Literature search strategy

The bibliographic search and selection process in this study was carried out using the PRISMA flow diagram, which is widely employed in systematic reviews to illustrate the flow of information at different stages of the review (Page et al., 2021). The diagram provides a transparent overview of the number of records identified, screened, included, and excluded, as well as the reasons for exclusion (Fig. 2). The literature search was initiated in November 2023 and no publication-year filter was applied. Data processing for this review was completed in January 2025. The keywords were entered in various combinations into search engines such as Google Scholar, Scopus, Web of Science, Research Horizon Navigator, Connected Papers and citation databases of peer-reviewed literature (Table 1). Only peer-reviewed articles in English were considered. As a result, abstracts of lectures, theses and grey

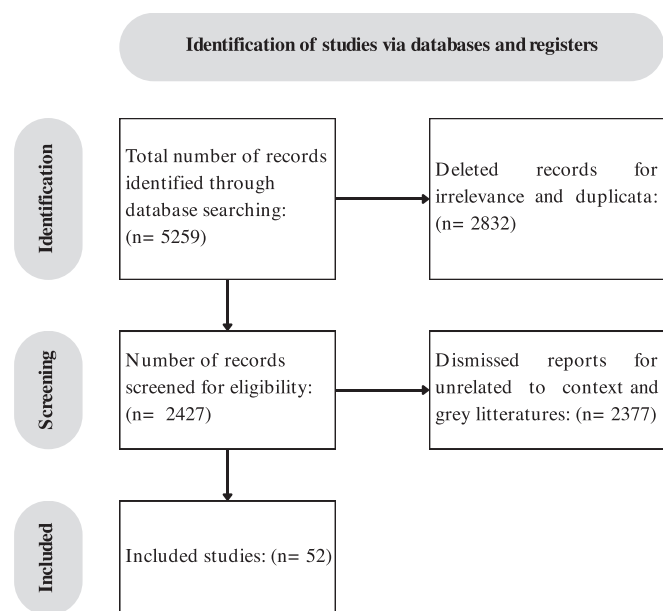


Fig. 2. PRISMA flow diagram depicting the review processes.

Table 1
Summary of bibliographic search queries and associated results.

Queries	Keyword
Q1: Macroalgae	KEY("macroalgae" OR "algae")
Q2: Seagrass	KEY("seagrass" OR phanerogam*)
Q3: Isotope	KEY("stable isotope" OR isotope* OR "stable nitrogen isotope" OR "nitrogen-15" OR "δ15N" OR "N stable isotope")
Q4: Nutrient	KEY(nutrient* OR pollution* OR loading* OR "nutrient loading" OR nitrogen* OR sewage* OR wastewater* OR eutrophication* OR "nutrient pollution" OR aquaculture* OR effluent* OR "nutrient enrichment" OR "shrimp farming" OR "Dissolved inorganic nitrogen" OR DIN OR "coastal contamination" OR "nitrogen sources" OR "sewage pollution" OR groundwater* OR fertilizer*)
Q5: Tropical	KEY("Tropical climate" OR "tropical" OR tropic*)
Q6: Monitoring	KEY(Monitoring* OR biomonitoring* OR "caging" OR "transplantation" OR "remote sensing" OR "macroalgal biomonitoring")

Queries combinations	Number of articles returned
Q1 AND Q3 AND Q4 AND Q5 AND Q6	1880
Q2 AND Q3 AND Q4 AND Q5 AND Q6	547
Q1 AND Q3 AND Q5	2260
Q2 AND Q3 AND Q5	572

literature were not taken into account. For the homogeneity of the study, the target area was the tropical zone, rated A, according to the Köppen-Geiger classification (Kottek et al., 2006; Beck et al., 2018). According to this classification, it includes any location where the average monthly temperature is above 18 °C, without a winter season, and with precipitation exceeding annual evaporation (Kottek et al., 2006).

2.2. Screening and eligibility criteria

The initial search yielded a total of 2427 articles (refers to queries Q1 and Q2 with all filter terms applied, Table 1). After removing duplicates and articles that were deemed irrelevant to the scope of this study (non-tropical, studies with uncharacterized or unspecified pollution sources, grey literatures and experimental manipulations including laboratory or mesocosm studies, as well as in situ nutrient-enrichment experiments) the final dataset comprised only 52 articles, which were thoroughly

reviewed for inclusion in the analysis (see supplementary for the list of the articles and full search string of the queries). Although the keyword "oligotrophic" was not explicitly included in the search strategy, restricting the search to tropical ecosystems (via the keyword "tropical") inherently targeted systems that are well documented as having oligotrophic baseline conditions. To ensure consistency across studies, articles were subsequently screened to verify that their study sites met operational oligotrophic criteria, i.e., systems with low ambient nutrient concentrations prior to anthropogenic enrichment, typically characterized by dissolved inorganic nitrogen (DIN) < 0.5 μM and dissolved inorganic phosphorus (DIP) < 0.1 μM when such data were reported in the original study. When nutrient data were unavailable, sites were retained only when authors explicitly described them as pristine, minimally impacted, or naturally nutrient-poor. To ensure comparability across studies, only shallow-water measurements (< 20 m) were retained.

The selected articles only concern the monitoring of pollution using the measurement of nitrogen isotopic compositions. All reviewed articles used nitrogen as an indicator of pollution in tropical areas and δ¹⁵N values are reported relative to atmospheric N₂ as the standard. All papers sufficiently elaborated their sampling method for the analysis of the data in this review.

2.3. Data extraction

Through each paper, the conditions and monitoring parameters were recorded in a spreadsheet. This brings together information on the species used, the monitoring method (with or without transplantation, acclimatization, caging), the collection methods (depth, nutrient-poor location), the exposure time to pollution sources, the transport and preservation conditions until isotopic analyses, the isotopic values, and the detected pollution. These parameters were selected because they influence the interpretation of nitrogen isotopic signatures and ensure cross-study comparability. Species identity and monitoring method allow assessment of potential methodological biases. Sampling context (depth, location, exposure time) helps evaluate environmental influences on δ¹⁵N values. Finally, preservation and analytical procedures ensure consistency in isotopic measurements. Compiling these variables enabled us to discuss the robustness of the dataset and to identify common patterns and sources of variability among studies.

2.4. Statistical analyses

All these data were recorded in a database for statistical analysis conducted using RStudio Version 1.4.1717. Statistical analyses were performed solely to describe and compare δ¹⁵N values across the different nitrogen-source categories reported in the reviewed studies. For each article, δ¹⁵N values were extracted at the level of the pressure categories identified by the authors (e.g., sewage, groundwater, wastewater, inorganic fertilizer). Because most studies reported δ¹⁵N values already averaged at the site or pressure level, the unit of analysis used in this review corresponds to one δ¹⁵N mean value per study and per nitrogen-source category. Individual sample-level data (n = 10,628 across all studies) were not available and were therefore not used to avoid pseudo-replication and inflated degrees of freedom. As a result, each data point included in the statistical analyses represents a single pressure-specific mean δ¹⁵N value, and the sample size (n) for each nitrogen source corresponds to the number of such pressure-level values.

Data were inspected for normality using the Shapiro-Wilk test (stats package) and variances were assessed with Bartlett's test (stats package). In the event of heterogeneous variances, non-parametric analyses were performed using the Kruskal-Wallis test (stats package). When significant differences were detected, a posteriori pairwise comparisons were conducted using Dunn's test (FSA package), with *p*-values adjusted using the rcompanion package. Data visualization was performed with ggplot2.

3. Results

3.1. Geographic and temporal distribution of studies

In total, the dataset encompasses 1395 sampling sites distributed across 28 tropical countries. The Pacific region is the area with the most biomonitoring studies representing 47% of the studies referenced, followed by the Gulf and Caribbean region (44%) and the Indian ocean area (8%) (Fig. 3). Few studies have been carried out in the tropical zones of South America and Asia, and none on the west side of Africa. These gaps highlight priority regions for future isotopic monitoring efforts. The macrophyte target species used in tropical environments are mainly macroalgae (66% of studies consulted). The use of seagrass is widespread and present in every ocean in our tropical context. However, the South American continent remains empty of any attempt at monitoring in seagrass beds.

Interest in monitoring coastal pollution in tropical areas has been growing over the years, with a more pronounced focus on macroalgae (Fig. 4). The development of isotopic analyses on marine organisms (seaweeds and others) was initiated in the 1950s. Thus, in 1955, Hoering study highlighted the natural isotopic variability of nitrogen and its links with biological, geological and chemical processes (Hoering, 1955). However, in our context, the first publication related to the monitoring of coastal pollution using an isotopic approach appeared in 1997 in Florida (Lapointe, 1997).

3.2. Species used as bioindicators of nutrient pollution across tropics

In total, 94 species of macrophytes (83 species of algae and 11 species of seagrass) were used in this review. The brown algae of the genus *Dictyota* is the preferred Ochrophyta, used 13 times across all studies, followed by the Rhodophyta genus *Laurencia* (9 times) and the Chlorophyta genus *Caulerpa* (8 times). Despite *Dictyota* being the preferred genus, Rhodophyta remain the most frequently cited phylum in the literature, with 48 occurrences (Fig. 5) Among the seagrass, the genus *Thalassia* is the most common, appearing 10 times (Fig. 5). Three articles did not specify the names of the algae on which they performed isotopic signatures. In sum, the number of samples examined amounts to 10,628 across the 28 articles which specified their number of samples per campaign.

3.3. Pollution sources and isotopic thresholds

The origins of pollution identified and counted across the studied publications in ascending order are industrial (3%), runoff (5%), particulate organic matter (POM 8%), aquaculture (9%), groundwater (12%), inorganic fertilizer (15%), sewage (21%) and wastewater (27%). The effluents grouped in the “sewage” category consider treatment plants, leaking cesspools, septic tanks and any water in contact with human excreta. The “wastewater” category includes effluent from injection wells, domestic discharges, and any water that has been

adversely affected in quality by anthropogenic influence. The “industrial” category covers contamination from factories, power plants, chemical plants, and other manufacturing facilities. Some of the pollution was not subject to statistical analysis because the sample size was less than three. This is the case for “runoff” and “industrial”.

Fig. 6 illustrates isotopic signature values found in macroalgae, assigned to the input sources identified by the authors. For the sake of homogeneity, all isotopic signatures are averaged and concern only shallow-water monitoring (< 20 m). A Kruskal-Wallis test indicated significant differences among nitrogen source categories ($H = 32.26$, $p < 0.001$). The nitrogen sources identified as groundwater, sewage and wastewater have no statistical difference ($p = 1.00$), but groundwater has a narrow range compared to sewage and wastewater. Groundwater ranges from 5.9 ‰ to 15.7 ‰, wastewater from 3.92 ‰ to 43.1 ‰ and sewage from 4.69 ‰ to the value of 50.1 ‰ found by Dailer et al. (2010) at the outlet of a wastewater treatment plant in Hawaii (Table 2). This similarity is not surprising, in fact, the groundwaters studied are often contaminated by wastewaters or sewages, due to the porous nature of certain soils (Kruczynski and McManus, 2002). The nitrogen elements of inorganic fertilizer stand out from sewage ($p = 0.0127$), wastewater ($p = 0.0034$) and groundwater ($p = 0.0368$), ranging between -4.5 ‰ to 3.03 ‰. The signatures assigned to particulate organic matter and aquaculture are similar ($p = 1.00$). They have a range of 3.1 ‰ to 7.9 ‰ and 3.6 ‰ to 9.6 ‰ respectively. Signatures associated with atmospheric nitrogen (-1.8 ‰ to 2.4 ‰) are found in pristine sites, though such sites have become increasingly rare due to anthropogenic influences. However, low $\delta^{15}\text{N}$ values are not exclusive to pristine environments, as they can also result from nitrogen fixation by diazotrophs or the input of inorganic fertilizers, both of which are characterized by isotopically light nitrogen. Unfortunately, the data collected do not satisfy the conditions for claiming statistical tests by genus or phylum. In summary, macroalgal $\delta^{15}\text{N}$ values associated with atmospheric N and inorganic fertilizer were significantly lower than those associated with groundwater, sewage and wastewater, while aquaculture and POM overlapped both groups.

Concerning seagrasses, the sample size was limited for most pollution sources, preventing any meaningful statistical comparison. Only wastewater and aquaculture categories contained enough observations to potentially allow a Kruskal-Wallis test, which is not statistically informative. For this reason, no inferential statistics were conducted for seagrass $\delta^{15}\text{N}$ values, which are instead reported descriptively. Wastewater ranges from 1.5 ‰ to 10.3 ‰. Other nitrogen sources like groundwater and inorganic fertilizers range from 2.31 ‰ to 8.5 ‰ and 0.91 ‰ to 2.31 ‰ respectively. Again, the data collected do not satisfy the conditions for claiming statistical tests by genus of seagrass. A comparison between the signatures of macroalgae and seagrasses is not relevant, as they have not been studied in the same sites or under the same exposures, and more importantly because they do not share the same metabolism.

Beyond these statistical differences, it is important to note that $\delta^{15}\text{N}$ ranges show substantial within-category variability and often overlap

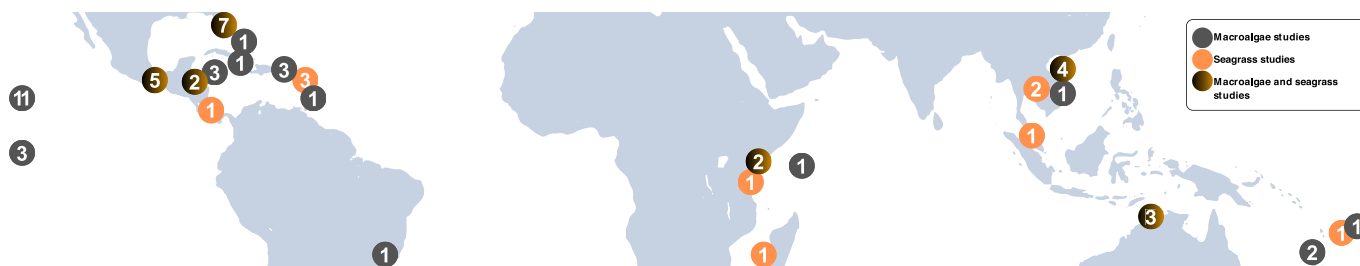


Fig. 3. Spatial distribution of study areas located within tropical regions. Areas using macroalgae are indicated by grey circles. Areas using phanerogams are indicated by orange circles. The two-color circles indicate the study areas considering macroalgae and phanerogams. Numbers indicate the number of studies recorded in each location.

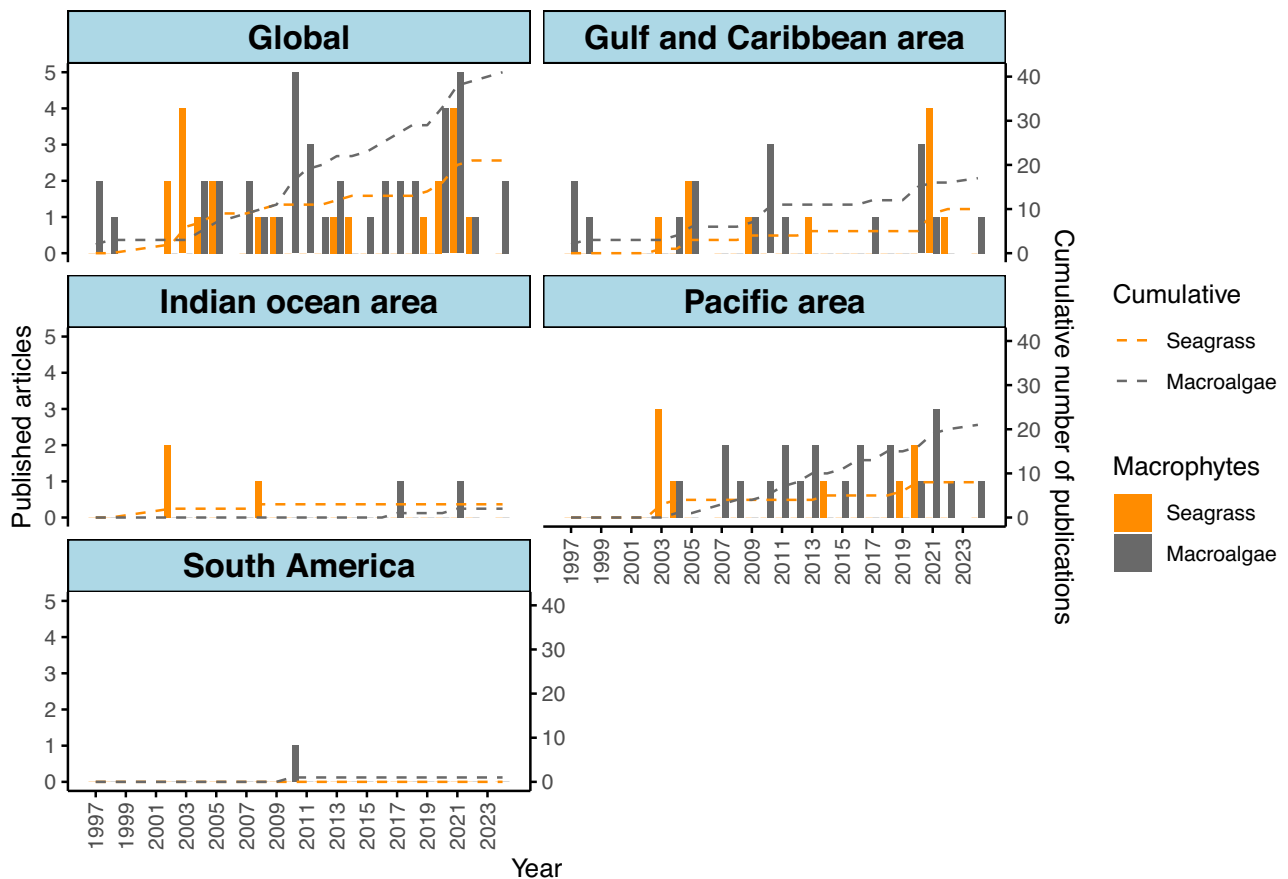


Fig. 4. Number of peer-reviewed published articles per year (1997–2024) and cumulative number of publications related to $\delta^{15}\text{N}$ monitoring using macrophytes, faceted by geographic region. Some studies may include multiple countries or regions.

between pollution sources.

4. Discussion

4.1. Geographic distribution and interpretation of nitrogen source signatures

The dataset shows a geographic bias, with a great number of studies originating from the Caribbean and Pacific regions and few from the Indian Ocean and South America regions. This geographic imbalance may partly reflect the early development of stable isotope ecology within North American research institutions, where many foundational methods, laboratories, and long-term monitoring programs were first established (e.g. Peterson and Fry, 1987; Fry, 2006). These historical research networks likely contributed to a greater number of studies in regions closely connected to these institutions, such as the Caribbean and Pacific region.

Wastewater effluents are the primary concern for the enrichment of oligotrophic waters in tropical systems (Figueroa-Nieves et al., 2014). Many monitoring studies have identified this source of pollution with macroalgae, as well as seagrass. The synthesis of results concerning macroalgae demonstrated that wastewater, sewage, and groundwater record the highest values of nitrogen, from $\sim 4\text{‰}$ to $>10\text{‰}$ (Fig. 6, Table 2). The high $\delta^{15}\text{N}$ values in wastewater nitrogen stem from nitrate denitrification and ammonia nitrification. During these processes, bacteria favor the use of the lighter isotope, ^{14}N , as it is easier to metabolize (Heaton, 1986). The highest values recorded in this review come from treatment plants employing Biological Nitrogen Removal (BNR), where microbes are utilized for the uptake of nitrogen and phosphorus levels in the wastewater. This leads to abnormally high $\delta^{15}\text{N}$ values ($>20\text{‰}$). The

signatures associated with effluents from synthetic fertilizers and atmospheric deposits (natural) range from -4.5‰ to $\sim 3\text{‰}$. These inorganic fertilizers are often composed of natural nitrogen compounds which, when incorporated into the soil, release nitrogen in forms similar to those present in the atmosphere, resulting in comparable isotopic signatures (Kendall et al., 2008). As a result, fertilizer-derived nitrogen and atmospheric deposition cannot be easily distinguished using $\delta^{15}\text{N}$ alone, and their overlapping ranges reflect their shared isotopic origin. Nitrogen wastes from aquaculture and particulate organic matter (POM) range from $\sim 3\text{‰}$ to $\sim 9\text{‰}$. Aquaculture sites are heavily enriched with nitrogenous elements due to animal excrement and food remnants (Naylor et al., 2000). Nearly 80% of the feed is not retained in the biomass of target organisms (Jackson et al., 2003) and all effluents flow directly into adjacent water streams (Hargreaves, 1998; Preston et al., 2000). In parallel, it has been demonstrated that rivers contribute to 50% of POM inputs on coral reefs (Briand et al., 2015).

For seagrass, despite the lack of data for more precise results, pollution thresholds seem to follow those measured in macroalgae, but with lower sensitivity (Table 2). As mentioned before, macroalgae and seagrasses both assimilate various forms of nitrogen, with a general preference for ammonium due to its lower energy cost compared to nitrate (Touchette and Burkholder, 2000). While macroalgae primarily absorb dissolved inorganic nitrogen (DIN) from the water column, seagrasses can also take up nitrogen through their leaves and roots from sediment porewater (Touchette and Burkholder, 2000; Engeland et al., 2011). Additionally, both macroalgae and seagrasses are capable of directly assimilating dissolved organic nitrogen (DON), allowing them to bypass certain nitrogen cycling processes and access alternative nitrogen sources in coastal environments (Vonk et al., 2008). The generally higher responsiveness of macroalgae to nitrogen enrichment

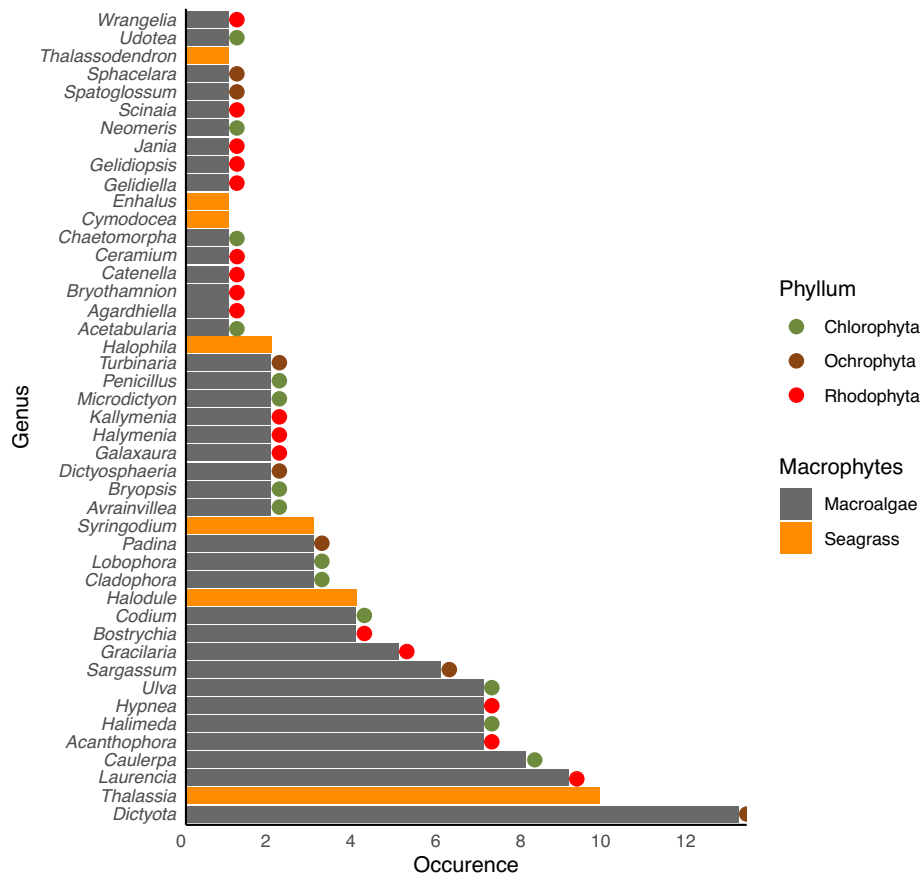


Fig. 5. The occurrence of various macrophyte genus in scientific literature focusing on the use of nitrogen stable isotopes (¹⁵N) to trace nitrogen pollution. Data are primarily drawn from research conducted in tropical coastal ecosystems, where macroalgae or seagrass serve as bioindicators of anthropogenic nutrient inputs.

Boxplot avec lettres des différences significatives

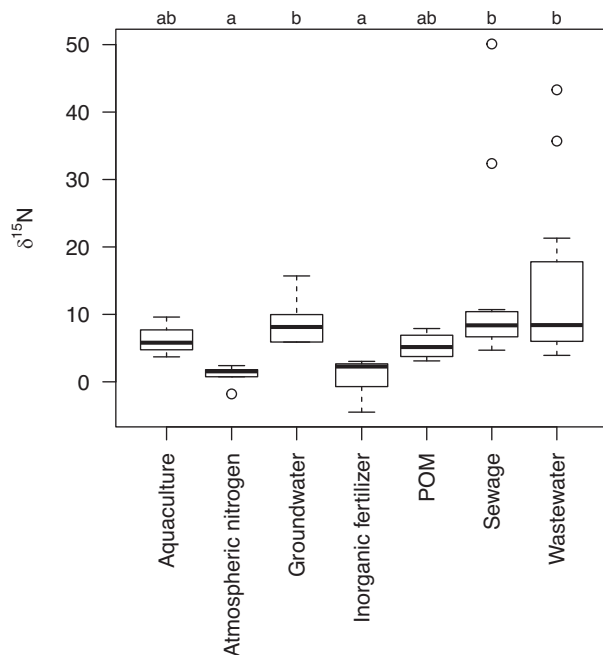


Fig. 6. Boxplot of ^{δ15}N values measured in macroalgae, by type of pollution in tropical areas. This graph compares ^{δ15}N values across 72 results for different nitrogen sources: aquaculture, atmospheric nitrogen (pristine sites), groundwater, inorganic fertilizer, POM, sewage, and wastewater. Letters above the boxes indicate significant differences between groups identified by Dunn's test.

compared to seagrasses can be attributed to several factors. Macroalgae often exhibit faster growth rates and higher tissue turnover, which leads to more immediate incorporation of available nitrogen and more rapid reflection of changes in ^{δ15}N signatures (Buapet et al., 2008; Wang et al., 2014). In contrast, the slower growth and turnover of seagrass tissues, along with their access to multiple nitrogen sources (water column and sediment porewater), could moderate isotopic shifts in tissues (Touchette and Burkholder, 2000). Experimental work on nitrogen uptake dynamics also indicates that although seagrasses can respond to short-term nitrogen pulses, their surge uptake rates are often lower than those of fast-growing macroalgae under enriched conditions (Alexandre et al., 2017). This reduced uptake efficiency limits both the magnitude and the immediacy of isotopic shifts in seagrass tissues, contributing to their lower sensitivity as bioindicators.

To summarize these differences, a comparative table outlining strengths and limitations of macroalgae and seagrasses as bioindicators, considering factors such as growth rate, isotopic fractionation, tissue turnover, nitrogen source accessibility, and sampling ease, provide a clear overview (Table 3).

4.2. Macrophytes selection

The most widely used genus of macroalgae in scientific publications is *Dictyota*, a brown algae with a wide distribution on reefs and a rapid growth rate, which can be significantly accelerated by a spike in nitrogen concentration. It is an opportunistic species in nutrient-rich environments and is available year-round, making it easy to collect. Among all phyla, Rhodophyta are known to have a better assimilation of nitrogen compared to Ochrophyta and Chlorophyta (D'Elia and DeBoer, 1978; Jones et al., 1996). The efficiency of nitrogen assimilation in macroalgae can also be influenced by their morphological

Table 2

Summary of $\delta^{15}\text{N}$ average isotopic values by pollution sources monitored in macroalgae and seagrass from studies referenced in the present review.

$\delta^{15}\text{N}$ Source		Min (‰)	Max (‰)	n	Mean SD (‰)	Median [95% CI] (‰)	Post-hoc grouping
Aquaculture	Macroalgae	3.7 (Costanzo et al., 2004)	9.6 (Cui et al., 2021)	3	6.37 (± 2.99)	5.8 [3.7–9.6]	ab
	Seagrass	7.8 (Herbeck et al., 2014)	12.4 (Thomsen et al., 2020)	3	9.23 (± 2.75)	7.8	
Atmospheric nitrogen	Macroalgae	-1.8 (Lapointe et al., 2017)	2.4 (Garrison et al., 2007)	5	0.94 (± 1.64)	1.6 [-1.8–2.4]	a
	Seagrass	1.62 (Yamamuro et al., 2003)	1.78 (Yamamuro et al., 2003)	2	1.7 (± 0.11)	1.7	
Groundwater	Macroalgae	5.9 (Abaya et al., 2018b)	15.7 (Amato et al., 2020)	6	8.95 (± 3.72)	8.1 [5.9–12.8]	b
	Seagrass	2.31 (Fourqurean et al., 2005)	8.5 (Kamermand et al., 2002)	2	5.4 (± 4.38)	5.4	
Industrial	Macroalgae	9.15 (Alquezar et al., 2013)	9.15 (Alquezar et al., 2013)	1	9.15	9.15	
	Seagrass	7 (Ashikin et al., 2020)	7 (Ashikin et al., 2020)	1	7	7	
Inorganic fertilizer	Macroalgae	-4.5 (Lapointe et al., 2021)	3.03 (Lapointe et al., 2004)	8	0.87 (± 2.75)	2.3 [-2.1–2.8]	a
	Seagrass	0.91 (Gaubert-Boussarie et al., 2021)	2.31 (Fourqurean et al., 2005)	2	1.61 (± 0.99)	1.6	
POM	Macroalgae	3.1 (Briand et al., 2015)	7.9 (Vaughan et al., 2021)	4	5.33 (± 2.06)	5.2 [3.1–7.9]	ab
	Seagrass	2.15 (Anderson and Fourqurean, 2003)	2.15 (Anderson and Fourqurean, 2003)	1	2.15	2.15	
Runoff	Macroalgae	9.75 (Cox et al., 2013)	11.2 (Lapointe et al., 2011)	2	10.5 (± 1.06)	10.5	
	Seagrass	NA	NA	0	NA	NA	
Sewage	Macroalgae	4.69 (Lapointe et al., 2004)	50.1 (Dailer et al., 2010)	12	13.5 (± 13.6)	8.4 [6.68–10.4]	b
	Seagrass	9.06 (Carruthers et al., 2005)	9.06 (Carruthers et al., 2005)	2	9.24 (± 0.25)	9.24	
Wastewater	Macroalgae	3.92 (González-De Zayas et al., 2020)	43.3 (Dailer et al., 2010)	14	13.8 (± 12.2)	8.4 [6–17.8]	b
	Seagrass	1.5 (Murphy et al., 2022)	10.3 (Ashikin et al., 2020)	4	6.45 (± 4.49)	7 [1.5–10.3]	

For each pollution source, the table reports minimum and maximum values, the number of observations (n), the mean \pm SD, and the median with its 95% confidence interval estimated by non-parametric bootstrap resampling. Different letters (a, b) denote groups that significantly differ according to Dunn's post-hoc test following a Kruskal-Wallis analysis ($p < 0.05$).

Table 3

Comparative characteristics of macroalgae and seagrasses as bioindicators of nitrogen enrichment.

Criterion	Macroalgae	Seagrasses
Growth rate	Fast; high tissue turnover	Slow to moderate; low tissue turnover
Speed of $\delta^{15}\text{N}$ response	Rapid reflection of changes in N sources	Slower response; integrates over longer time scales
Primary N uptake pathway	DIN + DON from water column	DIN + DON from water column and sediment porewater
Isotopic signal variability	High sensitivity to enrichment	High sensitivity to low enrichment and moderated by mixed N sources
Spatial resolution	High (fine-scale variability detectable)	Moderate (integrates over sediment + water signals)
Suitability for caging experiments	High; easy to transplant and deploy	Limited; caging not feasible due to roots systems
Sampling ease	Easy to collect; common along reefs and shores	Easy to collect; patchy distribution in some regions
Best use case	Detecting short-term or localized N enrichment	Detecting longer-term or integrated N loading

characteristics. For instance, the elongated form of certain algae, such as *Ulva intestinalis*, enables them to assimilate N more rapidly (Dailer et al., 2010; Vergara-Carranza et al., 2024). Some studies grouped species for isotopic analyses because the distribution of algae was not uniform (e.g. Barile and Lapointe, 2005; Abaya et al., 2018b), without specifying whether there were any potential differences. Other studies separated species by phylum (e.g. Briand et al., 2015) or selected a single species with a wide distribution (e.g. Lin and Fong, 2008; Teichberg et al., 2010; Adam et al., 2020). The choice of algae may also be based on available literature or previous studies established at the sampling location (e.g. Vaughan et al., 2021).

On the seagrass side, the genus *Thalassia* is the most represented across studies, particularly *T. testudinum* in the Gulf and Caribbean region (e.g. Anderson and Fourqurean, 2003; Carruthers et al., 2005; Sánchez et al., 2013). This species, widely known as turtle grass, is a sensitive indicator of increasing nutrient availability, exhibiting measurable changes in productivity, leaf area and shoot density under elevated nutrient loads (Green and Webber, 2003). In the Indo-Pacific

region, however, other taxa dominate monitoring efforts, including *Thalassia hemprichii*, *Enhalus acoroides*, and species of the genera *Halodule* (e.g. Yamamuro et al., 2003; Jiang et al., 2019; Ashikin et al., 2020; Thomsen et al., 2020). Among the studies, 43% did not further elaborate on their choice of seagrass species.

For optimal monitoring, the authors recommend targeting a local species with a wide distribution range in the case of a one-time harvest. The algae should preferably be non-calcareous, to ensure isotopic fractionation close to the original nitrogen source, and rapid N accumulation in the case of transplantation monitoring (Delgado and Lapointe, 1994). Also, to avoid any bias in the analyses, we recommend choosing an easily identifiable species. In the case of a comparative study between sites, we recommend using the same species to avoid natural interspecific variations in the isotopic signatures. If a similar species is not available for collection at all sites, preliminary studies should be conducted to determine if the various species collected exhibit close isotopic signatures. Otherwise, it can be useful to consider a caging experiment with transplanted specimens.

4.3. Caging methods

For macrophyte sampling, the collection of the whole plant is generally favored to proceed with direct isotopic analysis. Otherwise, a caging protocol can be proposed, that is to move the plant from a collection site to a monitoring site. In this case, the plant is first acclimatized for seven days in low-nutrient water, or taken from an already nutrient-poor environment, to ensure low internal ^{15}N levels ($\delta^{15}\text{N} < 3$ ‰, Alquezar et al., 2013). Only two studies using the caging method did not acclimatize their samples (e.g. Teichberg et al., 2010; Lapointe et al., 2017). No pollution monitoring carried out with seagrass has used the caging monitoring technique, likely because their root systems make them difficult to maintain in cages. However, among studies using macroalgae for mineral pollution monitoring, 37% employed the caging technique (e.g. Amato et al., 2016, 2020; Yoshioka et al., 2016; Abaya et al., 2018a). It is an essential tool when the site to be monitored is devoid of algae, and a solution for large-scale monitoring lacking homogeneity in the presence of target species (e.g. Lapointe et al., 2017). The cages are placed between 0.5 and 10 m deep, for an average of 10 days (3 days minimum, Lin and Fong, 2008; and 21 days maximum, Todd et al., 2009). Monitoring by caging method avoids potential

phenotypic or genotypic adaptation of native plants in polluted areas. Indeed, different levels of tolerance to pollution could influence data interpretation (Ritter et al., 2010).

For the preparation for caging, the authors recommend that cages should preferably be circular in shape, to reduce the impact of friction that can be encountered in a cubic cage. The quantity of algae introduced should not be too great, to allow water circulation and prevent the algae from suffocating and degrading. We also recommend that macroalgae be acclimatized before being placed on site, to optimize N uptake. The depth at which the cage is deployed should be adapted according to the coastal pollution to be monitored, the distance from the coast needed and light availability (i.e. turbidity).

4.4. Preparation for isotope analysis

After harvesting macrophytes in the field, 87% of studies washed their samples using either deionized water, distilled water, acid, fresh water, on-site seawater, or simply scraped off all of the epiphytes. Most studies dried the samples at 60 °C for 24 to 48 h, ground them to powder and sent them to a specialized laboratory for isotopic analysis. Some preferred freeze-drying at –80 °C for technical reasons. Only one article specified the storage time for samples before analysis, which was 24 h after sampling (Titlyanov et al., 2011).

After harvesting, we recommend thorough washing with distilled water to eliminate salt residue from samples. We recommend freeze-drying or drying samples in an oven at 60 °C to constant weight. If samples cannot be dried shortly after sampling, we recommend freezing the samples to avoid biological degradation and minimize isotope ratio variations (Hobson et al., 1997) as variations may occur if samples are placed under uncontrolled conditions (Bosley and Wainright, 1999; Kaehler and Pakhomov, 2001). Freeze-dried or dried samples are generally considered stable for long-term storage and are less prone to isotopic drift compared to other preservation methods, provided they are stored under appropriate conditions. For nitrogen isotopic analysis, studies suggest that freeze-drying effectively preserves isotopic integrity by halting microbial activity and preventing decomposition that might alter the isotopic composition (Bosley and Wainright, 1999). The mass submitted for isotope analysis should be in accordance with the sensitivity of the isotopic mass spectrometer used, while the number of replicates shall be a minimum of three.

4.5. Limits to isotope analysis interpretation

Although macrophyte-based isotope monitoring is conceptually simple, its implementation is not necessarily accessible. Many tropical countries lack local isotope ratio mass spectrometry (IRMS) laboratories, making analyses costly and dependent on partnerships with research institutions. These constraints partly explain why relatively few studies exist outside the Pacific, North and Central America, despite the method's demonstrated effectiveness.

Stable isotope values can vary considerably depending on environmental factors (such as temperature, salinity, and substrate composition), which complicates the attribution of specific signatures to pollution sources. Variations due to abiotic factors can alter the bioavailability of nitrogen in the environment. It is imperative to assess terrestrial pressures near sampling sites to provide discernment in result interpretation. It is also important to consider water movement and currents as they act as vectors for dispersion and distribution, as demonstrated in previous studies (Derse et al., 2007; Todd et al., 2009; Adam et al., 2020). Differences in the physiology of macroalgae and seagrass species can affect isotopic signatures, making comparison between sites or studies challenging. Some studies making cross-country comparisons have chosen a single species to overcome this bias (Lapointe, 1997; Kamermans et al., 2002; Teichberg et al., 2010). Another factor of variability is the season. The results of previous works show more variability in isotopic signatures in the wet season rather

than the dry season (Lapointe et al., 2010; Alquezar et al., 2013; Dell et al., 2020; Adam et al., 2020). Intense runoff periods can introduce massive amounts of nutrients, temporarily altering isotopic signatures. Nevertheless, Lapointe et al. (2011) did not find seasonal variation in their monitoring. In another study, Lapointe et al. (2017), the isotopic value of groundwater was lower during the wet season, explained by a higher water table and therefore reducing the amount of nitrification. Distance from the source and sampling depth are also considered as factors of variability. In all scenarios, the signals gradually decrease as one moves away from the source or deeper. However, the factor “depth” should be taken with caution because of the upwelling phenomenon that can influence results (Strait et al., 2022). Upwelling or internal waves from deeper waters (depths of 200 to over 350 m) can introduce nitrogen inputs with altered nitrate $\delta^{15}\text{N}$ values ranging between 6 and 7 ‰ (Casciotti et al., 2008; Knapp et al., 2011). Finally, the part of the macrophyte sampled can cause a variation in the interpretation of isotopic signatures. The young apical parts generally provide more recent information than the basal parts which have recorded months of information (Bryan et al., 1985; García-Seoane et al., 2018).

The dataset compiled in this review encompasses a wide range of tropical coastal environments, including river mouths and estuaries (e.g. Yamamuro et al., 2004; Lapointe et al., 2017), semi-enclosed bays and coves (e.g. Derse et al., 2007; Alquezar et al., 2013; Cui et al., 2021), and lagoonal systems (e.g. Gaubert-boussarie et al., 2021; Murphy et al., 2022). These environments differ strongly in hydrodynamics, water residence times, connectivity to the open ocean, and exposure to terrestrial inputs. For example, estuarine and river-influenced systems typically retain and transform nitrogen over longer residence times, while open or semi-open lagoons and carbonate-type systems are characterized by rapid flushing and limited nutrient accumulation. Such heterogeneity inevitably affects the baseline $\delta^{15}\text{N}$ signatures of macrophytes and the way isotopic enrichment propagates through the system.

In addition to shaping baseline conditions, these physical and biogeochemical processes also generate substantial within-source variability, meaning that a single pollution category (e.g., wastewater, sewage, groundwater) can encompass a broad isotopic range depending on mixing, transport distance, residence time and nitrogen transformations along the pathway.

Consequently, the thresholds proposed in this review should not be considered universal, but rather context-dependent benchmarks that require validation under local environmental conditions. $\delta^{15}\text{N}$ alone rarely provides a complete picture of eutrophication dynamics. Its interpretive power increases substantially when paired with complementary metrics such as macrophyte biomass or productivity, percent cover, C:N ratios, or in situ measurements of dissolved nutrient concentrations. Integrated biological and physico-chemical monitoring strengthens source attribution and reduces ambiguity linked to environmental variability.

5. Conclusion

The use of macrophytes as a tool for monitoring coastal pollution is a simple and reliable method to implement, suitable for both short-term and long-term monitoring. The caging method overcomes sampling limitations by offering the freedom to place macrophytes in specific target areas. This review proposes indicative reference ranges of nitrogen isotopic signatures to help identify pollution sources, which can provide useful guidance for addressing the challenges of eutrophication in tropical ecosystems. These ecosystems are highly sensitive to nutrient inputs and isotopic signatures provide a valuable tool to assess the impacts of nitrogen fluxes, especially where traditional monitoring methods (such as punctual water samplings) fail due to low nitrogen concentrations in tropical environments.

Advances in stable isotope measurement techniques may render some earlier studies less relevant or accurate, requiring continuous updating of knowledge. Isotopic analyses can also be performed on other

elements, such as carbon (C) and sulfur (S), providing valuable insights into pollutants and biogeochemical processes in coastal ecosystems. Future research should also expand beyond classical isotopes to include multi-element and compound-specific approaches, which offer finer resolution in tracing nutrient pathways and differentiating overlapping pollution sources. In particular, coupling $\delta^{15}\text{N}$ with other chemical tracers or emerging contaminants, such as pharmaceuticals, pesticides, or microplastics, could help reveal causal relationships between nitrogen enrichment and broader ecological degradation. Investigating how $\delta^{15}\text{N}$ signatures correlate with the presence, concentration, or bio-accumulation patterns of these contaminants would provide a more integrated understanding of coastal pollution dynamics. Additionally, stable isotopes of other elements (e.g., $\delta^{13}\text{C}$, $\delta^{34}\text{S}$, $\delta^2\text{H}$) remain underutilized in tropical systems despite their potential to shed light on organic matter sources, groundwater contributions, wastewater discharge, and acidification processes. Integrating these complementary isotopic tools into macrophyte-based monitoring could greatly enhance the diagnostic power of future studies. Furthermore, monitoring in macrophytes could be extended to include other types of pollutants, such as heavy metals, organochlorines, and microplastics, offering a more comprehensive assessment of contamination.

The recommendations provided in this review aim to guide future research and implementation of pollution monitoring, in order to better protect the biodiversity of tropical coastal ecosystems, which are particularly vulnerable to issues like coral mortality and coral diseases. Developing long-term, standardized monitoring networks that combine isotopes with metadata on hydrology, land use, and contaminant loads would greatly improve the comparability of studies and support adaptive management strategies in data-poor tropical regions. Strengthening collaborations between research institutions and local management agencies will also be essential to building capacity for isotope-based monitoring where analytical infrastructure is limited. The continued development and adoption of new measurement techniques will enhance the effectiveness of such monitoring and ensure more precise and sustainable environmental management.

Establishing indicative $\delta^{15}\text{N}$ thresholds is not only useful for identifying nutrient sources but also for anticipating ecological consequences. Elevated nitrogen inputs can trigger shifts in algal cover, seagrass decline, and coral stress, potentially leading to phase shifts in tropical coastal ecosystems. By providing reference isotopic ranges, this review offers a tool to detect early signs of nutrient enrichment, supporting proactive monitoring and management before irreversible ecological impacts occur. In regions where analytical capacities are limited, the synthesis provided here can still support preliminary assessments by managers. When signs of ecological change, such as shifts in algal cover or local declines in seagrass condition, raise concerns about nutrient enrichment, targeted $\delta^{15}\text{N}$ measurements on resident or caged macrophytes offer a practical first step for clarifying whether nitrogen inputs are likely involved. Comparing these values with the indicative reference ranges proposed in this review can help identify the probable origin and spatial extent of enrichment, guiding the need for complementary monitoring or management actions.

CRediT authorship contribution statement

Hanna-May Malahel: Writing – original draft, Visualization, Software, Methodology, Formal analysis, Data curation, Conceptualization. **Charlotte R. Dromard:** Writing – review & editing, Validation, Supervision. **Isabelle Mussio:** Writing – review & editing, Validation, Supervision. **Pascal Claquin:** Writing – review & editing, Validation, Supervision.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence

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Appendix A. Supplementary data

Supplementary data to this article can be found online at <https://doi.org/10.1016/j.jseares.2026.102669>.

Data availability

The full dataset used in this review, including all studies, macrophyte species, pollution sources, mean $\delta^{15}\text{N}$ values and other information, is openly available on SEANOE under the following DOI: [10.17882/110691](https://doi.org/10.17882/110691).

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