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Anthropogenic nitrogen pollution threats and challenges to the health of South Asian coral reefs

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Nitrogen pollution is a widespread and growing problem in the coastal waters of South Asia yet the ecological impacts on the region's coral ecosystems are currently poorly known and understood. South Asia hosts just under 7% of global coral reef coverage but has experienced significant and widespread coral loss in recent decades. The extent to which this coral ecosystem decline at the regional scale can be attributed to the multiple threats posed by nitrogen pollution has been largely overlooked in the literature. Here, we assess the evidence for nitrogen pollution impacts on corals in the central Indian Ocean waters of India, Sri Lanka and the Maldives. We find that there is currently limited evidence with which to clearly demonstrate widespread impacts on coral reefs from nitrogen pollution, including from its interactions with other stressors such as seawater warming. However, this does not prove there are no significant impacts, but rather it reflects the paucity of appropriate observations and related understanding of the range of potential impacts of nitrogen pollution at individual, species and ecosystem levels. This situation presents significant research, management and conservation challenges given the wide acceptance that such pollution is problematic. Following from this, we recommend more systematic collection and sharing of robust observations, modelling and experimentation to provide the baseline on which to base prescient pollution control action.

KEYWORDS
corals, Indian Ocean, South Asia, nitrogen pollution, anthropogenic impacts

1 Introduction

Occupying less than 0.1% of global ocean area coral reefs host around a quarter of all marine species and serve a variety of critical environmental, economic and cultural functions, valued at US$2.7 trillion per year (Birkeland, 1997; Hoegh-Guldberg, 1999; Moberg and Folke, 1999; Wilkinson, 2000; Harrison and Booth, 2007; Burke et al., 2011; Spalding et al., 2017; Souter et al., 2021). The high biodiversity associated with coral reefs supports fisheries and human livelihoods, whilst the reef structure provides key habitats, enhanced coastal protection, and a focus for tourism, which alone has been valued at US$36 billion in foreign currency exchange (Spalding et al., 2017; UNEP, 2019). Corals, however, face multiple and diverse threats at different spatial and temporal scales ranging from i) physical destruction of the reef structure as a result of fishing practices, coastal development, reef extraction or coral mining, ii) increased risk of mortality from pollution, disease, or competition with macroalgae, and iii) ocean-scale acidification and temperature-induced bleaching (Wilkinson et al., 2016; Souter et al., 2021). The long-term fate of coral reefs is predicted to be widespread ecosystem loss and reduced ecoservice provision with a recent global assessment of coral reef health describing a 14% reduction in coral reef coverage in the decade to 2018 (Souter et al., 2021). This ecosystem decline is driven by the combination of the global stressors of increased ocean temperature and ocean acidity resulting from rising atmospheric CO_2 levels, coupled in impact with more localized anthropogenic stressors (van Hooidonk et al., 2016; IPCC, 2019; UNEP, 2020; Eddy et al., 2021).

Long-term global temperature trends are expected to push corals above their optimal thermal tolerances over the coming century (IPCC, 2019), yet there is considerable uncertainty over the timing and severity of impacts at more regional scales. This uncertainty includes factors around potential rates of adaptation and acclimation, and related to environmental heterogeneity, particularly in coastal environments (Hughes et al., 2003; Smale et al., 2019; Guan et al., 2020; McClanahan et al., 2020; Oliver et al., 2020; UNEP, 2020; Ziegler et al., 2021). The cumulative effects and timings of anthropogenic stressors on coral mortality rates remain difficult to generalize because such impacts are frequently region specific, though it is acknowledged that those coral ecosystems most heavily impacted by anthropogenic stressors are those typically found along urbanized coastlines (Halpern et al., 2008; Rosenberg et al., 2022). Geographic remoteness from human impacts, however, does not necessarily provide additional protection from the impacts of global stressors (Strona et al., 2021; Baumann et al., 2022).

In South Asia, which contains ~7% of global coral coverage (Spalding et al., 2001; Wilhelmsson, 2002; Rajasuriya et al., 2004; GEER, 2008; Tamelander and Rajasuriya, 2008; Bahuguna et al., 2013), local stressors such as fishing and anthropogenic pollution are recognized as serious and immediate threats to coral ecosystems and have been argued to contribute to, or accelerate, regional coral loss (Rajasuriya et al., 2000; Rajasuriya et al., 2004; Tamelander and Rajasuriya, 2008). In this context pollution is frequently a catch-all term representing multiple pollution types that can include chemical, heavy metal or biological pollutants with diverse transport pathways including atmospheric deposition, industrial effluents, maritime activities, sewage discharge, terrestrial runoff and groundwater seepage. Nutrient pollution, primarily in the form of excessive nitrogen and phosphorus inputs to coastal waters, is widespread across South Asia and significant at the regional scale (BOBLME, 2015; Raghuram et al., 2021). Yet, despite the socioeconomic and cultural significance of corals throughout the region (Price and Firaq, 1996; Townsley, 2004; MEE, 2017a), the ecological impacts of anthropogenic nutrient inputs on the region's coral reef ecosystems remain widely alluded to but poorly studied (Selman et al., 2008; BOBLME, 2015; SACEP, 2019).

For the South Asian region in particular, quantitative information on coastal nutrient concentrations, how the nutrient seascape may be changing with time and how such changes impact coral ecosystems remains patchy. The aim of this study is to assess and review the impact of nutrient (nitrogen) pollution on coral reefs found in the central Indian Ocean waters of India, Sri Lanka and the Maldives with recommendations for future management and research directions. The paper consists of a short summary of reported nutrient impacts on corals (section 2), national assessments on water quality indicators, reported impacts of nutrient pollution on coral reefs and current legislative safeguards (section 3), a regional assessment of nutrient inputs from the identified dominant sources and recommendations for future research (section 4), and concludes with recommendations for improved regional monitoring (section 5).

2 Corals and their nutrient environment

2.1 Corals and anthropogenic nutrients

Corals grow best in shallow, optically clear, warm (>18-20°C), stratified, and nutrient poor waters of the tropics (Lewis, 1981; Lerman, 1986; Kleypas et al., 1999; Wilkinson et al., 2016), and are generally absent from regions receiving significant freshwater input or where nutrient concentrations, sediment loads and algal densities are high (Spalding et al., 2001; Spalding and Brown, 2015). The mean long-term environmental conditions experienced by healthy coral reefs are governed by the prevailing large-scale ocean circulation, but seasonality and physical mixing mechanisms are important for inducing short-term environmental variability around optimal growth conditions, which may include mild seasonal upwelling and changes to local nutrient fields (Doty and Oguri, 1956). Coral reefs therefore are dynamic environments that naturally experience fluctuations in temperature, salinity, light, turbidity, nutrients, wave action, weather, disease, and competition over multiple timescale (Hoegh-Guldberg, 1999; Burke et al., 2011).

Anthropogenic nutrient inputs arise primarily from land-use change, population growth and coastal development which lead to increased discharges of sewage, industrial or municipal effluents and terrestrial runoff (Robin et al., 2013; Karthik et al., 2020) but may also include changes to groundwater nutrient concentrations and subsequent seepage to coastal waters (Santos et al., 2021). Changes in the intensity of agricultural practices, the extent of land use change and the extent and speed of coastal developments generally influence the diffuse supply of nutrients from runoff to coastal waters whilst population growth and urbanization influence point-source discharges. Anthropogenic nutrient inputs can lead to elevated dissolved inorganic nitrogen (DIN; e.g. nitrate, nitrite and ammonium) or dissolved inorganic phosphorous (DIP; phosphate) concentrations in the coastal zone, may ultimately distort ambient nutrient fields and impact water quality (NRC, 2000; Cloern, 2001), and have become a focus of study for understanding human impacts on coastal habitats and coral health (e.g. (Dubinsky and Stambler, 1996; Fabricius, 2005; D'Angelo and Wiedenmann, 2014; Serrano et al., 2018)). The impacts on specific habitats such as coral reefs can be both direct and indirect (Risk, 2014; Fernandes de Barros Marangoni et al., 2020). Nitrogen pollution resulting from sewage discharge, terrestrial run-off and wastewater discharge is widely recognized (Wear and Thurber, 2015; Brodie et al., 2019; Tuholske et al., 2021) and has been linked to multiple impacts within coral ecosystems including i) increased macroalgal-coral competition (Dubinsky and Stambler, 1996; Karcher et al., 2020), ii) permanent shifts in coral-macroalgal dominance (phase shifts) (Bell, 1992; Done, 1992), iii) increased prevalence of coral disease (Antonius, 1985; Bruno et al., 2003), iv) impacts on coral growth and calcification rates (Ferrier-Pagès et al., 2000; Koop et al., 2001; Renegar and Riegl, 2005), v) alteration of the balance between calcification and bioerosion (Highsmith, 1980; Hallock, 1988), vi) impacts on coral reproduction, recruitment and larval success (Humanes et al., 2016; Liu et al., 2020), vii) reduced thresholds to bleaching (Wiedenmann et al., 2012; Donovan et al., 2020), viii) changes to coral-zooxanthella symbiosis (Schlöder and D'Croz, 2004), ix) impacts on coral larval performance (Serrano et al., 2018; Liu et al., 2020), x) changes to community species diversity (Fabricius, 2005) and xi) infestations of the corallivorous crown-of-thorns starfish (*Acanthaster planci*) (Brodie et al., 2005); Table 1. Nitrogen pollution poses a serious and complex threat to corals but with impacts that can be indirect, highly dispersed, variable in intensity and hard to quantify (Dubinsky and Stambler, 1996; Duce et al., 2008; Wilkinson et al., 2016), the range of reported impacts means that the subject of nitrogen enrichment has at times proven controversial and despite increased investigation much remains unclear (Bell, 1992; Szmant, 2002; Fabricius, 2005; D'Angelo and Wiedenmann, 2014; Risk, 2014; Lesser, 2021).

In addition to the impacts reported on corals, anthropogenic nutrient inputs are also associated with increased phytoplankton productivity, algal biomass and abundance, fish kills and nuisance blooms, leading to increased water column turbidity, increased particulate setting rates, and in some instances decreased dissolved oxygen concentrations resulting from increased bacterial remineralization. These additional responses to nutrient inputs can all alter the environmental conditions that corals experience even if the corals themselves appear unaffected. Anthropogenic nutrient inputs have also been linked to other important ecosystem changes such as increased nitrogen fixation activity, which can impact

TABLE 1 Summary of nutrient enrichment impacts on corals or coral ecosystems and selected example studies.

Known impacts of nutrient over-enrichment	Reference
Increased macroalgal dominance/macroalgal overgrowth/phase shift to turf-algae	Bell, 1992; Done, 1992; McCook, 1999; Vermeij et al., 2010; Brown et al., 2017; Karcher et al., 2020
Increased crown-of-thorn infestation	Birkeland, 1982; Lucas, 1982; Brodie et al., 2005; Miller, 2011
Increased bioerosion (sponges etc	Highsmith, 1980; Hallock, 1988; Risk, 2014; DeCarlo et al., 2015
Disruption of coral-zooxanthellae relationship (including photosynthetic response of endosymbionts, nutrient starvation in symbionts	Muscatine et al., 1989; Stambler et al., 1991; Stambler et al., 1994 Schlöder and D'Croz, 2004; Wiedenmann et al., 2012; D'Angelo and Wiedenmann, 2014; Morris et al., 2019
Changes to coral growth and calcification	Ferrier-Pagès et al., 2000; Koop et al., 2001; Renegar and Riegl, 2005
Increased susceptibility and prevalence of coral diseases	Bruno et al., 2003; Montano et al., 2016
Loss of coral cover, reduction in community diversity	Fabricius, 2005
Lowering of the thermal threshold for bleaching/increased bleaching susceptibility	Wooldridge, 2009; Wiedenmann et al., 2012
Species specific changes to larval survival, larval respiration, recruitment success, possible reduction in dispersal distances	Harrison and Ward, 2001; Serrano et al., 2018; Liu et al., 2020
Increased impact of ocean acidification/lowering of resistance to ocean acidification	D'Angelo and Wiedenmann, 2014
Increased turbidity (from sediment related nutrient enrichment and/or phytoplankton growth	Fabricius, 2005
Change to water column nutrient cycling (N_2 fixation	Bell, 1992; El-Khaled et al., 2020
Red tides/harmful algal blooms	Anderson et al., 2002
Reduced oxygen concentrations	BOBLME, 2013; Raj et al., 2020

nitrogen cycling through local pelagic ecosystems with the potential for secondary impacts on corals (Bell, 1992; El-Khaled et al., 2020). While the underlying mechanism linking ambient nutrient pools to coral health is still being debated (D'Angelo and Wiedenmann, 2014), there is general agreement that excessive and persistent increases in inorganic nutrient concentrations have a long-term detrimental impact *via* the promotion of macroalgal communities or stoichiometric imbalances impacting physiological processes (Wilkinson, 2008; Zhao et al., 2021; Buckingham et al., 2022). There is also growing acceptance that the impact of nutrient over-enrichment can be highly variable between sites (and species), which may partially explain some of the reported inconsistencies between studies (Szmant, 2002; Shantz and Burkepile, 2014; Lesser, 2021). Additional complexities arise due to coral species-specific responses to nutrient over-enrichment leading to both intra-regional and inter-regional differences in pollution impacts (Burkepile et al., 2020).

2.2 Nitrogen cycling in corals

Until quite recently a common understanding was that nutrient cycling within corals was largely determined by endosymbiont population dynamics. Under typical environmental conditions the autotrophic endosymbiont zooxanthellae (photosynthetic dinoflagellate of genera *Symbiodinium*), can provide around 30% of daily coral nitrogen requirements (Bythell, 1988), with the remaining 70% provided heterotrophically *via* particulate matter capture and ingestion (Bak et al., 1998). Under conditions of increased ambient nutrient concentrations, however, rapid growth of the algal symbionts can lead to competition between coral host and algae with the retention and increased usage of photosynthates by the algae at the expense of coral host metabolism (Ezzat et al., 2015; Morris et al., 2019). Corals, however, represent a complex assemblage of organisms living in symbiosis. This assemblage, referred to as the coral microbiome or coral holobiont, consists of the invertebrate coral and the endosymbiont zooxanthellae, but also includes associated bacteria, archaea, fungi and viruses. As each organism has its own physiological traits, the overall health of the holobiont can be disrupted by changes in the local environment impacting one or more of the individual component species. The significance of the holobiont model for understanding coral nitrogen demand lies in the emerging, yet poorly understood, relationship between the community of microorganisms contributing to the holobiont and how nitrogen is cycled and regulated within corals (Glaze et al., 2022; Mohamed et al., 2022). Recent studies indicate that the holobiont microbial community may be key to regulating fluctuations in environmental nitrogen availability and in maintaining the critical relationship between the coral host and its endosymbionts (Tilstra et al., 2021). More precisely, the holobiont makeup may be important for stabilizing nitrogen availability and maintaining nitrogen limitation in the endosymbiont *via* the relative actions of denitrifying and nitrogen-fixing bacteria. Whilst nitrogen fixation has been reported as a critical process supporting primary production by zooxanthellae endosymbionts during periods of low nutrient availability (Rädecker et al., 2015), it has also been reported that regulation of nitrate availability by microbial denitrification rates can occur when ambient nitrate concentrations are high thus potentially limiting the zooxanthellae response to elevated nutrient levels (Tilstra et al., 2021). Yet despite nitrogen fixation by free-living and colonial pelagic diazotrophs (Bell, 1992; El-Khaled et al., 2020), by benthic organisms (Cardini et al., 2014) and by bacterial symbionts (Lesser et al., 2007) being common within reef environments, nitrogen fixation is currently considered to be only a minor to moderate source of fixed nitrogen in the overall coral nitrogen budget, with estimates generally indicating a supply of <10%, but perhaps reaching as high as 20% under certain specific conditions (Rädecker et al., 2015; Benavides et al., 2017; Glaze et al., 2022; Moynihan et al., 2022).

The presence of highly efficient yet complex nitrogen cycling pathways within the holobiont not only supports the contradictory existence of productive coral reef ecosystems within otherwise nutrient-poor oligotrophic tropical waters but also highlights the challenges posed by anthropogenic nutrient inputs. As many of the pathways between bacteria, zooxanthellae and coral remain unclear there is considerable uncertainty over how and why increased ambient nutrient concentrations impact corals (Lesser, 2021). Multiple processes ranging from the acquisition of dissolved inorganic nitrogen, ingestion of particulate nitrogen, retention and intracellular translocation of dissolved organic nitrogen, conversion between nitrogen forms and inputs *via* nitrogen fixation are central to the productivity of coral reefs. Yet whilst elevated ambient nitrogen concentrations have been shown to broadly amplify coral bleaching impacts thus linking the impacts of nutrient inputs to global temperature trends (Donovan et al., 2020), studies have also shown that the form of the nitrogen substrate added is significant with greater sensitivity reported to elevated concentrations of nitrate than to ammonium or urea (Burkepile et al., 2020; Donovan et al., 2020; Fernandes de Barros Marangoni et al., 2020). Thus, whilst the overall health of the holobiont, and corals more generally, appears sensitive to chemical imbalances in the ambient environment (e.g. (Wiedenmann et al., 2012; D'Angelo and Wiedenmann, 2014)), there is still much to be understood particularly at regional scales. Nevertheless, long-term monitoring of the coral microbiome has been proposed as a useful means of obtaining early warning of the potential impacts of anthropogenic stressors on coral reefs due to the way in which microbial communities can change rapidly in response to environmental disturbances (Zaneveld et al., 2016; Glasl et al., 2019; Ziegler et al., 2019). Presently, however, there is a general lack of data on the temporal evolution of the microbiome in response to environmental (nutrient) stress, and insufficient understanding of the variability in bacterial communities between coral species to assess the usefulness of this approach for gauging the impact of specific stressors on corals or for understanding the implications for nitrogen cycling within the microbiome more generally (Glaze et al., 2022).

3 Coral reefs in the South Asia region

3.1 India

3.1.1 Coral reef extent, condition and environmental setting

India's peninsula coastline and its offshore islands face three biogeochemically distinct oceanic environments. To the west is the productive Arabian Sea, to the south is the subtropical Indian Ocean, and to the east is the comparatively fresh Bay of Bengal. Coral reef distribution in Indian waters is primarily restricted to the southern and western coasts and includes i) the Gulf of Kachchh and parts of the west coast, ii) the Gulf of Mannar and Palk Bay along the southern coast, iii) the offshore Andaman and Nicobar Islands and iv) the offshore Lakshadweep Islands (Figure 1). These regions are located far from the major riverine inputs occurring in the northern Bay of Bengal, where strong salinity fluctuations and high sediment loadings largely prevent coral colonization. The mean annual nitrate concentrations for the surface waters of India's Exclusive Economic Zone (EEZ) are typically 0-2.5 µmol L^{-1} (Arabian Sea), 0-3 µmol L^{-1} (Bay of Bengal) and 0-3.5 µmol L^{-1} (Andaman Sea) (Nair, 2010; Prema et al., 2017). Monsoon-driven seasonality is recognized as the dominant mode of environmental variability across the region with impacts that include the reversal of coastal currents (Shetye, 1998), seasonal coastal upwelling (Retnamma et al., 2020), substantial freshwater and nutrient discharge to Indian coastal waters (Singh and Ramesh, 2011; Krishna et al., 2016; Rao et al., 2017), and increased terrestrial runoff in the Andaman and Nicobar Islands (Raghuraman et al., 2013). Seasonal upwelling, particularly along the south-west Indian coastline, can increase surface nitrate concentrations 4-fold during the southwest monsoon months (Jun-Sep) while phosphate and silicate concentrations are reported to vary by less than 2-fold (Retnamma et al., 2020).

Reef types include fringing, barrier, atoll and patch reefs (Table 2). Fringing reefs are found in the Andaman and Nicobar Islands, the Gulf of Kachchh, Palk Bay and the Gulf of Mannar. Barrier reefs are found in the Andaman and Nicobar Islands whilst coral atolls are found in the Lakshadweep archipelago (De et al., 2017). Patch reefs are observed in the intertidal regions of Ratnagiri, Malvan and Redi as well as on the continental shelf between Vengurla and Vijaydurg. Submerged reefs patch reefs are also

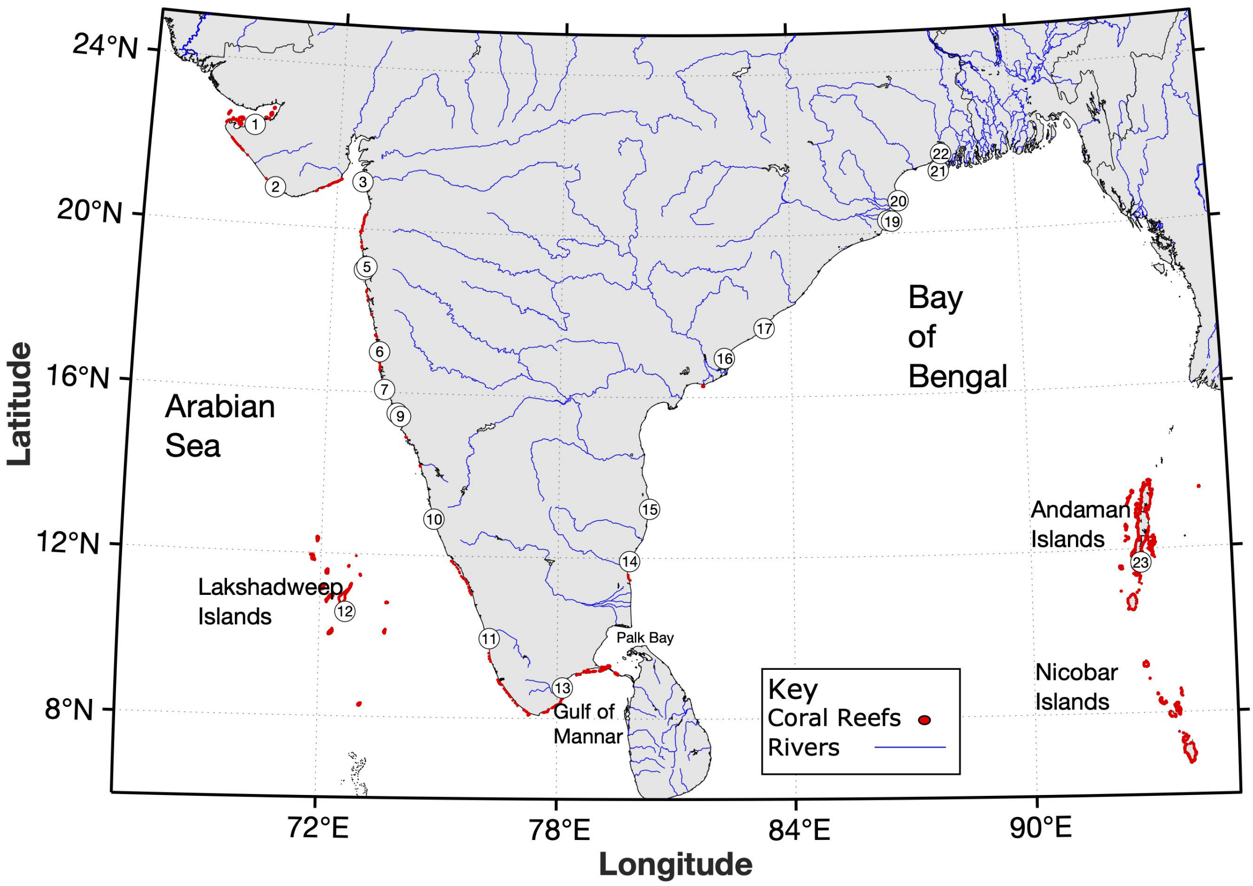

FIGURE 1
Distribution of corals in Indian waters. Coral distribution based on data compiled by UNEP-WCMC et al. (2018). Numeric indicators signify water quality monitoring locations shown in Figure 2.

TABLE 2 Literature reports of coral reef type, spatial coverage, diversity and status for the major coral reef locations in Indian waters.

Location (source)	Type	Dimension/coverage	Diversity	Status	Major threats
Gulf of Mannar					
Hoon, 1997	Fringing/ Barrier/Patch reef	140 km long, 25 km wide. Total area 94.3 km^2	96 species (36 genera)	Fair	Coral mining, turbidity, monsoons,
Rajasuriya et al., 2000		65 km^2	117		Destructive fishing, pollution, mining,
Rajasuriya et al., 2004		5790 km^2	104 species		Coral mining, destructive fishing, sedimentation
Tamelander and Rajasuriya, 2008			117 species		Sedimentation (monsoonal runoff), sewage, industrial pollutants, coastal development
Patterson et al., 2007			117 species, 40 genera		
Venkataraman, 2011	Barrier, patch, fringing		94 species, 32 genera Acropora, Montipora, Porites		
Palk Bay					
Hoon, 1997	Fringing reef	25-30 km long, 200 m wide	65 species	Degraded	Quarried, NE monsoon
Gulf of Kachchh					
Hoon, 1997	Fringing/patch reef	315 km^2	20 species	Degraded	Sedimentation, high temperatures, salinity changes,
Rajasuriya et al., 2000			37 species		Coastal development, pollution, mangrove removal
Rajasuriya et al., 2004	Fringing		42 hard coral/10 soft coral species		Coral mining, destructive fishing, sedimentation, pollution
Venkataraman, 2011	Fringing, platform		36 species		
Andaman and Nicobar Islands					
Hoon, 1997	Fringing reef	795.7 km^2	135 species (59 genera)	Degraded	Sedimentation (from mangrove removal), industrial pollution
Rajasuriya et al., 2000			203 species		Deforestation leading to sedimentation
Rajasuriya et al., 2004			203 species		
Venkataraman, 2011	Fringing				Poorly studied
Lakshadweep Islands					
Hoon, 1997	Atoll	136.5 km^2	105 species (37 genera)	Good to endangered	Storms, erosion, sewage
Rajasuriya et al., 2000			95 species		Crown of thorns, mining, dredging, fishing,
Venkataraman, 2011	Atoll	32 km^2	105 species, 37 genera		
West Coast of India					
Hoon, 1997	Patch	Unknown	29 species (17 genera)	Unknown	Sedimentation, low salinity

found along the Maharashtra coast, on Angria Bank off Sindhudurg in Maharashtra, around Grande Island in Goa, around Netrani Island on the Karnataka coast, on Gaveshani Bank off the Malpe coast, near Quilon and Vizhinjam along the Kerala coast, and Enayem along the Tamil Nadu coast. Mesophotic coral ecosystems are situated off the Puducherry coast (Jasmine et al., 2009; De et al., 2017; Kumar et al., 2019; Laxmilatha et al., 2019).

Coral species richness is variable between the main hosting regions and estimates vary significantly between studies due to occasional investigation. A latitudinal diversity gradient is recognized with species richness increasing from north to south. In the Gulf of Kachchh 36-68 species have been recorded (Venkataraman et al., 2003; MoEF, 2014; Marimuthu et al., 2018; Chandran et al., 2021), increasing to ~100 species in the Gulf of Mannar/Palk Bay (Bhatt et al., 2012b) and almost 150 species in the Lakshadweep Islands (Pillai and Jasmine, 1989; Government of India, 2012). Until recently around 200 species were thought to be present in Indian waters (Venkataraman et al., 2003; Venkataraman, 2011), though early diversity estimates remain uncertain (Bhatt et al., 2012b; Raghuraman et al., 2013). Species diversity estimates have now risen to 424 or even 588 species in more recent studies (Raghuraman et al., 2013; Majumdar et al., 2018; Jha et al., 2019) with the Andaman and Nicobar Islands alone potentially hosting a significant proportion of global coral diversity (Venkataraman et al., 2003; MoEF, 2009; Singh and Chaturvedi, 2017). Elsewhere however, a total of 560 species has been reported of which 478 are hermatypic (reef-building) representing around 60% of global species diversity (Raghuraman et al., 2013; MoEF, 2014).

Estimates of the total reef area in India also vary significantly between studies due to differences in the method of data compilation, method of analysis or resolution scale used (Garg, 2015). Based on remote sensing data published by different agencies during the periods 1992-1993, 2004-2007 and 2007-2008 estimates of total coral reef area fluctuated between 841 km^2, 2384 km^2 and 1420 km^2 respectively (Garg et al., 1998; SAC, 2011; Bahuguna et al., 2013). Spalding et al. (2001) estimated a total reef coverage of 5790 km^2 representing 2.04% of global coral area. More recent estimates indicate a reduced reef area of ~2375-2383 km^2 (SoE, 2009; Venkataraman, 2011; Bahuguna et al., 2013) – equivalent to 0.92% of global reef area (Souter et al., 2021). Coral coverage is presently estimated to be ~352 km^2 (Gulf of Kachchh), 76 km^2 (the Gulf of Mannar/Palk Bay), 1021 km^2 (the Andaman and Nicobar Islands) and 934 km^2 (the Lakshadweep Islands), (Bahuguna et al., 2013). Both the Lakshadweep Islands and the Andaman and Nicobar Islands contribute ~40% each to India's total reef area (Bahuguna et al., 2013; Saroj et al., 2016).

Despite uncertainties over the total reef area it is widely assumed that the spatial extent has declined in recent decades due to increased human pressures (Latha and Prasad, 2010; Ramadas and Rajeswari, 2011; Saroj et al., 2016; Rebekah and Inamdar, 2018) and the effects of significant coral bleaching events (Arora et al., 2019b). Widespread coral bleaching in 1998 reduced coral coverage by 20-40% in the Gulf of Kachchh and Gulf of Mannar, by 20-30% in the Lakshadweep Islands and by<10% in the Andaman and Nicobar Islands (Venkataraman, 2011). In 2016, approximately 70% of corals in Palk Bay and 46% of corals across the coastal Thoothukkudi Islands in the Gulf of Mannar experienced bleaching due to high sea surface temperatures of 34.0°C (Krishnan et al., 2018). Coral bleaching events in 1998, 2010, 2016 have had a distinct impact on India's corals, albeit with some notable variation attributed to species-specific resilience to increased temperatures (Krishnan et al., 2011; Jeevamani et al., 2013; Arora et al., 2019a; Arora et al., 2019b; Hussain and Ingole, 2020). Nevertheless, bleaching has led to widespread and significant mortality contributing to a general decline in coral reef coverage in India (Chandra et al., 2021).

The Andaman Islands suffered additional coral loss during the 2004 Indian Ocean tsunami which alone reduced coral coverage by 20-30% (Patterson et al., 2006; Majumdar et al., 2018). While the effects of bleaching can be severe, they may be temporary, and corals can recover over time. Repeated observations from Palk Bay, however, indicate a sustained decline in mean coral coverage which decreased by 37% between 2002 and 2018 (Marimuthu et al., 2020), and including an average decline of 2.5% per annum between 2007 and 2013 (Patterson et al., 2015). Anthropogenic factors (unspecified) have been implicated in the sustained deterioration of Palk Bay corals and as of 2018 live coral coverage in Palk Bay was only 17% (Marimuthu et al., 2020). Notably, whilst coral coverage has declined there has been a coincident increase in the prevalence of coral disease and of macroalgal coverage throughout Palk Bay which have been attributed to nutrient inputs (Patterson et al., 2015; Marimuthu et al., 2020). In the Gulf of Mannar coral coverage persists at around 30-37% (Magesh and Krishnakumar, 2019). Overall, the health of Indian coral reefs is generally reported as being poor (e.g. (Latha and Prasad, 2010; Ramadas and Rajeswari, 2011; Saroj et al., 2016; Rebekah and Inamdar, 2018) with pristine coral habitats described as now limited to a few areas of the Andaman and Nicobar Islands (Venkataraman, 2011; Bhatt et al., 2012b).

3.1.2 Water quality indicators

By the mid 1980's it was estimated that India's coastal population already discharged 4.1 x 10^9 m^3 yr^{-1} of domestic sewage to the coastal zone, with a further contribution of 50 x 10^6 m^3 yr^{-1} coming from inland communities *via* rivers (Qasim et al., 1988). By 2015, domestic sewage production in India had increased by over 350% reaching 22.5 x 10^9 m^3 yr^{-1} of which ~8.4 x 10^9 m^3 yr^{-1} (37%) was treated with the remaining ~15.1 x 10^9 m^3 yr^{-1} (63%) discharged as untreated sewage to seas and inland water bodies (Sahasranaman and Ganguly, 2018). In addition, it is estimated that between 1981 and 2014 the national consumption of agricultural fertilizer increased by ~400% (Ramesh et al., 2017), with around 5 x 10^6 tons of fertilizer residue being discharged to the coastal ocean each year (Latha and Prasad, 2010). There are few studies that specifically address the consequences of these temporal trends in pollution loadings on Indian corals (Sampath, 2003; Bhatt et al., 2012b; Jameson, 2012; Samuel et al., 2012; De et al., 2017; Balachandar et al., 2023). The knowledge base for assessing nutrient impacts on the general status and trends of corals, including biodiversity and ecosystem services, is considered patchy (Bhatt et al., 2012b) and quantitative physiological studies examining the impacts remain limited (Sampath, 2003). As a necessary first step towards understanding the impact of nutrient inputs on corals quantification of the extent and severity of nutrient enrichment of coastal waters remains either unclear or unavailable for many coastal areas (Ramadas and Rajeswari, 2011; Bhatt et al., 2012a), and recent efforts to assess nutrient pollution in Indian coastal waters have highlighted difficulties caused by the lack of openness of appropriate datasets (Ramesh et al., 2017).

Nevertheless, water quality monitoring within the coastal zone has a multi-decade history in India and began in 1986 with an exploratory program that collected seasonal observations of biological and chemical variables from 120 estuarine and coastal locations to identify sources and levels of pollutants in marine waters (Madeswaran et al., 2018). In 1990 this sampling effort transferred to the Coastal Ocean Monitoring and Prediction System program (COMAPS) of the Ministry of Earth Sciences (MoES), which continued regular sampling at 84 coastal locations but with annual, bi-annual or seasonal frequency. Following a subsequent program review water quality monitoring was relaunched under the Sea Water Quality Monitoring programme (SWQM) in 2012 but financial and logistical constraints reduced sampling to 24 locations where clear signatures of reduced dissolved oxygen concentrations and increased nutrient concentrations had previously been identified (Madeswaran et al., 2018).

The underlying data of the COMAPS/SWQM programmes are not publicly available. Madeswaran et al. (2018) however presented summary results of this sampling effort with the general conclusion being that there has been a widespread decline in coastal water quality throughout India, with increased discharge of untreated sewage being the major contributory factor. Increased phosphate and ammonium concentrations dominated observed changes to nutrient concentrations and were coincident with increased faecal coliform counts. Other studies, meanwhile, highlight nutrient enrichment resulting from agricultural runoff, which is said to be restricted to coastal waters near the Hooghly, Mahanadhi, Subernarekha, and Krishna Rivers and the Godavari estuary on the east coast (Sampath, 2003). BOBLME (2015) also reported nutrient enrichment within the Mandovi-Zuari and Cochin estuaries on the west coast. Notably, these (west coast) estuaries are not close to prominent coral hosting sites but the downstream transport pathways and residence time of any inputs are not clear. In contrast, results derived from a recent global assessment of coastal eutrophication potential indicated that ~6% of coastal waters along the west and south Indian coastline exhibited eutrophication potential, compared to only 2% of waters along the east Indian coastline (de Raus Maure et al., 2021).

In the final analysis of water quality indices for India presented by Madeswaran et al. (2018), water quality in the Gulf of Kachchh (Vadinar) was deemed to be poor, in the Gulf of Mannar (Tuticorin) and the Andaman and Nicobar Islands (Port Blair) it was deemed moderate, and in the offshore location of the Lakshadweep Islands (Kavaratti) it was considered good. However, for individual water quality parameters (e.g. faecal coliform counts, total N, total P), all locations were classed as having very poor water quality based on high faecal coliform counts highlighting the widespread prevalence of sewage in coastal waters (Figure 2). Notably, Tuticorin, Kavaratti and Port Blair were deemed to have good water quality based on observed nutrient enrichment levels suggesting that the impacts of nutrient input should not be generalized but should be examined for each region separately.

The detrimental effects of marine litter on reef ecosystems have also received increased attention recently (Mueller and Schupp, 2020), and are briefly mentioned here due to some similarity in impacts on coral systems. Recent studies indicate that the consequences of marine litter, notably plastic waste, have grown in both scale and impact (Mulochau et al., 2020; Karthik et al., 2022), and that litter contamination can now be linked to coral death and the proliferation of invasive species (Lamb et al., 2018; Valderrama Ballesteros et al., 2018; Patterson et al., 2020).

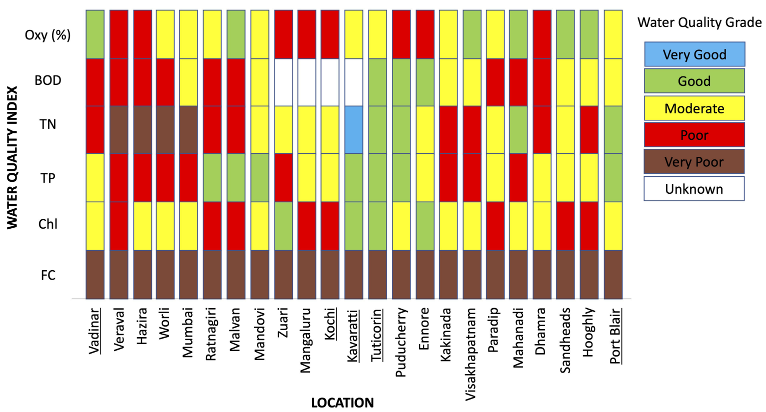

FIGURE 2
Water quality indicators at selected locations around India. Sampling sites close to major coral regions are underlined. All sampling sites are indicated on Figure 1 and numbered from left to right (Vadinar = 1,... Port Blair = 23). Indicators are dissolved oxygen saturation (Oxy %), Biological Oxygen Demand (BOD), Total Nitrogen (TN), Total Phosphates (TP), Chlorophyll-*a* (Chl) and Faecal Coliforms (FC). Figure modified from Figure 4.3.1 of Madeswaran et al. (2018).

3.1.2.1 Gulf of Kachchh

The Gulf of Kachchh experiences strong environmental variability including significant sediment and particulate loadings (Muley et al., 2000; Balasubramanian and Khan, 2001), and wide salinity ranges of 8.5 to 39 or higher (Nair et al., 1993; Mohandass et al., 2010). The sea surface temperature typically ranges from 24 to 30°C but can be higher in shallow waters. Recent water quality measurements are hard to find and can be inconsistent. Nair (2002) considered the water quality in the Gulf of Kachchh to be good, with limited potential for degradation around neighboring urban and industrial areas due to strong tidal flushing (Vethamony and Babu, 2010) and a residence time of approximately 1 day for the central Gulf (Patgaonkar et al., 2012). However, Patel et al. (2017) reported nitrate concentrations of 9.9 to 31.9 µmol L^{-1} and phosphate concentrations of 3.2 to 6.3 µmol L^{-1} specifically from reef waters in the Gulf, whilst Christian et al. (2019) reported even wider concentration ranges of 0.1 to 58.2 µmol L^{-1} for nitrate and 0.2 to 37.5 µmol L^{-1} for phosphate Saravanakumar et al. (2008), presented a two-year timeseries of nutrient concentrations that revealed nitrate and phosphate concentrations ranged from 0.23 to 7.26 µmol L^{-1} and 0.13 to 3.12 µmol L^{-1} respectively. It was further observed that nutrient concentrations were highest in August (the SE monsoon period), and lowest in March-May (summer). The maximum nutrient concentrations reported by Madeswaran et al. (2018) for the Vadinar monitoring location indicated DIN concentrations could reach ~18 µmol L^{-1} at inshore locations (0.5 km) and ~10 µmol DIN L^{-1} offshore (5 km), whilst DIP concentrations were less variable but elevated at 5.1 and 5.6 µmol L^{-1} for inshore and offshore waters respectively. Given the variability in reported nutrient concentrations and the absence of impact studies, the overall impact of nutrient inputs on corals is unclear though water quality indicators (Figure 2), suggest a degraded environment.

3.1.2.2 Gulf of Mannar/Palk Bay

The local circulation and ambient environmental conditions of the Gulf of Mannar and Palk Bay are strongly influenced by the changing monsoon seasons (Silas, 1968; Rao et al., 2011; Jagadeesan et al., 2013; Magesh and Krishnakumar, 2019). During the southwest monsoon, the prevailing current flow in the Gulf of Mannar is northwards towards Palk Bay, and a clockwise circulation may be determined throughout the Gulf. During the northeast monsoon, the West India Coastal Current transports low salinity waters from the Bay of Bengal into Palk Bay, but these waters are largely prevented from flowing into the Gulf of Mannar by the shallow bathymetry of Adam's Bridge, an extensive series of limestone shoals (Rao et al., 2011). The monsoon seasons drive significant temporal and spatial variability in ambient environmental conditions (Kumar and Geetha, 2012). Coral reefs in the Gulf of Mannar are found along a string of 21 (2 submerged) coastal islands that are located around 8 km from the main shoreline and thus potentially insulated from the worst effects of pollution by the prevailing circulation. Across the region high-density seagrass ecosystems likely serve as important filters for excessive land-based nutrient input to coastal waters (Purvaja et al., 2018), though seagrass research remains rather limited (Patro et al., 2017). Nevertheless, whilst coral reefs in the Gulf of Mannar are generally considered to be better studied than elsewhere, information on the general oceanography of the Gulf and neighboring Palk Bay has been described as being limited (Rao et al., 2008; Jagadeesan et al., 2013; Jyothibabu et al., 2013). Consequently, there is uncertainty over both ambient nutrient concentrations and nutrient pollution impacts, with anthropogenic pollution effects on corals described as severe (Ramadas and Rajeswari, 2011) and negligible (Kumar and Geetha, 2012) by different studies.

Temperature, salinity, pH, and nutrient concentrations fluctuate seasonally within both the Gulf of Mannar and Palk Bay (Jagadeesan et al., 2013; Anand et al., 2015; Krishnan et al., 2018; Abhilash et al., 2019) though the timing of annual minima may vary spatially. Temperature, pH, and salinity typically reach their annual maxima in May-June, while nutrient concentrations generally reach their annual minima at this time, though some studies have reported annual minima during January-March, particularly in northwest Palk Bay (Sridhar et al., 2006; Sridhar et al., 2008; Anand et al., 2015). Reported nutrient concentrations vary widely between studies (Table 3) but in the Gulf of Mannar highest concentrations are typically measured in November-December during the northern monsoon. Annual maximum nitrate concentrations of 7 to 10 µmol L^{-1} and annual minimum concentrations of ~0.2 µmol L^{-1} are considered broadly typical for the Gulf of Mannar (Anand et al., 2015) yet several studies have reported higher annual maxima which may be indicative of localized pollution sources or terrestrial runoff (e.g. (Thangaradjou and Kannan, 2007; Pitchaikani and Lipton, 2016). Water quality monitoring activities at Tutichorin reported a maximum DIN concentration of ~60 µmol L^{-1} at an inshore (0.5 km) location and ~42 µmol L^{-1} at the offshore location (5km), while DIP concentrations reached maxima of 3.9 and 2.7 µmol L^{-1} respectively (Madeswaran et al., 2018). Despite the reported variability in nutrient concentrations recent near-synoptic nutrient measurements along the southeast Indian coastline during the southwest monsoon period indicate that nutrient concentrations within the Gulf of Mannar may be less variable and lower on average than concentrations found further north along the Kerala coastline (Anandavelu et al., 2020).

Similar variability is reported in nutrient measurements from Palk Bay (Table 3) where significant spatial and temporal variability is evident (Sridhar et al., 2006; Sridhar et al., 2008; Purvaja et al., 2018). Along the northwest coastline of Palk Bay (Kattumavadi region), Sridhar et al. (2006) observed the highest concentrations of nitrate (8.3 µmol L^{-1}), phosphate (2.2 µmol L^{-1}), and silicate (12.5 µmol L^{-1}) variously during the summer/premonsoon months (Apr-Sep) whilst the lowest concentrations were measured during the postmonsoon months (Jan-Mar). Based on a comparison with previous observations, this study reported no detrimental impacts on nutrient concentrations from aquaculture effluents, which are common in this region. However, previous observations collected between 1989 and 1991 observed a broader range of nutrient concentrations and a difference in the timing of annual nutrient maxima, which occurred during the post-monsoon season (Jan-Mar) (Kannan and Kannan, 1996). Notably, terrestrial runoff

TABLE 3 Selected literature estimates of ambient nutrient concentrations within the Gulf of Mannar and Palk Bay.

Location	Timing	Nitrate ($\mu mol\ L^{-1}$)	Phosphate ($\mu mol\ L^{-1}$)	Silicate ($\mu mol\ L^{-1}$)	Ammonium ($\mu mol\ L^{-1}$)	Reference
Gulf of Mannar	Annual range Jan-Dec 2014	2.5 - 10.2	0.3 – 1.9	7.1 – 41.3		Anand et al., 2015
Gulf of Mannar Tamil Nadu coastline – early SW Monsoon)	May-Jun 2014	2.1 – 2.5	0.4 – 0.5	0.6 – 2.4		Anandavelu et al., 2020
Tiruchendur coast	Annual range Jan 2009 - Dec 2010	8.1 – 37.6	0.3 – 1.3	24.5 – 65.6	0.7 – 2.4	Pitchaikani and Lipton, 2016
Gulf of Mannar/Palk Bay (SW monsoon)	July-Aug 2015	0.39 – 14.0	0.01 – 4.3	0.05 – 15.3		Shaju et al., 2019
Gulf of Mannar bioreserve	Annual range Apr 2011- Mar 2012	0.2 – 2.2	0.03 – 0.4		0.5 – 7.1	Kathiravan et al., 2014
Gulf of Mannar bioreserve	Annual range July 1996 – June 1998	5.2 – 18.8	2.3 – 7.3			Thangaradjou and Kannan, 2007
Gulf of Mannar Tutichorin – Inshore - 0.5 km)	Annual range 1990-2015	0.6 – 60.2[a]	BDL[b] – 3.9	BDL [b] – 75.4		Madeswaran et al., 2018
Gulf of Mannar Tutichorin – Offshore 5 km)	Annual range 1990-2015	0.3 – 42.1[a]	BDL[b] – 2.7	BDL [b] – 155.2		Madeswaran et al., 2018
Palk Bay Kattumavadi & Kottaipattinam)	Annual range Apr 1989 – Mar 1991	0.03 – 21.3	0.06 – 8.2	0.2 – 30.5		Kannan and Kannan, 1996
Palk Bay Kattumavadi)	Annual range Apr 2002 – Mar 2003	2.2 – 8.3	1.3 – 2.2	5.2 – 12.5		Sridhar et al., 2006
Palk Bay Munaikadu & Devipattinum)	Annual range Apr 2002 – Mar 2003	0.3 – 7.3	0.1 – 4.1	0.6 – 7.4		Sridhar et al., 2008
Palk Bay (historic range prior to aquaculture – seaweed farming)	Annual range 1951 - 2003	1.8 – 5.2	0.08 – 2	6.4 – 10.2		Abhilash et al., 2019
Palk Bay (following commercial aquaculture – seaweed farming)	Annual range Aug 2013 – Mar 2014	5.8 – 10.0	0.02 – 5.0	1.9 – 14.6		Abhilash et al., 2019

[a]Concentration reported as Dissolved Inorganic Nitrogen (DIN; sum of $NO_3 + NO_2 + NH_4$)
[b]BDL, Below Detection Limit.

during and after the monsoon rains (Oct-Dec) was thought to have caused nutrient influx into coastal waters, resulting in a nitrate concentration of 21.3 $\mu mol\ L^{-1}$.

3.1.2.3 Andaman and Nicobar Islands

The Andaman and Nicobar Islands contain 40-50% of all coral reefs in India (Jeyabaskaran et al., 2007; Jha et al., 2019) and are keenly observed for anthropogenic impacts (Majumdar et al., 2018). Despite their remoteness, human impacts on the islands reefs and marine ecosystems have increased as the human population has grown from ~25,000 in 1901 to ~380,000 in 2011 (Jha et al., 2019), with ~125,000 people living in the capital Port Blair (Jeyabaskaran et al., 2007). Anthropogenic impacts stemming from fishing, sewage discharge, fertilizer, and pesticide runoff have been reported and recent coastal development related to tourism and human population growth is said to have led to significant anthropogenic pollution of coastal waters (Majumdar et al., 2018). Outside of Port Blair, sand mining, which increases water column turbidity and sedimentation, is considered the main cause of coral demise on many islands (Jeyabaskaran et al., 2007). Sea surface temperature typically ranges from 25-29°C, and salinity is comparatively fresh (~32-33) (Varkey et al., 1996; Balasubramanian and Khan, 2001; Brown, 2007). Heavy monsoon rains and terrestrial runoff influence coastal water quality by increasing turbidity and reducing salinity, with near coastal salinities around the Andaman Islands of ~25 or lower (Sampath, 2003).

Water quality studies in the main urbanized areas of Port Blair Bay (South Andaman) and Rangat Bay (Central Andaman) reveal contrasting results, potentially as a result of environmental heterogeneity. Poor water quality conditions and higher average nitrate concentrations have been reported within both Port Blair Bay and Rangat Bay (0.5-0.9 $\mu mol\ NO_3^-\ L^{-1}$) compared to less urbanized areas (<0.35 $\mu mol\ NO_3^-\ L^{-1}$) (Jha et al., 2013; Jha et al., 2015a; Jha et al., 2015b). Muduli et al. (2011) found that inorganic nitrogen ($NO_3^- + NO_2^- + NH_4^+$) concentrations gradual increased from 1.3 $\mu mol\ L^{-1}$ outside Port Blair Bay to 3.9 $\mu mol\ L^{-1}$ at the head of the bay, which they attributed to anthropogenic factors (urban development). Similarly, phosphate concentrations increased along the gradient from 0.2 to 0.9 $\mu mol\ L^{-1}$. Based on long-term nutrient measurements, however, Madeswaran et al. (2018), concluded that

Port Blair Bay had good water quality (Figure 2). Despite these contradictory findings, sewage and fertilizer nutrient inputs are currently thought to be localized to waters around the major urban and agricultural areas. Indeed, several studies demonstrate significant yet very localized impacts within Port Blair Bay or Rangat Bay from runoff and sewage discharge in the form of high faecal coliform bacterial abundances or elevated nutrient concentrations (Sahu et al., 2013; Dheenan et al., 2014; Dheenan et al., 2016). Such studies also suggest rapid dilution of bacterial abundances within ~2 km of discharge points and/or spatially variable impacts from runoff, suggesting that anthropogenic impacts, whilst present, remain spatially localized.

In the South Andaman Islands, elevated nutrient concentrations of 0.2 to 21 µmol NO_3^- L^{-1} and 0.04 to 5 µmol PO_4^{3-} L^{-1} have been reported in near coastal waters in response to land use change with significant seasonal variability in nutrient concentrations driven by monsoon rainfall (Ramesh et al., 2010). Coral reef degradation and coral loss have been reported from the South Andaman Islands, but this has been attributed to repeated temperature-induced bleaching and increased spatial competition with macroalgae and sponges though links to nutrient inputs remain unclear (Malakar et al., 2021).

Fertilizer usage in the Andaman Islands is reportedly below the national average for India (17-32 kg ha^{-1} vs 96.4 kg ha^{-1}; (Jeyabaskaran et al., 2007; Government of India, 2013). During the period 2007-2012 total fertilizer imports to the Andaman Islands peaked at 1800 tons per year (Jha et al., 2019). The increase in fertilizer consumption over time may be linked to the appearance of periodic algal blooms within Port Blair Bay, which have been attributed to terrestrial nutrient runoff, though nutrient concentrations coincident with observed blooms appear only moderately enriched (NO_3^- ~0.6 µmol L^{-1}, PO_4^{3-} 0.2-0.4 µmol L^{-1}) (Goswami et al., 2020). In the less urbanized waters of Aerial Bay on North Andaman Island Jha et al. (2014) reported nutrient concentrations of<0.48 ± 0.34 µmol NO_3^- L^{-1} and<0.35 ± 0.21 µmol PO_4^{3-} L^{-1}, though they noted that nearshore concentrations were generally 50% higher than those recorded in the outer bay area due to the influence of agricultural runoff.

3.1.2.4 Lakshadweep Islands

Located in the Arabian Sea the Lakshadweep Islands, also known as the Laccadive, Minicoy, and Aminidivi Islands, are a collection of 32 islands (10 inhabited) located 200 to 440 kilometers off the southwest coast of India in the Laccadive Sea (Purvaja et al., 2019). The islands make up India's smallest Union Territory, with a total surface area of 32 km^2, a lagoon area of 4,200 km^2, territorial waters of 20,000 km^2, and an exclusive economic zone of 400,000 km^2. Lakshadweep is the only atoll-formed reef in India and forms the northernmost segment of the Chagos-Maldive-Laccadive oceanic ridge. The fishing activities are mostly centered on the 11 islands and reef areas of Perumul Par, Valiyapani, and Cheriyapani, and several submerged banks and open reefs. The Lakshadweep atolls feature different morphological and ecological zones and are considered one of India's most biodiverse places. The Lakshadweep Archipelago has been reported to host 148 species of corals, 91 species of sponges, 114 species of seaweed, 7 species of seagrass, 150 species of crustaceans, 424 species of molluscs, 4 species of lobsters, 225 species of echinoderms, 300 species of ornamental fishes, 601 species of finfishes, 4 species of turtles, 4 species of mammals, 101 species of birds, and many other groups of reef organisms (Government of India, 2012).

Average sea surface temperatures around the islands have increased by 0.2°C per decade since the 1980's (Abhiya et al., 2015), and annual sea surface temperatures now range from ~28.1-31.4°C (Kumaresan et al., 2018). Salinity is typically 35-36 though seasonally can range from 34 to 38. Coral predation by Crown of Thorns starfish (*A. planci*) has increased in selected areas of the Lakshadweep Islands, though the cause remains unclear (Senthilnathan et al., 2014). Nutrient concentrations vary significantly between studies with strong seasonality driven by the monsoons and interannual variability associated with El Niño events likely to explain this (Arora et al., 2019a; Arora et al., 2019b). For the open waters of the central Lakshadweep Sea, Sengupta et al. (1979) reported average nutrient concentrations during October of 4.21 µmol NO_3^- L^{-1} and 0.97 µmol PO_4^{3-} L^{-1} and concentrations of 0.29 µmol NO_3^- L^{-1} and 0.46 µmol PO_4^{3-} L^{-1} during March/April, further noting the widespread seasonal appearance of nitrate depleted surface waters (0-75 m) and dense *Trichodesmium* blooms during April coincident with ammonium concentrations of up to 5.5 µmol NH_4^+ L^{-1}. In contrast, data reported by Kumaresan et al. (2018) indicated mean annual nutrient concentrations of 6.08 µmol NO_3^- L^{-1} and 0.93 µmol PO_4^{3-} L^{-1} for the Lakshadweep Sea and strong seasonal and spatial variability, with concentrations varying from 0.29 to 14.8 µmol NO_3^- L^{-1}, and from 0.29 to 2.1 µmol PO_4^{3-} L^{-1} across the study period. Notably, average nutrient concentrations were highest during the summer months (Mar-May) in contrast to the results of Sengupta et al. (1979).

In more localized studies conducted around the inhabited Kavaratti atoll, nutrient concentrations appear more variable. Citing studies from the 1970's, Balasubramanian and Khan (2001) inferred limited seasonality and low nutrient concentrations of<0.3 µmol NO_3^- L^{-1} and<0.4 µmol PO_4^{3-} L^{-1} both within the lagoon and in the surrounding open sea. More recently, concentrations of 1.95 to 8.96 µmol NO_3^- L^{-1}, 0.12 to 0.33 µmol PO_4^{3-} L^{-1}, and 2.47 to 5.93 µmol Si L^{-1} have been reported from within the extensive lagoon area (Antony et al., 2020). Notably, as measurements from two neighboring offshore sites indicated lower nutrient concentration ranges of 1.99 to 5.82 µmol NO_3^- L^{-1}, 0.07 to 0.21 µmol PO_4^{3-} L^{-1}, and 2.50 to 4.40 µmol Si L^{-1}, Antony et al. (2020) concluded that anthropogenic nutrient enrichment of Kavaratti lagoon waters was now evident. Whilst the observations reported by Antony et al. (2020) do appear substantially higher than those recorded from this lagoon previously (e.g. (Karati et al., 2017) (<0.5 µmol NO_3^- L^{-1}) they remain broadly comparable to concentrations reported by Kumaresan et al. (2018) for the surrounding Lakshadweep Sea such that high variability in lagoonal nutrient concentrations due to water exchange cannot be fully ruled out. Karati et al. (2017) reported spatially extensive nutrient measurements across the Kavaratti lagoon during the peak of the 2016 El Niño event and obtained an average nitrate concentration of only 0.46 µmol NO_3^- L^{-1}, with the average concentration reducing to 0.16 µmol NO_3^- L^{-1} as the

El Niño weakened. Similarly, average phosphate concentrations were 0.26 µmol PO_4^{3-} L^{-1} and 0.13 µmol PO_4^{3-} L^{-1} during and after the El Niño event. High ammonium concentrations of 5.4 to 8.7 µmol NH_4^+ L^{-1} were also recorded within the lagoon, which is noteworthy as recent estimates of sewage discharge volumes (50,000 to 120,000 liters per day (Purvaja et al., 2019)), have increased expectations of anthropogenic impacts. Yet in March 2006 Robin et al. (2012) measured concentrations of ~3.2 µmol NO_3^- L^{-1}, 0.65 µmol Si L^{-1}, ~0.25 µmol PO_4^{3-} L^{-1} and 0.34 µmol NH_4^+ L^{-1} thus, it is far from clear whether existing nutrient measurements from the Lakshadweep Islands are sufficient to document anthropogenic impacts here, with long-term observations suggesting an absence of nutrient impacts (Figure 2; Madeswaran et al., 2018).

According to climatology, the Arabian Sea generally sees only one cyclonic storm every year. However, cyclonic disturbances over the Arabian Sea have increased to nearly four per year on average between 2015 and 2020, with cyclones and severe cyclones increasing to two per year on average. A report published in 2015 by the National Disaster Management Authority identified a rise in the severity and frequency of cyclones on India's western coast as a result of global warming (NDMA, 2015). Two recent cyclones (Ockhi in Nov 2017; Tauktae in May 2021) caused severe devastation across the islands with impacts observed up to a depth of 20 m in the eastern reef system of Kavaratti Island following cyclone Ockhi (Riyas et al., 2020).

3.1.3 Reported impacts of nutrient pollution

Despite Indian corals having been widely studied, particularly since the 1960's (Saroj et al., 2016; De et al., 2017), it remains difficult to quantify or satisfactorily attribute nutrient pollution impacts to long-term changes in coral coverage or coral health in most cases. Indeed, most time-series observations of Indian coral ecosystems post-date the first global bleaching event in 1998, with the duration of many studies limited to only a few years or less (Chandra et al., 2021). Algal overgrowth is identified as a threat to Indian corals generally (MoEF, 2014), with exotic seaweed cultivation considered a particular concern (Patterson et al., 2015). Indeed, whilst coral overgrowth by green algae is reported along the northern (Keelakarai) coastline in the Gulf of Mannar and is generally attributed to nutrient inputs (Rajasuriya et al., 2000), with increased macroalgal coverage subsequently linked to reductions in reef-associated faunal biodiversity (Ramesh et al., 2020), the cause for such changes is not always clear. Effluent discharge from several major industries, including copper smelting, chemical manufacture, and salt panning, contribute to broader pollution impacts in this region likely obscuring impacts from nutrient inputs alone (Samuel et al., 2012; Anand et al., 2015). In addition, ash residue from a regional thermal power station (at Thoothukudi) is also released directly into the coastal zone, likely increasing water column turbidity or smothering localized benthic habitats (Anand et al., 2015). In neighboring Palk Bay, the appearance of epiphytes within seagrass meadows has been attributed to nutrient inputs though such appearances do not yet appear to be widespread (Purvaja et al., 2018). Abhilash et al. (2019) found no evidence of any negative impacts on water quality from commercial-scale seaweed cultivation activities in Palk Bay, though their data appear to show a significant temporal (decadal) increase in ambient nitrate and phosphate concentrations, which may be indicative of longer-term changes in seasonal nutrient inputs to Palk Bay from the Bay of Bengal (mainly riverine in origin) and neighboring coastal waters. In the Andaman and Nicobar Islands deforestation has increased sediment (and likely nutrient) loadings to nearshore reefs (Rajasuriya et al., 2000), while a growing tourism industry has increased sewage release within the Andaman Islands (Majumdar et al., 2018). Information on nutrient pollution impacts on coral reefs in the Gulf of Kachchh and along the west Indian coast remains limited, though in the Gulf of Kachchh corals are now considered exposed to significant anthropogenic disturbances (Raghuraman et al., 2013; Madeswaran et al., 2018) in addition to strong environmental fluctuations (Muley et al., 2000; Chandran et al., 2021).

Quantified impacts on corals and other related habitats from agricultural runoff remain limited, but river and sewage discharges have been linked to an increase in the frequency and distribution of phytoplankton blooms in Indian coastal waters over recent decades (Jameson, 2012; Padmakumar et al., 2012; Samuel et al., 2012; BOBLME, 2015; Ramesh et al., 2017; Karthik et al., 2020). Between 1908-2009, 101 algal blooms were reported along the western and eastern coastlines, and in particular along the southwest coastline of India, with an increase in frequency after 1950 (D'Silva et al., 2012). These algal blooms were found primarily in waters off the coasts of Goa, Mangalore, Kozhikode, and Kerala (west coast) and attributed to seasonal upwelling and/or high riverine discharge following the southwest monsoon rains. Padmakumar et al. (2012), reported a far greater frequency of blooms identifying 80 algal blooms around India between 1997 and 2010 alone, with a strong geographical focus around the southwest coast between Goa and Kollam. Historically, algal blooms within the main coral regions have been infrequent with only 3 cyanobacterial blooms reported in the Lakshadweep Islands, 2 dinoflagellate blooms in the Andaman and Nicobar Islands, 2 cyanobacterial blooms in Palk Bay, and a single dinoflagellate bloom in the Gulf of Mannar, nevertheless, almost all occurred after 1950 (D'Silva et al., 2012; Padmakumar et al., 2012). No blooms have been reported from the Gulf of Kachchh. More recently, Shaju et al. (2019) argued that aquaculture activities in the Gulf of Mannar were very likely linked to increased nutrient concentrations, decreased water quality conditions, and the appearance of a *Trichodesmium* dominated algal bloom. While unusual phytoplankton blooms can be indicative of long-term changes to nutrient conditions, recently reported blooms of the bioluminescent dinoflagellate *Noctiluca scintillans* within the Gulf of Mannar (Gopakumar et al., 2009; D'Silva et al., 2012; Samuel et al., 2012; Shunmugaraj et al., 2020) and their role in creating temporarily hypoxic conditions leading to increased coral mortality (Raj et al., 2020) should be interpreted cautiously. *N. scintillans* is not only widespread in the Arabian Sea (Gomes et al., 2008; Gomes et al., 2014) but its geographic distribution has changed significantly in recent years with these changes linked to freshwater inputs and snow loss over the Himalayan-Tibetan plateau (Goes et al., 2020). The presence of *N. scintillans* in Indian waters is also common and multiple blooms of this species have been recorded (D'Silva et al., 2012). Recent changes in the distribution of *N. scintillans* towards coral regions may therefore reflect wider ocean warming trends and the spread of hypoxic conditions throughout the Indian Ocean (Gomes et al., 2014) rather than the impact of local nutrient inputs alone.

3.1.4 Current protections and safeguards

Across India, around 35 million people live within 30 km of a coral reef, with many coastal communities dependent upon reefs for livelihoods and for the provision of edible protein (Burke et al., 2011; Balaji et al., 2012; Raghuraman et al., 2013). Ecosystem services rendered by coral reefs include fisheries, tourism and recreation, maintenance of biodiversity, construction materials for Islanders, and protection of coastal areas against erosion, flooding and salinity ingression (Hoon and Tamelander, 2005; Dixit et al., 2010). Despite legal safeguards and other conservation initiatives to protect India's reefs, localized anthropogenic factors continue to have significant detrimental impacts on the coastal zone and associated ecosystems, which are considered to be substantially damaged due to human pressures (Latha and Prasad, 2010). While destructive fishing practices, lime production and coral mining were once thought to be the most serious threats facing corals (Patterson, 2002), with the visual impacts of these activities perhaps explaining why reports of their negative impacts are more widespread and appeared earlier in the literature, the list of threats has grown. Damage due to bottom trawling, siltation due to coastal developmental, coral disease due to pollution and sewage discharge, exotic algal invasion, and increased coral mortality due to pelagic algal blooms are increasingly common (Wilson et al., 2005; Chandrasekaran et al., 2008; Thinesh et al., 2013; Machendiranathan et al., 2016; Asir et al., 2020; Raj et al., 2020). Growing tourist inflow and recreational diving have led to increased physical damage with subsequent impacts on the ecological dynamics of the reef (Purvaja et al., 2019). In more recent years, heavy industry, rapid coastal development, and anthropogenic shoreline change have added additional stress to the marine environment in the form of increased sedimentation, turbidity and industrial and municipal discharges (Patterson, 2002; MoEF, 2014; Magesh and Krishnakumar, 2019; Jinoj et al., 2020; Jinoj et al., 2021). Proposed major infrastructure projects, such as the long-planned Sethusamudram Shipping Canal, threaten to deliberately dredge the centre of the Gulf of Mannar and through the limestone shoals of Adam's Bridge to create a navigable sea route, with the full impacts on the surrounding marine habitats poorly known (Rao et al., 2008).

The negative effects of marine nutrient pollution are recognized regionally but quantification of long-term impacts on the coastal zone remain difficult to obtain due to the limitations of existing data (Kaly, 2004; BOBLME, 2010; BOBLME, 2015). There has been only limited specific study of the links between the discharge of dissolved nutrients and coral health in India. At the same time, various measures, both scientific and managerial, already exist to minimize pollution impacts particularly on human health or on fisheries but there remains a broader need to protect reef ecosystems for the benefit of communities that depend upon the services they provide. Such communities are at high risk and vulnerable to factors that degrade the reef environment. Coral transplantation involving local communities has been carried out in attempts to promote reef recovery and restoration in degraded areas (Mathews, 2008; Melkani, 2008; Subburaman et al., 2014; Kumar et al., 2017; Nanajkar et al., 2019). Such efforts, however, do not on their own address underlying pollution issues.

Proper management of coral reefs within Marine Protected Areas (MPAs) has a beneficial influence on the resilience and functioning of reefs (Roberts et al., 2017; Topor et al., 2019) and provides additional benefits to coastal communities. The mix of National Parks, Sanctuaries and Biosphere Reserves in India comprises 25 MPAs around the mainland and 106 MPAs across the island territories. In the MPA network, the Marine National Park and Marine Sanctuary in the Gulf of Kachchh, the Malvan Marine Sanctuary in Maharashtra, the Gulf of Mannar Marine National Park in Tamil Nadu (part of the Gulf of Mannar Biosphere Reserve), the many national parks, sanctuaries, and biosphere reserves in the Andaman and Nicobar Islands, and the Pitti Bird Sanctuary in the Lakshadweep Islands provide legal protection for coral reefs and their biodiversity, in some cases since the 1970s. In addition to location-specific protection status through MPAs, species-specific protection is also provided for corals in India. The Wild Life (Protection) Act, 1972 includes reef-building corals, black corals, organ pipe corals, fire corals, and sea fans as Schedule-I species and provides legal protection against harvesting them in India. Under the Coastal Regulation Zone (CRZ) Notifications of 1991, 2011 and 2019, corals, coral reefs, and associated biodiversity are also designated as Ecologically Sensitive Areas (ESA) - Coastal Regulation Zone-I (CRZ-I) areas. The ecological sensitivity of reef habitats is further specified under the Environment (Protection) Act, 1986, which both prohibits and regulates coastal development activities in and around reef ecosystems (MoEF, 1991; MoEF, 2011; MoEFCC, 2019). The CRZ Notification, 2011 and 2019, also established effluent standards under the Environment (Protection) Act, 1986, and specifically prohibits the discharge of untreated wastewater and effluents onto the coast.

In principle, therefore, coral reef ecosystems are accorded the highest legal protection status to ensure their long-term protection and conservation. As part of this protected status, detailed Coastal Zone Management Plans (CZMP) are required by the respective coastal state or union territory to monitor and implement the provisions of the CRZ Notification for conserving coastal habitats and to aid the sustainable development of coastal stretches along the mainland coast of India. Special attention is paid to the protection and conservation of ESA during the preparation of developmental plans.

Coral reefs are classified as CRZ 1A (ii), and the conservation, protection and management framework for ecologically sensitive areas is provided in Annexure 1, 1.2 – iv a & b which reads,

a. Active and live corals and coral reefs identified and delineated shall be declared and notified as an ESA under the Environment (Protection) Act 1986 (29 of 1986);
b. It shall be ensured that no activities that are detrimental to the health of corals, coral reefs and their associated biodiversity, such as mining, effluent and sewage

discharge, dredging, ballast water discharge, ship washings, fishing other than traditional non-destructive fisheries, construction activities and the like, are taken up in and around the coral areas.

The CRZ protects coastal areas along the coastline of India. Amendments to the CRZ Notification introduced simultaneously with the Island Protection Zone (IPZ) legislation in 2011 and 2019 resulted in a division of the protective legislation. Currently, the CRZ applies only to the mainland coast of India, while the IPZ applies only to the islands of Andaman and Nicobar and Lakshadweep. While the islands of a large geographical area are managed as per the Island Coastal Regulation Zone (ICRZ) Plan, the rest of the islands in Andaman and Nicobar and all the islands in Lakshadweep are to be managed in accordance with the Integrated Island Management (IIM) Plan, as provided under the IPZ Notification. The ICRZ shall ensure that no activities that are detrimental to the health of corals, coral reefs and their associated biodiversity, such as mining, effluent and sewage discharge, dredging, ballast water discharge, ship washings, fishing other than traditional non-destructive fisheries, construction activities and the like, are taken up in and around the coral areas.

That there exist clear requirements to consider the impact of effluent and sewage discharge on coral ecosystems and that the evidence to assess the impact of such discharges is currently limited argues strongly for improvements and broadening of coral monitoring activities and coordinated efforts within existing water quality monitoring programmes to ensure corals are fully considered going forward.

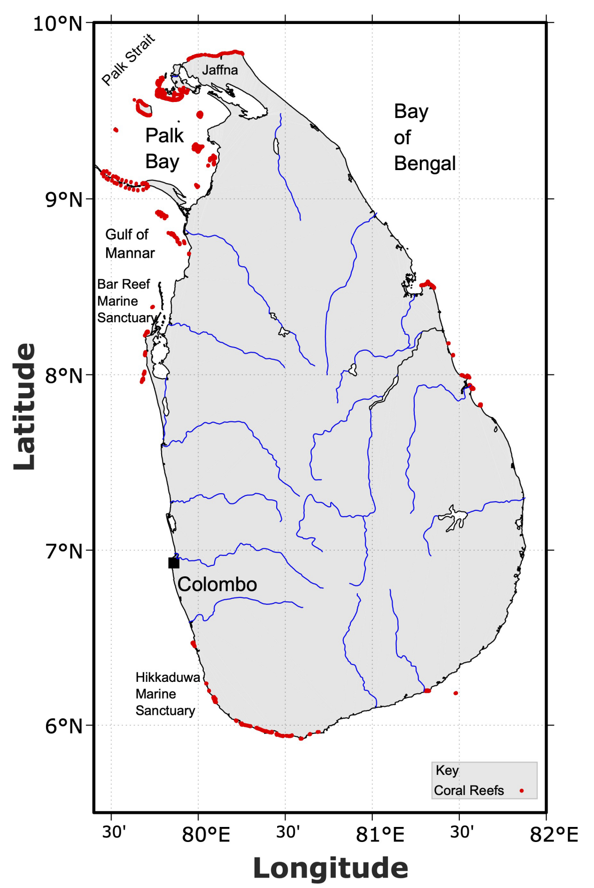

FIGURE 3
Map showing the distribution of coral reefs in Sri Lankan waters and placement of major river systems. Coral distribution based on data compiled by UNEP-WCMC et al. (2018).

3.2 Sri Lanka

3.2.1 Coral reef extent, condition and environmental setting

Coral reefs are found discontinuously around the Sri Lankan coastline (Figure 3). Estimates of total coral reef coverage vary widely from 680 km^2 or 0.24% of the global coral area (Spalding et al., 2001) to more recent estimates of 475 km^2 (0.18%; (Bahuguna et al., 2013)), though the latter estimate may be inaccurate as it omits an important area between Mannar Island and the Kalpitiya peninsula where the largest shallow water coral reef patches are found. Modelled projections of coral conducive habitat area however reach 2739 km^2 (1.05%; (Ellepola et al., 2021)). Coral reef types consist of fringing, patch and barrier reefs with the near-shore distribution of corals along approximately 2% of the Sri Lankan coastline (Rajasuriya and White, 1995; Kumara et al., 2008). Major coral habitats are found in the north/northwest, east, and southwest of the country, with the distribution said to be strongly influenced by ambient nutrient concentrations (Ellepola et al., 2021). Coral diversity is currently estimated to be ~245 species (Rajasuriya et al., 2004; Rajasuriya, 2012; Weerakoon et al., 2018; Arulananthan et al., 2021), rising from earlier estimates of ~170 species in the early 1990's (Rajasuriya and White, 1995). The full diversity is likely higher due to new species reports emerging from the poorly studied Jaffna Peninsula (Arulananthan et al., 2021).

Sri Lanka's climate is tropical, with two main seasons namely the southwest monsoon (May-Sep) and the NE monsoon (Oct-Feb) periods, both of which bring heavy rain. The 103 rivers in Sri Lanka discharge considerable river-borne sediment loads creating a particularly turbid coastal environment. Mean sediment loads within Sri Lankan rivers are above average compared to other Asian rivers (Silva et al., 2005). The continental shelf around Sri Lanka covers approximately 30,000 km^2, has an average width of 22 km, and typical water depths of 20-65 m, which increase rapidly beyond the shelf break to depths of 3500 m (UNEP, 1986; BOBLME, 2013). Sea surface temperatures typically range from 26-30°C over the year, and like the prevailing surface currents, are heavily influenced by the monsoon seasons (Rajasuriya, 2002; Survey Department of Sri Lanka, 2007). Ocean currents reverse seasonally along the eastern, western and southern coasts due to the monsoons (Schott and McCreary, 2001; Shankar et al., 2002; de Vos et al., 2014) whilst upwelling occurs along the southern and southeastern coastlines during the southwest monsoon period (Vinayachandran, 2004; Yapa, 2009; de Vos et al., 2014).

Reefs on the west coast of Sri Lanka are mainly based on sandstone/limestone foundations and typically have very low coral coverage of less than 5%, an exception being Bar Reef which is a comparatively biodiversity rich coral habitat. The majority of coral species belong to the family Faviidae. Small colonies of tabulate Acroporids are found occasionally. However, the reefs support high fish diversity as well as many species of invertebrates, including 5 species of spiny lobsters (*Panulirus Homarus*, *P. longipes, P. pencillatus, P. ornatus, and P. polyphagus*). Isolated coral patches with an extent of about 1 hectare occur 15 to 20 km offshore on the west coast at a depth of about 20 m (Survey Department of Sri Lanka, 2007). Many coral species live in these coral patches, mainly in the families of Faviidae, Poritidae, Mussidae and Pocilloporidae, but due to the water depth, they lack the branching Acroporids like *Acropora formosa*, tabulate *A. hyacinthus, A. cytherea* and the foliose *Montipora* spp. that are common on shallow coral reefs.

Coral reefs in northern Sri Lanka, particularly around the Jaffna Peninsula and in the Gulf of Mannar, are relatively undamaged and minimally impacted by human activities (Rajasuriya, 2008). In contrast, coral reefs found elsewhere around Sri Lanka have suffered extensive degradation, and few, if any, can be considered pristine. Rapid deterioration due variously to mining, blast fishing, *A. planci* infestation, and sedimentation was noted almost four decades ago (UNEP, 1986). Since then, repeated bleaching events, tsunami damage, and poor fishing practices have continued to significantly degrade Sri Lanka's reefs (Rajasuriya, 2005; Rajasuriya, 2008; Souter et al., 2021). While the overall condition of corals in Sri Lanka is considered poor (Tamelander and Rajasuriya, 2008), resident coral populations have recovered well following bleaching events, indicating that environmental conditions remain favourable (Chandra et al., 2021). Coral reefs along parts of eastern Sri Lanka, particularly within Pigeon Island National Park (off Trincomalee), have also experienced and recovered from *A. planci* infestations including a notable outbreak in the late 1960s and early 1970s that led to the localised loss of coral cover (De Bruin, 1972; Rajasuriya et al., 2005). While this outbreak was ultimately short lived, with coral recovery noted over subsequent decades (Rajasuriya et al., 2005), ongoing local efforts to locate and remove *A. planci* occasionally return large numbers of this corallivorous starfish.

The major coral regions include notable formations at Bar Reef, Silavathurai, and Arippu and Vankalai in the Gulf of Mannar, in the northern islands around Jaffna Peninsula, and along the southern and eastern coastlines (Rajasuriya, 2002). Current reef locations, status, and major threats are summarized in Table 4. Coral reefs around Sri Lanka have long been recognized for their rich biodiversity, yet almost all reef formations have endured threats from destructive fishing practices, overfishing, coastal development (sedimentation and alteration of nearshore currents), mining, and pollution for at least the last 20 years and very often longer (Rajasuriya et al., 2000; Rajasuriya et al., 2004).

3.2.2 Water quality indicators

Pollution of inland and coastal waters has been a noted problem in Sri Lanka for several decades and arises largely from agricultural runoff, industrial activities, coastal development, sewage, and maritime activities (UNEP, 1986; Bandara, 2003; Joseph, 2003; BOBLME, 2013; BOBLME, 2015). Despite the coastal zone hosting 4.6 million people, or around 25% of the country's population, and over 60% of industrial activity (Rajarathna and Nianthi, 2019), studies on coastal water quality remain very limited (Ileperuma, 2000; Joseph, 2003; BOBLME, 2013; Niroshana et al., 2013; BOBLME, 2015; Manage et al., 2022). Many industries continue to openly discharge effluents without pre-treatment directly into fresh and marine waters (UNEP, 1986; Bandara, 2003; Geekiyanage et al., 2015). Riverine habitats have undergone a rapid environmental change in response to anthropogenic activities (Amarathunga et al., 2013), whilst estuaries and coastal lagoons experience notable pollution (Gammanpila, 2010). Historically, there has been heavy national usage of fertilizers in Sri Lanka compared to other South Asian countries, with fertilizer application increasing over 30-fold between 1950 and 2000 reaching over 600,000 tons per annum (Joseph, 2003). Recent Government efforts since 2015 have decreased fertilizer consumption significantly (World Bank, 2021). Widespread contamination of coastal aquifers (Jayasingha et al., 2011; Jayasingha et al., 2012), coastal lagoons (e.g. (Gammanpila, 2010)) and the occurrence of coastal algal blooms have been linked to nitrate pollution of coastal waters from agricultural runoff, though the evidence base to assess the breadth and severity of pollution impacts on the marine environment remains small, and many impacts are largely unknown (Ileperuma, 2000; Joseph, 2003; Geekiyanage et al., 2015). In recent years, high levels of organic and inorganic nutrient pollution have been reported in the coastal waters along parts of the west coast, south of Negombo (Hettige et al., 2014), inferred along parts of the eastern coast (Perera, 2019), or measured in coastal lagoons (Gammanpila, 2010; Harris and Vinobaba, 2013), while oil and organic nutrient pollution has been reported within fishing areas along the southern coast (Niroshana et al., 2013; Weerasekara et al., 2015). Common indicators of coastal nutrient enrichment, such as increased chlorophyll concentrations or decreased dissolved oxygen levels, have not been widely recorded from Sri Lankan coastal waters (Bandara, 2003; BOBLME, 2013), and only rarely within surface water and estuarine settings (Silva, 1996; Dahanayaka et al., 2013). Recent observations of high biological oxygen demand in coastal harbor settings may have multiple origins, including fish processing, oil and grease discharge, and sewage input (Bandara, 2003; Niroshana et al., 2013; Weerasekara et al., 2015; Manage et al., 2022). In some coastal areas, eutrophication effects may be masked by seasonal upwelling, particularly along the southeastern and southern coastlines during the southwest monsoon months, where reduced sea surface temperatures and elevated chlorophyll concentrations may be advected and overlap with known coral formations (Vinayachandran, 2004; Yapa, 2009).

Industries like agriculture, aquaculture, and fisheries remain central to the Sri Lankan economy, yet with approximately one-third of the land area in Sri Lanka cultivated, including 17% of the coastal zone, widespread and heavy usage of fertilizers, pesticides, and herbicides result in high nutrient discharge to the coastal environment (BOBLME, 2013). Despite existing detailed summaries of the major pollution sources, the magnitude of pollution inputs to coastal waters and the impacts on specific habitats like coral ecosystems remain poorly documented. A general lack of observational data, particularly time-series data,

TABLE 4 Major coral reef locations in Sri Lanka, their status and major threats facing their long-term future.

Location	Status	Major threats
Arippu, Silavathurai, Vankalai in the Gulf of Mannar	Arippu and Silavathurai reefs are relatively healthy, some bleaching in the past, but recovery is good. Vankalai reef is degraded due to bleaching and human impacts, poor recovery	Coral bleaching, destructive fishing, and overexploitation of resources
Maldiva Bank, Palk Bay (Maldiva Bank is a small coral reef located within the Vidattaltivu Nature Reserve.	Partially degraded	Coral bleaching, destructive fishing, and high sedimentation
Coral reefs around the Jaffna Peninsula including Palk Bay	Partially degraded, recovery evident.	Coral bleaching, physical damage due to construction of anchorages, destructive fishing, high sedimentation.
Bar Reef Sanctuary	Shallow coral patches (up to 8m depth), heavily degraded. Recovery can be seen among some patches.	Coral bleaching, overexploitation of fish, destructive fishing, boat anchors, Crown of Thorns starfish
Kandakuliya	Heavily damaged, patchy recovery can be seen	Coral bleaching, destructive fishing, boat anchors, harvesting
Talawila	Shallow areas damaged, recovery is visible	Coral bleaching, destructive fishing, boat anchors, harvesting
Chilaw	Degraded	Coral bleaching, destructive fishing, harvesting
Negombo	Coral areas are below 12 m and offshore. Damage is visible. Recovery is also evident	Coral bleaching, destructive fishing, harvesting, sedimentation, pollution, sand extraction for coastal development
Colombo	Coral reefs are offshore and below 20 m. Nearshore sandstone reefs degraded. Offshore coral areas are relatively good and recovery can be seen.	Sedimentation, harvesting, pollution, heavy fishing pressure, coastal development and pollution. Ship anchoring on offshore coral reef area is a threat as they are located within the limits of Colombo Port anchorage area
Ambalangoda to Hikkaduwa	Nearshore reefs are degraded, offshore reefs are mainly rock and sandstone.	Coral bleaching, sedimentation, destructive fishing, uncontrolled harvesting, pollution and coastal development
Hikkaduwa Marine Sanctuary	Degraded, partially recovery	Coral bleaching, high level of sedimentation, boat anchoring, reef trampling, pollution, nutrients, use of glass bottom boats, tourism, increase of macroalgae, coastal development
Galle (including Rumasssala reef)	Degraded, chances of recovery are very low	Coral bleaching, high level of sedimentation, uncontrolled harvesting, destructive fishing, urban pollution, nutrients and coastal pollution and nearby Galle Port activities.
Unawatuna	Degraded, very low recovery	Coral bleaching, sedimentation, pollution, nutrients, reef trampling, tourism, anchors, uncontrolled harvesting and coastal development
Kapparatota, Weligama	Degraded, chances of recovery extremely low	Coral bleaching, sedimentation, pollution, nutrients, reef trampling, anchors, destructive collecting, uncontrolled harvesting and coastal development
Polhena, Matara	Degraded, very low recovery	Coral bleaching, sedimentation, harvesting, coconut husk seasoning, tourism
Tangalle	Degraded, very low recovery	Coral bleaching, sedimentation, reef trampling, souvenir collection, tourism, pollution, coastal development
Great and Little Basses Sanctuary (mainly rock ridges with<10% coral cover)	Good	Uncontrolled harvesting
Kalmunai & Kathankudi	Bleached and recovering	Coral bleaching, sedimentation, uncontrolled resource exploitation, and pollution
Passikudah - Kalkudah	Bleached and low recovery	Coral bleaching, sedimentation, freshwater, tourism, and pollution
Kayankerni Marine Sanctuary	Severely bleached in 2019 – 2020. Recovering	Coral bleaching, sedimentation, resource exploitation, pollution.
Dutch Bay, Trincomalee	Degraded, low recovery	Coral bleaching, sedimentation, pollution
Pigeon Island Marine National Park in Trincomalee	Coral reefs are recovering	Coral bleaching, reef trampling, visitor pressure. Areas outside the national park have destructive fishing, uncontrolled harvesting.

with which to study water quality, ascertain temporal trends, or identify pollution hotspots, makes quantification of pollution impacts difficult. Consequently, Joseph (2003) ranked pollution as fourth on the list of causes of reef damage and degradation in Sri Lanka, after coral mining, coral bleaching, and destructive fishing practices. Coral mining, however, stopped after the 2004 Indian Ocean tsunami due to the strict enforcement of a national ban on the use of lime kilns and the prohibition of using coral lime for construction purposes (Rajasuriya, 2005). The degradation of Sri Lankan coral reefs due specifically to nitrogen inputs to coastal waters has not been adequately ascertained, and the present evidence base is insufficient for quantifying the extent of nutrient over-enrichment of coastal waters. There are currently no studies that investigate the physiological impacts of nutrient enrichment on Sri Lankan corals, and past studies on coral reef habitats are now considered to be outdated (Ellepola et al., 2021).

Nutrient observations for the coastal zone are sparse. Along southern Sri Lanka, Ekanayaka et al. (2016) reported mean concentrations of 7.07 ± 5.9 µmol L^{-1} for nitrate, and 1.42 ± 0.78 µmol L^{-1} for phosphate during the southwest monsoon and inter-monsoon months (August to October). This study included sites with known coral coverage, including Rumassala Marine Sanctuary and Polhena Reef Fishery Managed Area (Perera and Vos, 2007), suggesting that relatively high ambient nutrient concentrations may occur seasonally, possibly in response to upwelling. Such concentrations are comparable to typical nutrient concentrations reported for Sri Lankan rivers of ~10 µmol N L^{-1} and 1.6 µmol P L^{-1} (Silva et al., 2005), but higher than those reported along the western coastline (0.3-3 µmol NO_3^- L^{-1}; (Hettige et al., 2014)). Historic nutrient data in the World Ocean Database (WOD 2018) for the region between 5 and 10°N and 79 and 82.5°E, consists of only 27 observations of nitrate and 37 observations of phosphate for the period 1960 to 1995, with observations mainly from waters southwest of Sri Lanka. The available data, however, is useful for providing an order of magnitude estimate of ambient nutrient concentrations in offshore waters and indicates near surface (5 m) nitrate concentrations of 0.01–2.2 µmol L^{-1} (mean ± stdev 0.25 ± 0.47 µmol L^{-1}) and phosphate concentrations of 0.04 –0.86 µmol L^{-1} (mean ± stdev 0.23 ± 0.19 µmol L^{-1}). Near surface N:P molar ratios range from 0.01 to 5.44 (1.18 ± 1.1) indicating a nitrogen limited environment.

During a year-long study of the coastal Batticaloa lagoon (east coast), Harris and Vinobaba (2013) observed significant nitrate concentrations of 12 to 82 µmol L^{-1} and phosphate concentrations of ~0.9 to ~11 µmol L^{-1}. The lagoon itself has a limited and intermittent exchange with the open ocean, but does receive water from the surrounding land, so the reported nutrient concentrations are likely elevated in the lagoon due to a lack of flushing. Weerasekara et al. (2015) meanwhile, investigated water pollution in several fishing harbors along the south and southwest coasts that do exchange freely with the open ocean and observed typical nitrate concentrations of ~7 to ~9 µmol L^{-1} and phosphate concentrations of ~4 to 10 µmol L^{-1}. A similar study from another fishing harbor reported comparable maximum phosphate concentrations of ~9 µmol L^{-1} (Niroshana et al., 2013). Both studies confirmed severe oil pollution, chemical pollution, and microbial contamination of harbor waters. Perhaps the most extensive study of coastal water quality to date is that of Manage et al. (2022), who surveyed conditions around the southern and western coastlines, a region home to over 40% of Sri Lanka's population and a region receiving considerable anthropogenic inputs. Though the overall conclusion was that environmental conditions had been degraded by anthropogenic activities, with widespread observations of faecal coliform and oil and grease contamination, the reported nutrient observations broadly suggested the presence of nitrogen limitation within coastal waters and of limited nitrogen enrichment. Mean nitrate concentrations along both the western and southern coastlines were only 0.7 µmol L^{-1}, and despite sampling a range of locations including fishing harbors, exposed beach front areas, tributaries, river plumes, coastal lagoons and sea canals, no enrichment was observed. Ammonium concentrations were more variable reaching ~2.5 µmol L^{-1} along the western coastline and 4.4 µmol L^{-1} along the southern coastline but in both regions the mean concentrations were comparable at 0.23 and 0.27 µmol L^{-1}, respectively. Phosphate concentrations meanwhile reached ~13.6 µmol L^{-1} and 7.2 µmol L^{-1}, along the western and southern coastlines, and were generally elevated averaging 5.3 µmol L^{-1} and 4.5 µmol L^{-1}.

3.2.3 Reported impacts of nutrient pollution

Sri Lankan reefs have been extensively damaged by human activities related to coral mining, sedimentation, destructive fishing practices, boat anchors, and latterly by various chemical pollutants, though the effects and significance of any particular factor vary regionally (Öhman et al., 1993; Rajasuriya, 1997; Perera et al., 2002; Rajasuriya, 2002; CC & CRMD, 2018; Weerakoon et al., 2018). Degradation of corals due to pollution from discarded plastics and oil, deliberate discharge of sewage or industrial effluents, and agricultural runoff was noted almost 30 years ago (Öhman et al., 1993), but more recent assessments suggest that the impacts of sewage and other pollutants, including oil, may now be acute in some regions (BOBLME, 2013; Niroshana et al., 2013; MoMD&E, 2016; CC & CRMD, 2018). Currently, the impact of destructive fishing practices, sedimentation and bleaching is generally more evident and considered more directly connected to the long-term decline of Sri Lankan coral reef habitats than pollution (Rajasuriya and White, 1995; Joseph, 2003; MoMD&E, 2016; CC & CRMD, 2018). Recent maritime disasters, such as the sinking of the X-Press Pearl container ship in June 2021 (UNEP, 2021; Rubesinghe et al., 2022), have raised public awareness of the environmental and human health consequences of marine pollution, which may yet lead to longer-term and more widespread monitoring of coastal waters.

Nevertheless, extensive coral bleaching due to increased ocean temperatures with little or no post-bleaching recovery of coral communities in regions where local stress factors are evident represents a major existential threat to Sri Lankan corals (Perera et al., 2002), and suggests that that recovery from bleaching may be weaker in those regions where local stress (e.g. pollution) is high. However, monitoring efforts to quantify coral bleaching impacts on biodiversity, coral reef health and the contribution to the loss of fisheries do not specifically consider the additional pressures represented by nutrient pollution. Similarly, while overfishing and destructive fishing practices (including the use of explosives) are acknowledged challenges facing coral reef management, overfishing persists due to lack of enforcement, despite decreased fishery catches (Silva et al., 2018).

3.2.4 Current protections and safeguards

Until recent maritime accidents such as the X-Press Pearl sinking caused financial hardship for fishers and their dependents, many coastal communities were said to be unaware of the links between marine ecosystem health and their livelihoods and subsequently would not engage in management efforts (Joseph, 2003). The financial benefits provided to the Sri Lankan economy by healthy coral reefs, specifically through tourism, fisheries and coastal protection, are significant (Berg et al., 1998). Growth in the tourism sector, which before the COVID-19 pandemic represented 4.9% of the national Gross Domestic Product (GDP) and contributed almost US$4.4B in foreign exchange earnings (SLTDA, 2018), is partly linked to successful advertising of Sri Lanka's coral reefs.

Management practices and local Sri Lankan laws intended to preserve and protect coral ecosystems are described as adequate but also as being largely unenforced (Rajasuriya and White, 1995; Weerakoon et al., 2018). Presently, however, multiple agencies oversee environmental protection. Management of the natural environment is the responsibility of the Ministry of Environment and Renewable Energy, with marine pollution matters expressly handled by the Marine Environmental Protection Agency (MEPA). The conservation of the aquatic environment is the responsibility of the Ministry of Fisheries and Aquatic Resources. Marine protected areas declared under the Fauna and Flora Protection Act (1972, 1993) meanwhile, are managed by the Department of Wildlife Conservation, which sits within the Ministry of Wildlife and Forestry Conservation.

The first marine protected area targeting coral reef protection was declared at Hikkaduwa in 1979. Since then, several further areas have been declared as marine protected areas. All MPAs are declared under the Fauna and Flora Protection Ordinance (FFPO) administered by the Department of Wildlife Conservation. The three MPA categories are: Marine National Park (MNP), Nature Reserve (NR) and Sanctuary. The MNP has the highest level of protection, and the sanctuary has the lowest level of protection under the FFPO. Although several protected areas have been declared only three of them have management plans, and relevant details are given in Table 5.

The Coastal Zone Management Plan (CZMP) addresses mainly the development, planning, and permitting systems for state and private development activities. The CZMP is reviewed and updated periodically, and the current plan is the CZMP of 2018. Adoption of the CZMP in 2004 strengthened many aspects of coastal management including the introduction of Special Coastal Management Areas (SCMA), which included community involvement in the comprehensive management of marine natural resources (Rajarathna and Nianthi, 2019). Co-management of the marine environment had been attempted previously in some coastal areas by the Department of Fisheries and Aquatic Resources, with limited success due to a lack of inclusiveness in engaging fishers in the planning and management of fisheries. The Marine Environment Protection Authority (MEPA) is the mandated government organization addressing pollution in the coastal and marine environment. However, water quality monitoring in coastal and marine waters lies primarily with both the National Aquatic Resources Research and Development Agency (NARA) and MEPA. Furthermore, while universities have conducted water quality studies as part of various projects, there is no centralized collation of that data due to the lack of a national database for water quality in the coastal and marine environment. Coordination between the various agencies has also proven challenging, with historic data scattered in hard-to-find reports and publications. Many data gaps prevent appropriate management decision-making and implementation of actions, particularly concerning pollution from agrochemicals and fertilizer use.

As a result of the complex management arena, the complications in aggregating historic measurements, and the apparent sparsity of appropriate observations it is presently difficult to synthesize a credible impact statement on the role that nitrogen pollution has played in the long-term degradation of coral habitats in Sri Lanka.

3.3 Maldives

3.3.1 Coral reef extent, condition and environmental setting

The Republic of the Maldives is a low-lying archipelago consisting of 1,192 islands within 26 natural atolls that stretch from 0.5°S to 7°N in the central Indian Ocean, (Figure 4). The total land area is approximately 227 km^2 with most islands lying at an average height of 1 m above sea level (MEE, 2015b). Many islands are less than 0.5 km^2 in area and are covered with nutrient-poor, carbonate sand saline soils that are highly porous and prone to leaching (Herzog and Konrad, 1992; UNEP, 2005). Coral reef coverage varies significantly between studies with estimates ranging from 14,533 km^2 (Bahuguna et al., 2013), 8920 km^2 (Spalding et al., 2001), 6372 km^2 (Souter et al., 2021), to ~4,500 km^2 (Naseer and Hatcher, 2004). Tropical conditions have contributed towards the growth of the Maldivian atoll reef system which is the seventh largest system in the world (Spalding et al., 2001). The reef comprises a combination of oceanic faros, oceanic platform reefs, patch reefs, and fringing reefs, all as part of the atoll system (Naseer, 1997), that support over 250 species of hermatypic corals (Pichon and Benzoni, 2007), 36 species of sponges, 285 species of algae, and over a 1000 species of fish (MEE, 2015b; MEE, 2017a).

The country experiences two monsoon seasons (southwest and northeast), has an average annual temperature of 30.7 °C and receives 2,124 mm of rainfall. The influence of the monsoon winds on local ocean circulation is significant and induces an appreciable island mass effect around the Maldives enhancing surface chlorophyll concentrations (Radice et al., 2019; Su et al., 2021). A recent numerical study of the island mass effect around the

TABLE 5 Management status of Sri Lanka Marine Protected Area's.

Name	Year of declaration	Gazette no	Extent (ha)	Remarks	Management plan
Adam's Bridge Marine National Park	2015	1920/3 of 22.06.2015	18,900	100% marine area	No
Pigeon Island Marine National Park	2003	1291/16 of 04.06.2003	471.43	100% marine	Yes
Hikkaduwa Marine National Park	2002	1257/14 of 08.10.2002	250	100% marine	Yes (Special Area Management Plan under SAMP)
Delft Island National Park	2015	1920/3 of 22.06.2015	1,846	100 m wide belt in the sea. Rest is land	No
Chundikulam National Park	2015	1920/3 of 22.06.2015	19,565.33	43% marine area in coastal waters	No
Vidattaltivu Nature Reserve	2016	1956/13 of 01.03. 2016	29,180	77% marine area	No
Nagarkovil Nature Reserve	2016	1956/13 of 01.03. 2016	7882	66% marine area in coastal waters	No
Nayaru Nature Reserve	2017	2003/10 of 24.01.2017	4464.35	25% lagoon. (This PA has a narrow strip of the coastal waters)	No
Bar Reef Sanctuary	1992	708/24 of 03.04.1992	30,670	100% marine	Yes (Special Are Management Plan under SAMP)
Vankalai Sanctuary	2008	No. 1566/3 08.09.2008	4838.95	62% marine	No
Paraitivu Parititivu Island Sanctuary	1973	No. 60 of 18.05.1973	97.1	100% marine	No
Kayankerni Sanctuary	2019	2118/59 of 11.04.2019	953.25	99% marine	No
Great and Little Sober Islands Sanctuary	1963	21.06.1963	64.7	100% marine	No
Rekawa Sanctuary	2006	25.05.2006	271	83% marine (includes 500 m belt of sea)	No
Godawaya Sanctuary	2006	25.05.2006	230.99	83% marine (includes 500 m wide belt of sea)	No
Kalametiya Sanctuary	1984	28.06.1984	2,525.2	13% marine	No
Great and Little Basses Sanctuary	2019	2144/60 11.10.2019	67,282.3	100% marine	No
Rumassala Sanctuary	2003	03.01.2003	170.7	90% marine	No

Maldives found both upwelling and downwelling favorable conditions during both monsoon seasons, but it was discovered that the seasonal development of downstream island wakes was limited to atolls above 4°N, with atolls further south being more heavily influenced by the zonal movement of equatorial currents (Su et al., 2021).

Existing nutrient measurements from the oceanic waters around the Maldives are sparse. For the region from -0.26 to 7.65°N, 69.5 to 77.5°E, WOD2018 indicates 22 observations of nitrate and 102 observations of phosphate. Near-surface (5 m) nitrate concentrations range from 0.27 – 2.6 µmol L^{-1} (mean ± stdev 0.46 ± 0.6 µmol L^{-1}) with phosphate concentrations ranging from 0.01 – 0.8 µmol L^{-1} (mean ± stdev 0.25 ± 0.14 µmol L^{-1}). Surface waters of the surrounding tropical ocean are nutrient-poor and naturally depleted in N relative to P (mean N:P of 2.39 ± 1.58) compared to the Redfield ratio, a measure of both average deep water nutrient ratios and mean phytoplankton composition (Redfield, 1958; Redfield et al., 1963; Tyrrell and Laws, 1997; Tyrrell, 1999). The range in observed nutrient concentrations is large compared to the mean annual climatological conditions derived from the same dataset for the Maldives archipelago (~0.15-0.2 µmol NO_3^- L^{-1}; ~0.2 µmol PO_4^{3-} L^{-1}) (Garcia et al., 2019).

Coral bleaching events have devastated the Maldivian reef system over the last 20 years, and the 1998 global bleaching event alone killed >98% of branching corals species locally and reduced average live coral cover from >40% to close to 0% in some places (Edwards et al., 2001; Loch et al., 2002; Zahir, 2002; Pisapia et al., 2016). Much of the structure and coral diversity lost during the 1998 bleaching event was rediscovered in subsequent years, though recovery was not consistent across the archipelago (Zahir, 2002;

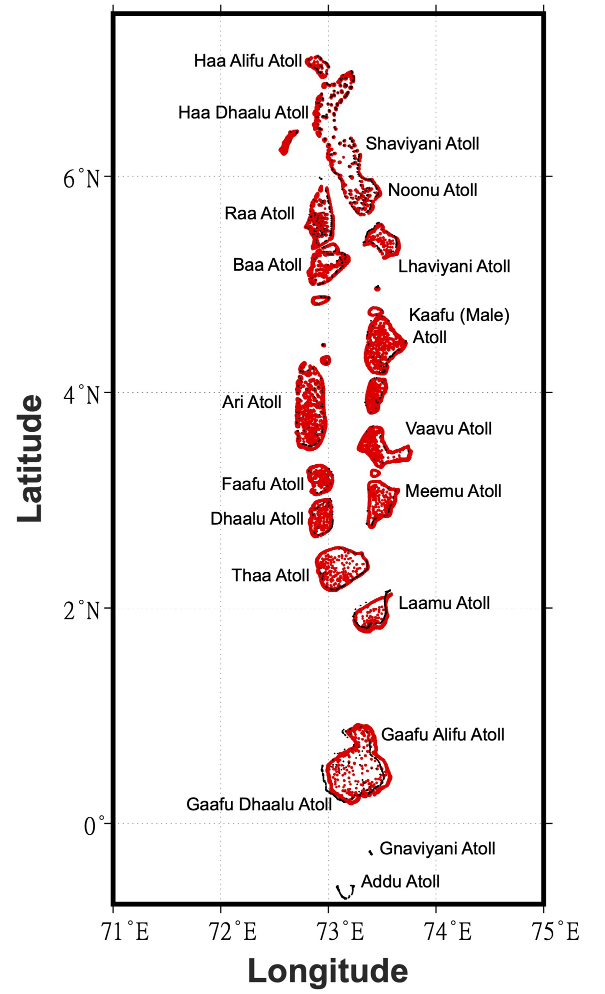

FIGURE 4
Map showing the distribution of coral reefs in Maldivian waters. Coral distribution based on data compiled by UNEP-WCMC et al. (2018).

Zahir et al., 2002; Lasagna et al., 2008; Morri et al., 2010; Morri et al., 2015). The 2009 survey of Maldivian National Coral Reef Monitoring (NCRM) sites, conducted a decade after the 1998 global bleaching event, indicated mean coral coverage varied between 7.5% and 59.4% (Zahir et al., 2010). However, bleaching again in 2016 resulted in live coral cover dropping close to or below 20% at studied sites, with associated mortality of ~80% (Ibrahim et al., 2017). Recent surveys by the Maldives Marine Research Institute (MMRI) at the NCRM sites indicate current live coral coverage continues to persist at these reduced levels with dead coral cover exceeding 70% in many areas, though a number of recruits were observed at many sites suggesting reefs have a capacity for resilience and recovery. This is supported by a recent nationwide assessment that reported recruitment densities averaging 14.3 individuals per m^2, densities that exceed other similar reef systems in the region (Noo Raajje, 2021). Additionally, several other notable events occurred between 1998 and 2016 that affected the Maldivian reefs although their impacts were minimal in comparison to these two major bleaching events. These other events include the 2004 tsunami (Government of Australia and Marine Research Centre, 2005), localized bleaching events in 2003, 2007 and 2010, and outbreaks of the corallivorous crown-of-thorns starfish *Acanthaster planci* across Ari Atoll (Saponari et al., 2014; Pisapia et al., 2016; Saponari et al., 2018), though the extent and severity of *A. planci* infestations across the wider Maldivian atolls remains unclear.

3.3.2 Water quality indicators

There is little information available about the water quality in Maldivian waters. As ocean atolls experience considerable advective flushing there has long been an assumption that land-derived nutrient inputs have only minimal impact being either rapidly advected away from the atolls or perhaps being of smaller magnitude than any upwelling resulting from the island mass effect (e.g. (Szmant, 2002; Radice et al., 2019; Su et al., 2021)). For example, the 2011 State of the Environment Report issued by the Government of the Maldives (MEE, 2011) concluded that nutrient over-enrichment was not a significant problem for corals, though the evidence base was acknowledged as being slight. Subsequent studies by Jaleel (2013) and Radice et al. (2019) either reiterated or found little or no evidence for land-derived nutrients impacting coral reefs. Nevertheless, the 2016 State of the Environment Report (MEE, 2017a) concluded that land-derived nitrogen and phosphorous inputs may pose a more serious threat to Maldivian coral ecosystems than previously realized. Increased nutrient fluxes from expanding agricultural activities and from larger sewage and food waste discharges were highlighted as causes for concern. However, while fertilizer imports into the Maldives more than doubled between 2010 and 2015, corresponding with increased fertilizer usage across the country (MEE, 2015a), evidence for an impact on the marine environment remains limited. Key examples cited as potentially illustrating the impacts of nutrient inputs include i) increased seagrass coverage around agricultural islands (Laamu Atoll) (Miller and Sluka, 1999; BOBLME, 2010; MEE, 2017a), ii) the occurrence of "red tide" algal blooms, iii) several massive fish kills, and iv) the recognition that farmers may not be using fertilizers in accordance with manufacturer guidelines, leading to overuse and/or increased leaching (MEE, 2017a). In a specific example, coral survey results reported by Ibrahim (2019) from Huvadhu Atoll demonstrated that the benthic community structure of reef flat areas was negatively impacted by proximity to inhabited islands, with nutrient input derived from tuna fish processing, sewage and sediment input from development activities implicated as drivers of change.

Despite growing concerns about nutrient pollution impacts, a broad-scale isotopic survey of corals across the central Maldivian atolls demonstrated that deep ocean nitrate remains the dominant source of nitrogen for corals, supplemented with particulate organic matter (POM) filtered from seawater (Radice et al., 2019). Based on this approach, the impacts from land-derived nutrients are either spatially limited and not yet fully quantified or entirely minimized by strong oceanic flushing through the atolls by the monsoon winds. The novel use of benthic foraminifera to assess water quality and coral health across North Ari Atoll indicates that water quality generally remains suitable for healthy coral development, yet around inhabited islands detectable changes in the benthic foraminiferal assemblage indicate the onset of water quality degradation, though not yet sufficient to inhibit coral development (Pisapia et al., 2017). Miller and Sluka (1999) meanwhile, suggested that nutrient leaching may be driving long-term growth in sea grass coverage around agricultural atolls. While changes in seagrass coverage remain a potentially useful indicator of nutrient seascape changes, there are important subtleties between the impacts of moderate and excess nutrient enrichment that should be considered in addition to changes in seagrass spatial coverage (Mazarrasa et al., 2018). In particular, a comparable study from southern Indian waters concluded that (excessive) nutrient enrichment was likely driving a reduction in seagrass coverage (Purvaja et al., 2018).

More traditional indicators of water quality and nutrient enrichment, such as increased phytoplankton productivity or biomass, are difficult to distinguish as anthropogenic or natural in origin due to the island mass effect. A recent analysis of satellite surface chlorophyll concentrations revealed the substantial impact monsoon-driven upwelling has on the spatial and temporal distributions of chlorophyll concentrations around the Maldives (Radice et al., 2019). Whilst more intensive analyses related to the phytoplankton community, particularly concerning sporadic accounts of red tides or other harmful blooms (MEE, 2017a), may become an important future monitoring activity, not least because of the public health perspective (e.g. (Tamele et al., 2019)), they are not yet undertaken within the context of coral monitoring or water quality surveys. Furthermore, increased occurrences of red tides and fish kills across the wider Indian Ocean have been reported in recent years, though the cause is often uncertain (Amla and Uranie, 2014; Uranie and Vannier, 2014; Uranie, 2015; Uranie and Meriton-Jean, 2015), and a wider shift in phytoplankton community structure is increasingly evident across the much of the western and central Indian Ocean (Goes et al., 2020). An investigation into multiple fish kills within the Maldives concluded that only one such event could be linked to an algal bloom, though limitations in infrastructure and research capacity prohibited further investigation while alternative fish kill events lacked a clear connection to algal blooms (Naeem and Sattar, 2007).

3.3.3 Reported impacts of nutrient pollution

Studies on nitrogen pollution impacts are currently very limited for the Maldives. Regular national reports on bleaching (e.g. (Zahir et al., 2010; Ibrahim et al., 2017) acknowledge that pollution impacts, mainly from sewage and wastewater, are likely to be problematic, but no quantitative investigations have yet been undertaken. Long-term monitoring of Maldivian coral reefs implemented following significant bleaching in 1998 and now under the auspices of the MMRI, does not yet include monitoring of water quality specifically for nitrogen loadings, and the significance of terrestrial nitrogen transfer remains unclear.

Furthermore, whilst Environmental Impact Assessments assess and evaluate coral reefs near project developments (commonly harbor development and land reclamation projects) and numerous parties investigate coral reefs in various capacities, these efforts often capture similar aspects of known major reef stressors and focus on coral reef status and health only. The disparity between what is recorded and what needs to be recorded is pressing, not only because chemical fertilizer management has been identified as poor (MEE, 2017a), but also because the country has a growing agricultural sector dependent on increasing usage of various combinations of organic and inorganic fertilizers (Mohamed, 2018; FAO, 2020). In addition, the need to improve sewage facilities is widely recognized (MEE, 2017b), particularly where there are known instances of point source pollution from poor, insufficient, or failing sewage systems with the capital city of Male' being a notable example (FAO, 2011). As the impacts of nutrient over-enrichment in the Maldives archipelago may become apparent at relatively minor levels of enrichment compared to ambient nutrient concentrations (~10%; as derived from the results of Guan et al. (2020)), proper management measures and enforcement of those measures are crucial.

Though the available evidence is limited, there are indications that proximity to human populations and the effectiveness of associated management regimes do significantly affect coral reefs in the Maldives. For example, Heikoop et al. (2000) investigated the difference in nitrogen content in corals between the sewage-affected Male' reef and reference corals in Addu Atoll in the far south and found that nitrogen content, linked to nitrogen loading from untreated sewage, was significantly higher in the sewage-affected reef samples. Such results therefore differ from the findings reported by Radice et al. (2019). Separately, dead coral cover and filamentous algae cover were found to be higher close to areas with denser human populations (Brown et al., 2017), with human population density also shown to impact coral community composition during post-bleaching recovery periods (McClanahan and Muthiga, 2014). Both the population of the Maldives and the tourism sector have continued to grow in recent decades (Montefalcone et al., 2020), leading to increased anthropogenic pressures on the marine environment and increasing the demand for agricultural produce. Oil pollution stemming from the maritime sector is also problematic, largely due to the lack of appropriate disposal options (MEE, 2017a), and the high reliance on maritime craft for movement between atolls.

Nutrients, chemicals, and sediments released from tourist resort activities generally have a negative impact on resort reefs, even though resort or house reefs are considered a pseudo- marine managed area within the country. The overall negative impacts may still therefore outweigh the benefits of protection (Cowburn et al., 2018), though this is far from clear for all reef-associated organisms (Moritz et al., 2017), while the severity of anthropogenic impacts may also differ with depth (Nepote et al., 2016).

Coral diseases have been little studied in the Maldives but the general prevalence rate is estimated to be around 1.5% (Montano et al., 2016). Spatial variability in disease prevalence throughout the central region of the Maldives may potentially be linked to human population densities though regions with low coral densities and/or larger colonies were found to be more susceptible to infection than elsewhere (Montano et al., 2016). The incidence of disease amongst northern and southern atolls has not yet been sufficiently well investigated to allow assessment of disease prevalence.

3.3.4 Current protections and safeguards

Environmental degradation resulting from coastal development is recognized as a major anthropogenic threat to coastal ecosystems in the Maldives (Nepote et al., 2016; Stevens and Froman, 2019). Islands within the central Ari Atoll have undergone significant development, including the construction of an airport, which increased sedimentation on the surrounding reefs (Nepote et al., 2016). While coastal development is generally detrimental to coral reef management, it has responded to and created growth in the tourism sector, which benefits the Maldives economy as a whole.

Aside from the effectiveness of legislative and regulatory mechanisms, community attitudes can sometimes be a challenge in addressing marine threats (Nickerson and Maniku, 1997). There is a distinct lack of opportunities in terms of income, growth, and development within islands which forces people to pursue a limited range of professions and which hinders the willingness, capacity, and capability to address marine threats (Mancini et al., 2017). This is exacerbated by the fact that, while several Marine Protected Areas (MPA) exist within the country, capacity and resource constraints mean that many MPAs lack effective management mechanisms. However, these limitations can be addressed through an improved understanding of the relationship between local human populations and coral reef areas and reef conditions across the country (Agardy et al., 2017), which can enable the effective utilization of locally managed marine areas.

Successful management of the marine environment is central to the tourism sector, and regulations exist to manage the development of tourism successfully alongside the desire to protect the coral reef environment (MEE, 2017a). Nevertheless, it has been shown that reef conditions generally improve with distance from the capital, Malè, and from other community islands, whereas reef conditions can deteriorate with resort age (Price and Firaq, 1996). There is high public awareness of the need to conserve the marine environment and reef ecosystems, which is said to have resulted in considerate waste disposal by many resorts (MEE, 2017a).

4 The challenge of nutrient enrichment

Global stressors (e.g. ocean temperature rise, pH decline) and local stressors (e.g. fishing, pollution) both contribute to the degradation of coral habitats. While both type of stressors are ultimately anthropogenic in origin, local stressors are more important in the short-term because they contribute more to coral reef loss and habitat decline. Land-based pollution is already recognized as the top issue for marine ecosystem management both globally and for South Asian countries in particular, with sewage discharge, agricultural runoff, and industrial effluents recognized as the major pollution sources (Qasim et al., 1988; GESAMP, 2001; Kaly, 2004). Globally, 20-25% of anthropogenic nitrogen inputs to coastal watersheds are exported to coastal ecosystems primarily *via* rivers and atmospheric transportation pathways (Malone and Newton, 2020), though in some settings submarine groundwater discharge may also be a significant additional pathway (Santos et al., 2021; Selvam et al., 2022; Sajeev et al., 2023). Across South Asia, there is wide variation in the sources, nutrient loadings, and ability to monitor for marine pollution impacts (BOBLME, 2010; BOBLME, 2011; BOBLME, 2013; BOBLME, 2015). Sewage is considered the most significant source of marine pollution in the Maldives, while effluents from agriculture, aquaculture, and sewage are the most problematic in Sri Lanka and industrial effluents and sewage are the most significant in India (BOBLME, 2011; BOBLME, 2015). With few exceptions however, there is generally insufficient publicly available data to adequately assess the magnitude, severity, and trend of nutrient pollution impacts on coastal waters let alone on regional coral ecosystems. In this regard research focused on South Asian corals is not unique for there is arguably a general lack of research globally on the topic of nutrient enrichment and coral health (Li et al., 2022). Yet, wide variations in reported nutrient concentrations between studies, as well as the scope and duration of water quality monitoring programmes for the coastal zone in general, impede regional efforts to understand and quantify such impacts (BOBLME, 2015). Consequently, pollution impacts on corals have generally received less attention than the impacts from fishing or other (visually) destructive activities.

In the broader discussion, however, pollution (of all origins) is already recognized as a leading factor inhibiting coral recovery following bleaching events (Wilkinson, 2008; Wilkinson et al., 2016) and for the South Asian region marine pollution in conjunction with increased sedimentation due to land-use change and in some areas unregulated or destructive exploitation of marine resources (fishing) has been suggested as the major anthropogenic threat to the region's corals (Wilhelmsson, 2002). Indeed, differences in pollution intensity across the region were previously suggested as a factor obscuring a clear and consistent regional trend in coral reef recovery following widespread coral bleaching (Tamelander and Rajasuriya, 2008). Yet despite this awareness quantitative studies linking nutrient pollution to the degradation of coral habitats across South Asia remain limited (Chandra et al., 2021), which raises the question - is it possible to infer the scale of the potential problem in the absence of sufficient experimental data to quantitatively demonstrate this impact?

4.1 Fertilizer inputs

Regional average fertilizer consumption across South Asia rose by over 7000% from ~2.4 kg ha^{-1} yr^{-1} in 1961 to ~170.1 kg ha^{-1} yr^{-1} by 2018 (World Bank, 2021). Fertilizer consumption across South Asia is 24% higher than the global average rate of 137 kg ha^{-1} yr^{-1} of arable land (World Bank, 2021). Within the region, average national consumption rates for all fertilizer types vary significantly, and for 2018 were 175 kg ha^{-1}, 138 kg ha^{-1} and 115 kg ha^{-1} for India, Sri Lanka and the Maldives respectively (Figure 5; note that the total fertilizer consumption rate includes nitrogen, phosphate and potash-based fertilizers). Consumption rates in Sri Lanka have historically been higher than elsewhere in South Asia and peaked at over 300 kg ha^{-1} in 2008. Due to growing concerns about soil health (FAO and ITPS, 2015), ground-water contamination (Imbulana et al., 2006; Prabagar et al., 2020) and

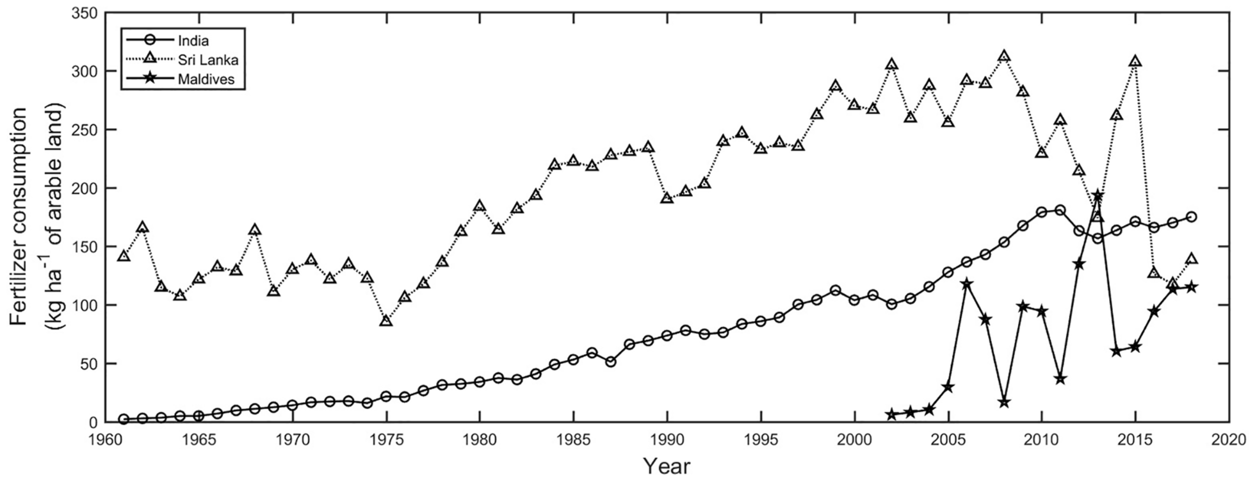

FIGURE 5
Annual fertilizer consumption rates for India, Sri Lanka and the Maldives for the period 1960-2018. Note that the estimates of fertilizer consumption are based on products that include nitrogen, phosphate and potash-based fertilizers. Total fertilizer-N consumption is thus lower than indicated here. Data from World Bank database (https://data.worldbank.org/indicator/AG.CON.FERT.ZS).

negative impacts on human health, which includes high rates of chronic kidney disease (e.g. (Abraham et al., 2016; Kafle et al., 2019), fertilizer consumption rates in Sri Lanka have rapidly declined in recent years due to Government initiatives to control fertilizer pollution even extending to a short-lived ban on the use of chemical fertilizers imposed by the Sri Lankan Government in 2021. Meanwhile, consumption rates in India continue to rise, increasing by 5-6% per year (Sutton et al., 2017), while reported consumption rates in the Maldives have increased by nearly 2000% since 2002.

Despite significant recent efforts to quantify nitrogen fluxes across South Asia (e.g. (Kaly, 2004; BOBLME, 2010; BOBLME, 2014; BOBLME, 2015)), and particularly within India *via* national nitrogen assessments (Abrol et al., 2017), much remains unknown about the fate and impact of anthropogenic nitrogen on coastal environments. For example, the Indian Nitrogen Assessment (Abrol et al., 2017), highlighted both an 11-fold increase in fertilizer N consumption within India between 1970 and 2011 and a 4-fold increase in the loss of fixed nitrogen to the environment (Abrol and Adhya, 2017) but concluded that the evidence to document an impact on the marine environment was currently patchy and incomplete (Prema et al., 2017; Ramesh et al., 2017). Consumption of fertilizer nitrogen in India now exceeds 18.8 Mt N per year (as of 2019-2020; (The Fertilizer Association of India, 2022)), with over 80% of this fertilizer nitrogen represented by urea-based products (Tewatia and Chandra, 2017). Detailed studies suggest an average consumption rate of ~89 kg N ha^{-1} yr^{-1}, but strong regional variability can see this range from 5 to >200 kg N ha^{-1} yr^{-1} at the district level (Tewatia and Chandra, 2017). In contrast, total nitrogen inputs to agricultural lands, which in addition to synthetic fertilizer application also considers biological nitrogen fixation, atmospheric deposition, and organic manure application, are higher at around 135 kg N ha^{-1} yr^{-1} (Bijay-Singh, 2017). Estimates indicate that 65-68% of applied N (equivalent to around 93 kg of total N applied ha^{-1} yr^{-1}) is subsequently lost to the environment (Abrol and Adhya, 2017; Bijay-Singh, 2017), with a significant proportion entering ground and surface water systems with the potential for export to coastal waters (Prakasa Rao et al., 2017). Yet, despite improved efforts to quantify and track fertilizer nitrogen application and loss across agricultural lands and through the land-ocean continuum, the lack of comprehensive monitoring of coastal waters severely limits efforts to quantify the export of applied nitrogen to the ocean (Kaly, 2004; BOBLME, 2010; BOBLME, 2014; BOBLME, 2015; Abrol and Adhya, 2017). Prema et al. (2017) estimated that India's rivers export 1.6-1.8 Tg N yr^{-1}, an amount equivalent to around 9% of applied fertilizer nitrogen. On this basis, they argued that the remaining 91% of nitrogen fertilizer was converted to crops, lost to the atmosphere, or recycled and retained within rivers and estuaries. This exported nitrogen remains significant, however, and was estimated to support ~13% of coastal new production (Prema et al., 2017). Fertilizer application and coastal discharge are not, however, uniform with 73% of fertilizer N applied to catchments that drain to the east coast (Bay of Bengal), and the remaining 27% draining to the west coast (Arabian Sea) (Swaney et al., 2015; Ramesh et al., 2017). This may signify limited potential for impact on coral ecosystems due to the broad absence of coral reefs in the Bay of Bengal (e.g. Figure 1). Yet, when fertilizer consumption rates are normalized to river catchment area it appears that west coast rivers export 4-times more nitrogen per unit area than east coast rivers, particularly during the monsoon months (west coast: 607 ± 700 kg N km^{-2} yr^{-1} vs east coast:161 ± 369 kg N km^{-2} yr^{-1}; (Krishna et al., 2016; Ramesh et al., 2017). More detailed studies of fertilizer consumption show that >50% of fertilizer consumption within India occurs within just five states, (Uttar Pradesh, Maharashtra, Punjab, Gujarat, Madhya Pradesh), two of which have an Arabian Sea coastline (Bhattacharya et al., 2017).

While efforts to quantify riverine dissolved inorganic nitrogen fluxes to the coastal ocean and the contribution attributed to fertilizer application are improving, there is still a chance that a significant proportion of applied nitrogen that is later relevant to coral habitats is being overlooked in export estimates. Ramesh et al.

(2017) noted that the ratio of inorganic to organic nitrogen exported by Indian rivers strongly favored organic forms with typically around 10% of riverine nitrogen exported as inorganic nitrogen and 90% as organic nitrogen compounds. Given India's reliance on organic nitrogen-based fertilizers (Tewatia and Chandra, 2017), and the fact that the coral holobiont is sensitive to both inorganic and organic nutrient concentrations (e.g. (Rädecker et al., 2015), it appears critical that all nitrogen forms be captured in coastal water quality monitoring activities.

Comparable information on fertilizer inputs to the coastal regions of Sri Lanka or the Maldives is harder to find and currently less well understood than for India. The historically high fertilizer consumption rates for Sri Lanka and the increasing consumption rates for the Maldives (Figure 5), suggest high or increased inputs of fixed N to the coastal regions but as with India, the observational data needed to document impacts on the marine environment is limited (Manage et al., 2022).

4.2 Wastewater inputs

Nutrient losses from agriculture and wastewater discharge from households and industry represent the two most important sources of nutrient input to freshwater systems (Bouwman et al., 2005). By extension, they are also critical to assessing nutrient pollution inputs to coastal seas though a considerable reduction in aquatic nutrient loadings, by some estimates ~70% (Billen et al., 2013), occurs along the land-ocean continuum ultimately limiting the impact of nutrient inputs to freshwater systems on the coastal zone. Global raw sewage production is currently estimated to be around 19 Tg N yr^{-1}, with wastewater treatment reducing net sewage inputs to ~6 Tg N yr^{-1} (Billen et al., 2013). By 2050, it is estimated that the nitrogen input to surface waters from sewage could reach 12-18 Tg N yr^{-1} (Van Drecht et al., 2009; van Puijenbroek et al., 2019) representing a significant and growing environmental perturbation.

The availability and accessibility of data relating to sewage generation, wastewater treatment, and discharge vary by country and are not always readily accessible. Summary statistics on access to different sanitation services from the WHO/UNICEF Joint Management Programme (WHO/UNICEF, 2021) are presented in Table 6 to illustrate the regional variability in access to different sanitation services. The data show, for example, that access to sewer connections by urban populations ranges from ~9% (Sri Lanka) to 100% (Maldives), or that at the national level use of latrines dominates over septic tanks in Sri Lanka (94.3% vs 1.5%), whilst in India use of latrines and septic tanks is more equal (34.6% vs 36.2%).

For India, as of June 2021, domestic sewage generation by urban populations was reportedly 72,368 million liters per day (MLD; (CPCB, 2021)). Total sewage treatment capacity is less than the generation rate and can handle only 31,841 MLD, so on average only 44% of India's daily urban sewage generation is treatable. Operational treatment capacity however is lower at 20,236 MLD, indicating that on average only 28% of daily urban sewage generation is treated. The remaining 72% of daily urban sewage

TABLE 6 Estimates of access to sanitation services as a proportion of the population.

Country	Total Population (%)	Urban Population (%)	Rural Population (%)	Sanitation Service
Sri Lanka	94.3	84	96.7	Latrine
	1.5	3.5	1.1	Septic Tank
	1.8	9.1	0.1	Sewer Connection
	0	0	0	No Service
	–	–	–	Safely managed[1]
India	34.6	13	46.2	Latrine
	36.2	51.4	28	Septic Tank
	12.6	34.2	1	Sewer Connection
	15	1	22	No Service
	46	37	51	Safely managed[1]
Maldives	1.2	0	2	Latrine
	32.5	0	54.8	Septic Tank
	65.8	100	42.3	Sewer Connection
	0	0	0	No Service
	–	–	–	Safely managed[1]

[1] Estimate not available, see WHO/UNICEF Joint Management Programme for further details.
Data from WHO/UNICEF Joint Monitoring Programme (JMP) for water, sanitation and hygiene database (WHO/UNICEF, 2021). Safely managed sanitation services are defined as the use of an improved sanitation facility that is not shared with other households and where excreta are disposed of *in situ* or transported and treated offsite.

generation (~52,105 MLD) is released untreated. Based on the latest national survey results approximately 36% of treatment capacity is under-utilized or non-operational (CPCB, 2021), but even if treatment facilities were operating at full capacity less than half of the daily urban sewage generation can be treated. Sewage generation and treatment capacity also vary significantly at the state level (CPCB, 2021). Summary information for the offshore islands and west coast states, which are contiguous with major coral localities, is presented in Table 7. At best, a maximum of 54% of daily sewage generation can be treated (Gujarat). At worst, there is zero capacity to treat sewage (Andaman and Nicobar, Lakshadweep Islands), with septic tanks or direct discharge to the ocean the only option. Provisionally, around one-third of the daily sewage generation from the island and (west) coastal states is treated, with the remainder discharged untreated.

The current National Water Assessment for the Maldives (UNESCAP, 2021), indicates that sanitation services in the form of sewage treatment plants or septic tanks are available for 48% of the population covering 66 out of 187 inhabited islands. There are, however, many unknowns with regards to sewer outfall operations due to limited data availability, which makes it harder to assess any subsequent environmental impact. For example, flow data from various industries and households to the sewerage system (i.e. inputs) is insufficient and, in some cases, not measured, whereas total discharge from the sewerage system to the environment (i.e. outputs) is unquantified or unavailable (UNESCAP, 2021). The National Water Assessment highlights that discharges to the environment, which include island groundwater aquifers and the ocean, include brine from desalination processes, generator cooling waters, waste seawater, treated wastewater (from septic tanks or sewage treatment plants), and minimally or untreated wastewater from the sewerage system. The sewerage system's treated and untreated wastewater is preferentially discharged to the ocean. Given current data availability, it is not possible to obtain an order of magnitude estimate of wastewater input to the ocean for the Maldives, but the presumption is that this is growing due to population increases and resort development and expansion. The uncertainty in discharge volumes is already a noted concern (UNESCAP, 2021), but it is also clear from the National Water Assessment that current wastewater discharges include significant thermal, saline, faecal, and chemical loadings with the potential for wide-ranging negative impacts on coral ecosystems.

The situation in Sri Lanka is more complex as data on wastewater production is neither readily available nor compiled by a single agency. Wastewater regulations stipulate that wastewater must undergo secondary level treatment and conform to strict water quality standards before discharge, particularly as treated wastewater is widely returned to inland surface waters and internal irrigation networks for subsequent reuse within the agricultural sector (Jayalal and Niroshani, 2012). Data from the WHO/UNICEF Joint Monitoring Programme (Table 6) indicates that only 9% of urban populations are connected to a sewerage network. Around Colombo, however, the combination of an aging sewer network unsuitable for modern demands and the cost of treatment means that sewage is discharged without even preliminary level treatment *via* two 1.5 km long offshore outfalls with the potential for subsequent impact on the marine environment (Mysan and Ranasinghe, 2014).

4.3 But what is the true impact of nitrogen pollution on South Asian coral ecosystems?

In examining the role of reactive nitrogen pollution in Indian coastal waters Prema et al. (2017) noted that whilst there are many generalized statements about pollution impacts it remains rather difficult to find quantitative evidence or even qualitative observations to support such statements in practice. Similar concerns have been raised elsewhere and stem from significant regional differences in water quality or habitat monitoring activities (BOBLME, 2015; Abrol and Adhya, 2017; Ramesh et al., 2017). Collectively, there is recognition that the South Asian region is at increased risk of pollution-induced eutrophication and that some areas may already be experiencing such impacts, but also that quantitative evidence to determine, track, and inform on these developments remains limited (BOBLME, 2015; Abrol and Adhya,

TABLE 7 Summary estimates of urban sewage generation and treatment in India (CPCB, 2021). MLD = million litres per day.

State	Total sewage generation (MLD)	Treatment Capacity (MLD)	Utilised Capacity (MLD)	% treated
A & N Islands	23	0	0	0
Lakshadweep Islands	13	0	0	0
Gujarat	5013	3378	2687	54
Maharashtra	9107	6890	4242	47
Goa	176	66	25	14
Karnataka	4458	2712	1786	40
Kerala	4256	120	47	1
Tamil Nadu	6421	1492	995	15
Total	29,467	14,658	9782	33

2017; Ramesh et al., 2017). In reviewing the existing literature for known nitrogen pollution impacts on coral ecosystems in the South Asia region it has become clear that there is significant variation in the extent of research between countries and consequently at the regional level there is inadequate information with which to diagnose impacts, trends, or confidently attribute specific changes to nutrient inputs (Table 8). In most cases no evidence exists to definitively link nutrient inputs to known impacts.

There remains considerable difficulty in translating national fertilizer consumption rates or sewage trends into a nitrogen flux to the coastal environment, and thereafter to a specific habitat such as corals. The difficulty in assessing the impacts of nutrient pollution on the marine environment is also compounded by the complexities of identifying coastal transport pathways, national and transboundary pollution sources, managing rapidly growing human populations, enforcing existing legislation (Edeson, 2004), the financial cost of initiating and maintaining appropriate monitoring programmes, the wide range of pollution impacts on corals (Table 1), and the far more visible impacts from fishing or bleaching which garner greater attention. While the effects of bleaching have been better studied in South Asia in recent decades, more focused studies on pollution impacts have rarely been included. Generally, coral reef monitoring has not been undertaken with nutrient pollution in mind. Consequently, general statements regarding the impact of nutrient pollution are widespread but are rarely supported with definitive experimental observations, or even with qualitative observations covering a sufficient temporal period to support the validity of such statements (Bhatt et al., 2012a). The situation is however changing, driven in part by recognition that nutrient enrichment of coastal waters is a global and growing problem (NRC, 2000; Selman et al., 2008; Rabalais et al., 2009; Selman and Greenhalgh, 2009a; Selman and Greenhalgh 2009b; BOBLME, 2015; Malone and Newton, 2020), but also from greater understanding and concern about the consequences of high faecal coliform loads for human health and fisheries (Shuval, 2003; Islam and Tanaka, 2004). Even so, quantitative information on nutrient inputs, and movement within the coastal zone is patchy, and while efforts to evaluate the magnitude of national nutrient inputs to the coastal zone are growing (Mayorga et al., 2010; Seitzinger et al., 2010), it remains difficult to assess the environmental and ecological consequences of those inputs, particularly for the regions corals (Table 8).

Recent studies suggest that it may now be prudent to rapidly address deficiencies in data collection for a variety of reasons. Eutrophic and hypoxic conditions have been reported for coastal sites around India and Sri Lanka (Diaz et al., 2011), whilst model studies show that corals in the Maldives, and possibly more generally across the South Asian region, appear highly sensitive to low levels of nutrient enrichment (Guan et al., 2020). This latter study is particularly concerning as it indicated detrimental impacts on Maldivian coral reefs following an increase of only 0.02 µmol NO_3^- L^{-1} above local optimum conditions. Nutrient concentration

TABLE 8 Assessment matrix summarizing existing literature evidence for known nitrogen pollution impacts on coral reefs in India, Sri Lanka and the Maldives, the direction of change (if possible to determine) for the South Asia region, and whether such changes can clearly be attributed to nitrogen pollution.

Change due to nutrient enrichment	Does Observational Evidence Exist			Regional direction of change	Can be attributed to nitrogen input
	India	Sri Lanka	Maldives		
Increased macroalgal dominance	✓	✓	X	↑	X
Crown-of-thorns outbreak	?	✓	✓	?	X
Increased bioerosion (sponges etc)	?	?	?	?	X
Disruption of coral-zooxanthellae relationship	?	?	?	?	X
Increased prevalence of coral diseases	?	?	?	?	?
Reduction in community diversity	✓	✓	✓	↑	X
Increased bleaching susceptibility	?	?	?	?	X
Changes to recruitment success	?	?	?	?	X
Reduced resistance to ocean acidification	?	?	?	↑?	X
Increased water column turbidity	✓	✓	✓	↑	X
Change to water column nutrient cycling	?	?	?	?	?
Red tides/harmful algal blooms	✓	?	✓	↑	?
Reduced oxygen concentrations	✓	✓	?	↑	✓

Key to symbols: Tick marks in columns 2-4 signify that observational evidence has been reported to support the change due to nutrient enrichment (column 1), cross marks signify that no observational evidence has been reported, question marks signify that existing studies are inconclusive. In column 5 arrows are used to signify the direction of regional trends (increasing or decreasing) whilst question marks signify that trends remain inconclusive or poorly documented. In column 6 tick marks signify a clear attribution of the regional trend to nitrogen inputs, cross marks that the link to nitrogen cannot be attributed with current available evidence and question marks that the reported regional trend is unclear or due to additional factors.

increases of this magnitude relative to the regional mean are currently difficult to identify from extant nutrient observations (Boyer et al., 2018) and thus require more frequent targeted sampling efforts. Attributing nutrient pollution as a contributory and possibly even causative factor behind coral loss across the South Asian region in recent decades is compounded by the limited quantitative information available. Such knowledge gaps place significant limitations upon coral management and conservation decision making and do little to highlight the true scale of the nutrient problem or the severity of its impact on marine ecosystems.

To address this, six key research topics for future research activities have been identified as the most critical for improving abilities to quantify and manage nutrient enrichment impacts on South Asian coral habitats.

4.3.1 What is the magnitude of anthropogenic nutrient inputs to South Asian coral reefs?

Quantification of this most basic of requirements remains extremely difficult. Efforts to assess regional and national nutrient inputs to the ocean have developed rapidly in recent decades but as yet, the ability to quantify nutrient inputs to specific marine habitats remains challenging. Identifying, quantifying and evaluating nutrient inputs to coral habitats is a key objective of further study. Linked to this is a clear need to incorporate marine biogeochemical measurements within coral monitoring activities.

4.3.2 What are the regional environmental optimum conditions for coral reef growth and are anthropogenic nutrient inputs distorting this optimum?

Identifying baseline conditions against which to assess anthropogenic inputs is essential. What are those baseline conditions?, are they changing?, and if so how rapidly? are questions that need answering. Only once such baseline conditions are understood, can impacts and the severity of those impacts be identified and addressed. Presently, anthropogenic nutrient inputs are typically viewed in a qualitative or semi-quantitative sense, and additional quantitative measures are urgently required.

4.3.3 What is the regional connectivity between different coral sites and pollution sources?

Nutrient inputs from riverine inputs, contaminated groundwater, and terrestrial runoff are key variables to quantify and model when assessing nutrient inputs to coastal waters, but once nutrients have reached the ocean, how does the local circulation retain or disperse nutrient inputs to increase or decrease the risk to specific habitats and ecosystems? Presently, the pathways linking nutrient sources and coral habitats are poorly known, with even some aspects of local oceanography and the regional circulation considered incomplete. Improved understanding of regional connectivity, particularly *via* numerical models, may help identify regions more prone to enrichment problems due to water retention or low flushing rates, as well as those areas potentially less prone to advective problems.

4.3.4 To what extent is it possible to predict nutrient enrichment impacts?

There is growing awareness that inadequate knowledge of how environmental factors influence the symbiotic relationship between coral and zooxanthellae hinders the ability to predict how that relationship may break-down or change when environmental conditions themselves change. While much effort has been put into understanding the effects of heat stress on the symbiotic relationship, more effort should be put into understanding how nutrients influence that relationship. Separately, efforts to identify low cost indicator tools that can be widely distributed to aid monitoring efforts, including the use of indicator species, should be considered within preventative management plans.

4.3.5 Do coral growth metrics (calcification, linear extension, and bulk density) evidence nutrient enrichment problems?

Recent research suggests that coral skeletal formation may be measurably impacted by nutrient enrichment (e.g. Iijima et al., 2021; Buckingham et al., 2022). Though still an emerging field of research it follows that if coral growth metrics such as calcification rate or bulk density can be easily measured in the field, that they may provide additional indications, or even early warning, of nutrient enrichment problems and inform mitigation actions. Presently such growth metrics remain poorly documented for the South Asian region but with further research may represent a valuable source of new information on nutrient enrichment impacts.

4.3.6 What management actions are needed?

The need to preserve and protect coral reef habitats must be handled within appropriate management and legislative structures. The imposition of marine protected areas may protect corals from physical destruction, but how effective are such protections from dissolved nutrient pollution? Are nutrient dispersal pathways adequately considered? Could more be done to enforce existing legislation to minimize or remove the impacts of pollution? Such questions link to wider aspects of socioeconomic development, raising the prospect of difficult decision-making while balancing coral protection and economic prosperity, but it is precisely these connected issues that must be tackled.

5 A way forward

Addressing the identified research gaps will require coordinated regional efforts due to the transboundary nature of pollutant dispersal and the variability in regional research infrastructure and capability. Yet precisely this coordinated effort is already handled under the auspices of the South Asia Co-operative Environment Programme (SACEP), a regional inter-governmental organization that promotes the protection, management, and enhancement of the regional environment. In particular, the problems of marine nutrient pollution and coral habitat loss are already recognized and actively addressed (SACEP, 2008; SACEP, 2017; SACEP, 2019). However, the two problems of nutrient pollution and coral habitat loss appear to be currently handled separately, so coral reef monitoring has not generally been undertaken with nutrient pollution in mind, nor do water quality monitoring activities fully consider impacts on all aquatic habitats or biomes. Small changes or additions to existing activities including measurement of additional environmental parameters such as ambient inorganic and organic nutrient concentrations or more frequent monitoring of water quality in sensitive habits such as coral reefs may therefore be a cost-effective means of resolving the true impact of anthropogenic nutrient inputs on the region's coral reefs, whilst also strengthening the protection and management of the region's marine environment without resorting to the establishment of bespoke sampling networks or surveys.

Author contributions

SP and AT conceived the study, SP coordinated contributions from all authors and wrote the first draft. All authors contributed to manuscript revision, read and approved the submitted version.

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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