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Tracing the Environmental Pathways of Emerging Contaminants: From Coastal and Urban Areas to Desert and Mangrove Sinks in the UAE and the Maldives

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Contents

ABSTRACT

Anthropogenic contamination is impacting our modern world in unprecedented ways, starting from urban areas and impacting uninhabited remote locations. Microplastic contamination poses critical threat to important blue carbon ecosystem impacting environmental degradation and seriously affecting human health and biota. Understanding environmental, particularly atmospheric transport of microfibers is crucial because so far scientific studies have focused a lot on microplastic pollution, however, there remains significant knowledge gap in terms of in-depth understanding of the airborne pathways of contamination. There is a critical threat arising from microfibers as it has been discovered practically everywhere, not only densely populated industrialized urban centers but also from remote locations like deserts, mangrove ecosystems, uninhabited mountain catchments and even in Antarctica. The absence of significant attention to this airborne contamination amounts to severe potential damage in the blue carbon ecosystems like mangroves. Moreover, contamination affecting the ecosystems and biota can easily translate to human health hazard as multiple communities depend on blue carbon ecosystems, drawing resources and food from them. It is crucial to address this issue as microfiber contamination impacts not only the food chain, but microplastics can act as reservoirs and pathways for environmental contaminants as these tiny contaminants exhibit sponge like behaviour absorbing, concentrating, and transporting toxic chemicals and pathogens to ecosystems, wildlife, and humans. The vector effect originating from these pathways is concerning due to

1. Enabling of trophic transfer among microorganisms like zooplankton impacting the food chain
2. Long range transport impacting remote areas and transforming them into sinks for contamination
3. Human health hazards stemming from direct ingestion or inhalation of contaminants

In this thesis, we present results from our analysis of microfiber contamination in the uninhabited remote deserts of the United Arab Emirates, and also investigate the threat unmanaged anthropogenic microplastic contaminants impacting the blue carbon mangrove ecosystem of the Republic of Maldives. We seek to have a broader understanding of the environmental pathways of emerging contaminants so that future work can be implemented in improving monitoring of vector effects of airborne contaminants and thus enable suitable policy formulation, development of effective strategies for mitigating microfiber contamination.

Keywords – microfiber, FT-IR/FT-NIR, UAE deserts, contaminant pathways, long range transport

Chapter 1 General introduction

1.1. Mangrove and algal blue carbon ecosystems under pressure from climate change and environmental contamination.

Mangrove forests are often called “blue carbon” ecosystems because they capture carbon dioxide through photosynthesis and store large amounts of organic carbon for long periods, especially in waterlogged, oxygen-poor sediments. This long-term burial makes mangroves important for climate-change mitigation, while also supporting coastal protection and fisheries. However, these benefits depend on mangroves remaining healthy and able to keep pace with sea-level rise and other climate-driven stressors (Howard et al., 2017; Macreadie et al., 2022). Mangrove sediments have been acting as long-term sinks for plastics, with burial increasing since ~1950, clearly linked to mass plastic production (Huang et al., 2024).
At the same time, mangrove sediments are increasingly recognized as traps for human-made debris, including microplastics. The structure of mangroves—dense roots, low-energy water flow, and high sedimentation—encourages particles to settle and remain buried. Evidence from coastal regions shows that plastic burial in mangrove sediments has increased strongly since the mid-20th century, supporting the idea that mangroves can act as long-term sinks for microplastics and other particles (Martin et al., 2020). This “sink” role is not necessarily permanent. Microplastics retained in sediments can be resuspended by tides, storms, and erosion, meaning mangroves can shift from being storage sites to being secondary sources that re-release plastics to the coastal ocean. Recent field-based work has emphasized these sink–source dynamics, including retention of floating microplastics during tides and resuspension from sediments (Huang et al., 2024). This is important because climate change is expected to increase extreme events (strong storms, flooding) in many regions, potentially increasing the mobilization of buried microplastics (Huang et al., 2024).
Microplastics also interact with other contamination problems. They can act as vectors for plastic additives and adsorbed pollutants, including persistent organic pollutants and metals, and they can be ingested by invertebrates and fish that use mangroves as nursery habitats (Rochman et al., 2013). Even when microplastics remain buried in sediments, they represent a long-term stressor that adds to existing pressures from nutrient enrichment, heavy metals, and habitat disturbance (Martin et al., 2020; Alongi, 2014). In this way, mangroves illustrate a key challenge for blue carbon management: ecosystems that efficiently store carbon may also efficiently retain contaminants, while climate-driven disturbance can influence both carbon stability and pollutant remobilization (Howard et al., 2017).

Within marine ecosystems, blue carbon ecosystems, such as mangrove forests, seagrass meadows, and salt marshes are particularly important due to their roles in carbon sequestration, shoreline stabilization, and protection against floods and storms. However, these ecosystems are increasingly vulnerable to plastic pollution. Primary functions of blue carbon ecosystems such as Mangrove wetlands are defense against floods / storms and Carbon sequestration. Furthermore, blue carbon ecosystems can block and intercept MPs. The blue carbon ecosystem adeptly blocks and intercepts microplastics, which results in significant benefits. Evidence shows that wetland ecosystems with high sedimentation rates like mangroves are a natural sink of MPs. Previous investigation suggests that mangrove ecosystems act as a filter of MP and POCs from surface sediments (Furukawa et al., 1997) (Kristensen et al., 2008). Liu et al. (2022) describes that mangroves ecosystem clean and retain contaminants and furthermore act as an ecological intercept for microplastics. Mangrove forests possess dense networks of above-ground roots, trunks, and pneumatophores that efficiently trap plastic litter during tidal cycles. Studies indicate that the quantity, weight, and surface area coverage of microplastics within mangroves are strongly influenced by landscape characteristics and vegetation structure (Cappa et al., 2023). While this retention provides a valuable ecosystem service, prolonged accumulation of plastic debris can severely impair mangrove health. Plastic entanglement and the covering of pneumatophores can restrict oxygen exchange, leading to physiological stress, reduced root growth, and limited space for seedling establishment. Plastic layers over aerial roots can induce anaerobic sediment conditions, negatively affecting benthic organisms such as mangrove crabs, benthivorous fish, mollusks, and oysters. In some cases, plastic debris obstructs water flow during high tides, causing filter-feeding organisms to starve. Larger marine fauna, including sea turtles, may ingest plastic bags mistaken for prey, often with fatal consequences. Additionally, plastic debris can generate noise under windy conditions, disturbing migratory birds and crustaceans (Kesavan et al., 2021; Bijsterveldt et al., 2021). Tourism-related activities further exacerbate plastic deposition in mangrove ecosystems, with non-degradable items such as plastic bags, bottles, and food packaging frequently becoming lodged within mangrove root systems (Cappa et al., 2023). The persistence of plastics within blue carbon ecosystems is linked to their physicochemical properties and environmental exposure. Over the past seven decades, plastic production has increased dramatically due to attributes such as durability, flexibility, low cost, and resistance to water. In marine and coastal environments, plastics do not fully degrade but instead fragment under the influence of ultraviolet radiation, temperature fluctuations, salinity, biological activity, and mechanical stress from waves and currents (Bandh et al., 2023). These processes result in the formation of plastic fibers, fragments, and particles smaller than 5 mm, classified as microplastics. Coastal wetlands receive plastic inputs from both terrestrial sources, such as river runoff, sewage discharge, fishing activities, and industrial effluents, as well as from oceanic transport driven by winds and currents. Low-energy environments like mangrove ecosystems are particularly prone to plastic accumulation. Microplastics, commonly composed of polyethylene, polypropylene, and polystyrene pose significant risks to marine organisms by impairing growth, reproduction, and adaptive capacity. Moreover, microplastics can adsorb organic pollutants and heavy metals, further amplifying their ecological toxicity (Bandh et al., 2023). Despite growing recognition of plastic pollution across marine environments, blue carbon ecosystems particularly mangroves, remain understudied, even though they are highly productive systems and effective sinks for microplastics. Mangroves are often regarded as the last line of defense, intercepting land-based plastic pollution before it enters open oceans. Their strong adsorption and filtration capacity enables them to trap contaminants; however, extreme wave events, flooding, and storm surges may remobilize stored plastics, shifting their role from sinks to secondary sources of pollution. While natural mangroves are known to intercept tidal microplastics efficiently, especially within sediments, the retention capacity of artificially planted mangroves remains poorly understood (Ding et al., 2022). Furthermore, the role of mangroves as Nature-Based Solutions (NBS) for mitigating plastic pollution has received limited scientific attention. Significant knowledge gaps persist regarding the relationships between waste management practices, ecosystem services, and plastic accumulation in mangrove habitats. The absence of standardized sampling methods further complicates comparative assessments and the development of effective mitigation policies (Luo et al., 2021).

Poor waste management in the Maldives has encouraged widespread open dumping and frequent burning of household waste on inhabited islands, largely because of limited space for disposal and insufficient waste infrastructure. Open burning of mixed waste, especially plastics, creates substantial local air and soil pollution and can generate pyroplastics, meaning plastics that are partially combusted and physically reshaped by heat. Under uncontrolled burning conditions, plastics that begin as macro, meso, or micro sized items can undergo thermal degradation, charring, and fragmentation, increasing brittleness and promoting the release of secondary microplastics into nearby terrestrial and marine environments (Turner et al., 2019; Hess et al., 2025). The practice of burning waste can be particularly harmful as they contain heavy metals like cadmium and lead within the well-rounded clasts, originating from unused electrical and packaging plastics (Turner et al., 2019). That is a major concern from pyro plastic pollution as the likelihood of heavy metals entering the food chain can impact environmental health, biota and also human health (Turner et al., 2019). Work from the Maldives, particularly in the Faafu Atoll, has reported both microplastics and charred microplastic particles in coastal sediments. The most frequently observed polymers include polyethylene (PE), polypropylene (PP), polyethylene terephthalate (PET), and polystyrene (PS), with concentrations in the same general range as those reported from other coastlines influenced by open burning and poorly managed waste (Saliu et al., 2018; Utami et al., 2023). Comparable polymer mixtures and pollution signatures have also been described on Indonesian beaches and along South African shores, where uncontrolled burning is linked to plastiglomerates and pyroplastics that can be enriched in organic contaminants, suggesting this pathway is relevant well beyond the Maldives (Utami et al., 2023; Zardi et al., 2025). Although reported concentrations differ across locations and sampling periods, these studies collectively point to open dumping and burning as under-recognized yet important contributors of microplastics to both terrestrial and marine systems. This evidence complicates the assumption that incineration or thermal processing necessarily removes plastic pollution, since incomplete combustion can instead transform plastic waste into altered fragments that persist and disperse (Yang et al., 2021; Turner et al., 2019).

Algae, ranging from microscopic phytoplankton to large macroalgae, play a fundamental role in marine ecosystems by supporting food webs, contributing substantially to global primary production, and influencing oxygen and nutrient cycling (Falkowski et al., 1998; Field et al., 1998). In coastal regions, macroalgae such as kelps also act as habitat-forming species that provide structure and shelter for diverse marine communities (Teagle et al., 2017). Algal systems are increasingly exposed to climate-driven stressors, particularly ocean warming and marine heatwaves, which can disrupt photosynthesis and growth. At the physiological level, algae often respond to thermal and light stress through changes in pigment composition, including reductions in chlorophyll content and adjustments in carotenoid-based photoprotective mechanisms (Fernández-Marín et al., 2021; Perin et al., 2023). When algae experience heat stress, their photosynthetic machinery can become less efficient. A common pattern is that chlorophyll levels decline under strong or prolonged stress (Hu et al., 2020), reflecting pigment degradation and damage to photosystems. In parallel, carotenoids often become relatively more important because they help protect cells by quenching excess energy and limiting oxidative damage. This shift is frequently observed as a decrease in the chlorophyll-to-carotenoid ratio during heat stress (Napaumpaiporn et al., 2016). Carotenoids are not just additional pigments they are part of active photoprotection. In many algae and plants, the xanthophyll cycle (involving pigments such as violaxanthin, antheraxanthin, and zeaxanthin) helps safely dissipate excess absorbed light energy as heat, which reduces the formation of harmful reactive oxygen species. Heat stress can alter xanthophyll cycle pigments and their balance, linking thermal conditions to changes in photoprotection and photosynthetic performance (Perin et al., 2023; Fernández-Marín et al., 2021). Mangroves and algal ecosystems are central to coastal resilience: mangroves store carbon in sediments and protect shorelines, while algae power food webs and regulate biogeochemical cycles. Yet both are increasingly exposed to a "double pressure" of climate change and contamination. Mangroves can trap microplastics effectively, but disturbance may remobilize stored particles. Algae, meanwhile, show measurable physiological responses to warming often involving chlorophyll decline and carotenoid-linked photo protection signaling potential shifts in productivity and ecosystem structure. Managing blue carbon and marine health therefore requires addressing carbon storage, biodiversity, and pollution together, rather than treating them as separate problems.

1.2. The rise and swift proliferation of plastic materials globally.

Plastic pollution has become one of the most pressing and pervasive environmental issues globally, largely because of the dramatic increase in plastic production over the last century (Geyer et al., 2017). Modern plastic manufacturing began in the early 1900s with the invention of Bakelite in 1907, which marked the beginning of the synthetic plastics era. This breakthrough introduced durable, heat-resistant material that could be easily molded, fundamentally transforming industrial production and enabling numerous technological advancements. Nevertheless, it was not until the 1950s that plastic manufacturing expanded rapidly, leading to profound changes in industries and daily life worldwide. Since that period, global plastic production has increased nearly 230 times, reaching approximately 460 million tonnes per year by 2019. Alarmingly, production has more than doubled over the past two decades alone, highlighting both the escalating demand across multiple sectors and society's growing dependence on plastics materials that are highly functional yet persist in the environment.

Plastics are valued for their lightweight nature, strength, and durability, making them suitable for a broad range of applications. They are widely used in construction, agriculture, electronics, healthcare, and transportation, and have become integral to modern economies (Thompson, 2006). Among these applications, consumer packaging represents the largest driver of plastic demand. Nearly one-third of global plastic resin production is used for packaging, particularly for single-use products such as bags, bottles, wrappers, and containers that are often discarded within a year of manufacture (Plastics Europe, 2022). These short-lived items dominate waste streams and frequently end up in landfills, incineration facilities, or more problematically are dispersed within natural environments. This culture of disposability has intensified reliance on plastics and significantly contributed to the escalating global waste crisis.

Currently, the world generates approximately 350 million tonnes of plastic waste each year, a figure that continues to grow in parallel with production levels (OECD, 2022). However, the challenge extends beyond the sheer volume of waste to include deficiencies in waste management practices. An estimated 82 million tonnes of plastic waste are improperly managed through insecure landfills, inefficient recycling systems, or inadequate incineration processes (OECD, 2022). Weak waste management infrastructure, particularly in developing regions, allows substantial quantities of plastic to escape into the environment. Each year, roughly 19 million tonnes of mismanaged plastic waste enter terrestrial and aquatic ecosystems (OECD, 2022; Ritchie, 2023). Urban and rural landscapes are often the first affected, accumulating around 13 million tonnes of plastic, while an additional 6 million tonnes are transported into rivers and coastal zones. From these sources, approximately 1.7 million tonnes of plastic reach the oceans annually, with 1.4 million tonnes delivered via rivers and 0.3 million tonnes originating from coastal activities (OECD, 2022; Ritchie, 2023). Even though it accounts for a mere 0.5% of the planet's total plastic waste, the resulting harm to marine life and habitats is devastating and persistent. In the European Union, roughly 50% of maritime litter consists of disposable plastic products designed for a single application before being discarded. In response, the EU enacted regulations to prohibit several types of single-use items such as straws, plates, cotton buds, and balloon sticks from its markets effective July 3, 2021 (European Commission. Directorate General for Environment., 2021). These concerns are further addressed by the United Nations' Sustainable Development Goals, specifically Goal 14, which focuses on the conservation and sustainable management of oceanic resources. A key component, target 14.1, aims to substantially decrease all forms of marine pollution by 2025, with progress monitored through the density of floating plastic debris and coastal eutrophication (Walker et al., 2021). While these legal frameworks are a major advancement, the scale of the crisis requires continued global expansion of such efforts. Additionally, because consumer habits are central to the problem, it is vital to promote recycling and sustainable substitutes through awareness campaigns. Ultimately, solving the complex challenge of plastic pollution necessitates a collaborative approach involving industrial changes, government policy, and a fundamental shift in public behavior toward reusable goods.

1.3. Primary origins of microplastic pollution in the ocean

Plastic entering the oceans is widely linked to two core problems: littering and poor waste management. These behaviors do not only create visible pollution on coastlines bringing negative social and economic effects (Aretoulaki et al., 2021) but also create direct risks for marine ecosystems. Marine animals are frequently harmed by plastic debris through ingestion, injury, entanglement, and suffocation, which can lead to death (Gregory, 2009). Data from recent studies suggests that an immense 8.3 billion tons of plastic have been manufactured since the material was first introduced (Geyer et al., 2017), yet a staggering 91% of that total has never been recycled, with only 9% actually undergoing the recycling process (Geyer et al., 2017). Every year, it is estimated that roughly 1.7 million tonnes of plastic refuse find their way into marine environments (OECD, 2022). This calculation is based on several factors, including the volume of waste produced per person annually, the plastic density of that waste, and the specific portion of mismanaged refuse that is likely to become aquatic pollution. Once in the environment, mismanaged items like improperly discarded shopping bags, food containers, and other single-use plastics are subjected to intense weathering. These changes are sparked by abiotic environmental factors, such as the ultraviolet (UV) radiation from the sun, mechanical friction caused by wind, and the ongoing chemical and physical interactions with seawater. Over time, these conditions trigger specific breakdown mechanisms, such as photodegradation, oxidative degradation, and hydrolysis, which cause larger plastics to splinter into tiny fragments known as secondary microplastics (Andrady and Koongolla, 2022). These secondary particles present a particular difficulty for researchers because their specific origins are hard to identify, making it complex to calculate exactly how much macroplastic waste is being converted into microplastics. While common perception often blames mismanaged bulk waste for the majority of ocean contamination, scientists have identified seven other primary sources of microplastics that add significantly to the problem: vehicle tires, synthetic clothing fibers, marine coatings, road markings, personal care products, industrial plastic pellets, and urban city dust (Lassen et al., 2015; Magnuson et al., 2016; Essel et al., 2015).

Each of these categories plays a major role in the spread of microplastics throughout the sea. These tiny microplastic particles escape into the environment at various points in a product's life, particularly during routine maintenance and use, such as cleaning clothes or operating vehicles (Boucher and Friot, 2017). It is therefore vital to understand that everyday human habits have a direct impact on the pollution of the seas, endangering both human health and global biodiversity. A detailed look at the global discharge of microplastics reveals a notable fact: nearly two-thirds, or 63.1%, of these pollutants originate from just two main sources. The largest single contributor is the laundering of synthetic fabrics, accounting for 34.8%, followed by the erosion of tires during vehicle use at 28.3% (Boucher and Friot, 2017). These statistics emphasize the urgent need to target these specific sources to protect the marine environment. Effectively solving the pervasive challenge of microplastic pollution requires a comprehensive, holistic strategy that focuses on both the prevention of waste and the mitigation of current pollution. This approach must include the enforcement of stricter industrial regulations regarding plastic production and use, as well as the improvement of global waste management systems to increase recycling rates and stop the breakdown of larger plastic waste into secondary microplastics.

1.4 Microplastics as reservoirs and pathways for environmental contaminants

The ecological consequences of microplastics are exacerbated by their capacity to sequester and redistribute environmental toxins. Within aquatic environments, these particles function as transport mechanisms for diverse chemical pollutants (Fred-Ahmadu et al., 2020). Due to their hydrophobic characteristics and elevated surface-area-to-volume ratios, microplastics readily adsorb trace metals and persistent organic pollutants (POPs) (Ashton et al., 2010). Evidence suggests that organic pollutants can accumulate on plastic surfaces at concentrations many times higher than those found in the surrounding water, a process supported by the water-polymer partition coefficient (KP/W [L/kg]), which drives the retention of these substances (Atugoda et al., 2021). This sorption dynamic is further modified by the chemical profile of the contaminants and the degree of surface weathering (Fred-Ahmadu et al., 2020). Indeed, aging through exposure to mechanical, thermal, radiative, biological, and oxidative stressors heightens the affinity of plastics for harmful organic compounds and metals (Albertsson et al., 1987). For example, UV-induced aging has been shown to substantially improve the adsorption capacity of polyvinyl chloride and polystyrene for hydrophilic pollutants like ciprofloxacin (Liu et al., 2019). Such aging also diminishes the hydrophobicity of the plastic, thereby increasing its ability to attract hydrophilic substances (Hüffer and Hofmann, 2016). In addition to external pollutants, plastics are manufactured with various additives including stabilizers, plasticizers, fillers, and reinforcing fibers intended to enhance performance, which may subsequently migrate into the environment (Deanin, 1975; Sridharan et al., 2022). Since these additives are generally not chemically bonded to the polymers, they readily leach out, particularly as microplastics break down into smaller pieces. This process exposes marine life to a complex toxicological mixture when plastic debris is consumed (Hermabessiere et al., 2017). Organisms may consume plastic particles along with their associated load of absorbed metals and chemicals, potentially leading to bioaccumulation and eventual transfer through the food web to human consumers (Seltenrich, 2015). Common plastic additives identified in marine fauna include nonylphenols (NPs), phthalates (PAEs), and polybrominated diphenyl ethers (PBDEs) (Hermabessiere et al., 2017). Studies have documented PBDEs and PAEs in fish (Mariussen et al., 2008; Peng et al., 2007; Kelly et al., 2008; Panio et al., 2020), marine mammals (Yogui et al., 2011; Rotander et al., 2012; Andvik et al., 2023; Routti et al., 2021), mussels (Johansson et al., 2006; Dosis et al., 2016), and corals (Raguso et al.,2022). Simultaneously, NPs have been found in algae, zooplankton, and fish species (O'Halloran et al., 1999; Naylor et al., 1992; Graff et al., 2003). Multiple investigations have highlighted the acute toxicity resulting from plastic leachates in marine organisms. For instance, leachates from recyclable materials such as polycarbonate (PC), polypropylene (PP), and high-density polyethylene (HDPE) interfere with the settlement and survival of barnacle larvae (Amphibalanus amphitrite) (Li et al., 2016). Bejgarn et al. (2015) tested 21 weathered plastic products on the marine copepod Nitocra spinipes, finding that 38% of the samples produced toxic leachates, harmfully impacting the copepods. Notably, the toxicity of several samples increased significantly following UV irradiation, suggesting that sunlight accelerates the release of hazardous compounds. Furthermore, Drosophila melanogaster exposed to leachates from oxo-degradable and virgin polyethylene and polypropylene demonstrated increased loss of heterozygosity (LOH) and genomic instability, which can facilitate tumor development (Cappucci et al., 2024). Toxic effects have also been documented in Daphnia magna following exposure to polyvinyl chloride (PVC) leachates (Lithner et al., 2012). These collective observations highlight the severe environmental dangers of microplastic pollution, specifically regarding the impact of leachates and chemical additives on marine systems. As these substances accumulate within food webs, they present a persistent risk to both marine biodiversity and human health.

1.5 The veiled threat of microplastic fibers and shedding of microfibers from synthetic fabrics

Microfibers released from synthetic clothing during washing are now recognized as a major source of plastic contamination in aquatic environments (Boucher and Friot, 2017; Granek et al., 2022). These synthetic microfibers (MFs) consist of non-biodegradable polymers—including polyester (PE), nylon, polyethylene terephthalate (PET), rayon, acrylic, polypropylene (PP), and spandex—and they are typically defined as having a diameter below 5 mm (Sanuj et al., 2024). During laundering, mechanical agitation and chemical exposure cause fibers to break and detach from fabrics (De Falco et al., 2019). Many fibers are too small to be fully captured by standard wastewater treatment plants (WWTPs), so they can be released into rivers, lakes, and oceans, creating major risks for biodiversity and ecosystem function (Gago et al., 2018; González-Pleiter et al., 2020; Miller et al., 2017).

A key challenge is that microfibers persist for long periods. These materials resist biodegradation, allowing them to remain for decades and accumulate in marine habitats (Andrady, 2011; Siddiqui et al., 2023). Studies in Sweden, Finland, Australia, and San Francisco Bay show that even advanced wastewater treatment does not remove all microfibers from effluent (Magnusson & Wahlberg, 2014; Talvitie et al., 2017; Granek et al., 2022; Sutton et al., 2016). This shows that microfiber contamination is a global issue and is not limited to places with weak infrastructure (Henry et al., 2019).

Estimates of fiber release indicate very large emissions. Browne et al. (2011) reported that a single garment may shed more than 1,900 microfibers in one wash. Napper and Thompson (2016) found that a 6 kg wash load of acrylic textiles can release over 700,000 fibers. Using household-scale measurements, Galvão et al. (2020) estimated even higher release rates: on average, 18,000,000 microfibers per 6 kg load of synthetic fabric. Considering how often laundry is done worldwide, the amount of microfibers entering aquatic systems is extremely large and is expected to grow with urbanization, population increase, textile demand, and rising water use (Devi and Devi, 2024).

Microfibers are also widely ingested by marine organisms. Ingestion has been documented in crabs (Watts et al., 2015), lobsters (Murray and Cowie, 2011), birds (Zhao et al., 2016), and many fish species (Siddiqui et al. 2022; Macieira et al., 2021; Lusher et al., 2013; Possatto et al., 2011). This can occur through direct feeding, through respiration, or through trophic transfer (Covernton et al., 2021). In organisms, ingestion can reduce energy available for growth, lower nutritional intake, and change behavior (Watts et al., 2015; Siddiqui et al., 2023; Stienbarger et al. 2021). Because microfibers can accumulate in plankton, they can move up food webs to predators, including humans (Setälä et al., 2014; Torres et al., 2023). This creates public health concerns about seafood consumption (Barboza et al., 2018).

To reduce microfiber pollution, several interventions are needed, including more sustainable textile design. Researchers are studying what controls fiber shedding. Vassilenko et al. (2021) reported that shedding from polyester and nylon increases with fabric thickness and that shedding rates can differ by up to 850-fold between textile types. Adding filtration to washing machines is a practical step: lint traps can capture up to 46% of nylon fibers and 90% of polyester fibers (Vassilenko et al., 2021). In Parry Sound, Ontario, a community pilot project showed that installing filters in only 10% of households captured an average of 6.4 g of lint per week, equivalent to about 179,200 to 2,707,200 microfibers (Erdle et al., 2021). Overall, reducing microfiber pollution requires a combined approach using better filtration, improved fabric technology, and stronger public awareness, in order to protect marine ecosystems and reduce risks to human health. This multi-part strategy is essential to lowering microfiber emissions and supporting long-term ocean sustainability.

1.6. The paradox of microplastic fiber contamination in remote environment

Micro plastic pollution has been talked about by a plethora authors for quite some period of term; ranking microplastic pollution issues quite high in the list of environmental concerns. However, the threat of small micro fibers especially synthetic microfibers (MFs) consists of non-biodegradable polymers including polyester (PE), nylon, polyethylene terephthalate (PET), rayon, acrylic, polypropylene (PP), and spandex which are typically 5 mm or less (Sanuj et al., 2024) are much less talked about. We found a scarcity of literature on the topic of microplastic fibers, more so if we consider the phenomenon of occurrence of micro fiber in remote environments. Microplastic fibers are found in urban space, it is an expected scenario to find considerably abundant amount of micro fibers in large metropolis. Multiple studies demonstrate evidence of micro plastic fiber being ubiquitous; across continents in major cities like Shanghai, Taipei, London, Paris, Hamburg and urban conglomerates of South Africa (Liu et al., 2019; Dris et al., 2016; Wright et al., 2020; Han et al., 2024, Mutshekwa et al., 2025; Chen et al., 2024; Klein and Fischer 2019). Urban areas face the challenge of micro fiber contamination with abundance of fiber ranging from (2 to 355 particles$\cdot m^{-2}\cdot day^{-1}$ in Paris, 175 to 313 particles$\cdot m^{-2}\cdot day^{-1}$ in Dongguan, China; 575 to 1008 microplastics$\cdot m^{-2}\cdot day^{-1}$ in London, 1.41×10^4 to 1.67×10^4 fibers/m^3 in Beijing (Dris et al., 2016 ; Cai et al., 2017; Wright et al., 2020; Li et al., 2020). The paradox emerges when there are several studies reporting presence of micro fibers remote environments; alarmingly micro fiber has been detected in Antarctica (Aves et al., 2022), French Pyrenees Mountain catchments (Allen et al 2019, deserts of Central Asia and Arizona (Chandrakanthan et al 2023), Southern Ocean in the southern hemisphere (Chen et al 2023), western Pacific, (Liu et al 2019). Superficially it might seem that it is a paradox that remote uninhabited locations have presence of copious quantities of micro fibers. This paradox has been addressed by some scientific studies which agree upon the hypotheses of very long-range atmospheric transfer of micro fibers (Aves et al., 2022). The study points out that microplastics were detected in every Antarctic snow sample, with a mean concentration of 29 particles per liter. This finding suggests that the particles may have been transported over distances of up to approximately 6,000 km, based on an assumed atmospheric residence time of 6.5 days. Microfibers, owing to their small size, low density, and aerodynamic properties, can remain airborne for extended periods and be transported over long distances from urban and coastal sources into remote areas (Wang et al. 2021; Chen et al. 2024). Evidence of such transport were collected during dust events in Arizona, where MPFs resulted preferentially transported by wind compared to other plastic fragments, achieving high deposition rates in desert soils (Chandrakanthan et al. 2023). Wind driven micro fiber transfer transport is driven by some key factors; Xiao et al.,(2023) in their work has identified that shape of microplastic fibres is critical in determining movement and residence time in the atmosphere. The authors elaborate that previous models often treated particles as spheres, but realistic non-spherical shapes behave very differently in air, their study incorporates modelling of microplastic fibres using their actual shapes, especially flat fibres; reaching the conclusion that flat fibers have a much longer atmospheric residence time (on average over 450 % longer) compared to equivalent spheres. This work establishes that smaller aerodynamic size increases transport potential, because of their shape, real fibers have a smaller effective aerodynamic size than a sphere with equivalent volume. Smaller aerodynamic size reduces the settling velocity speed at which particles fall out of the air allowing them to travel longer distances. The aerodynamic size determines how the air moves around a particle and that is influenced by shape and orientation relative to wind. The effective aerodynamic size of flat fibres is much smaller than their physical length would suggest, meaning they can stay suspended longer. Thus, shape assumptions in models matter, treating flat fibres as cylindrical or spherical (as many models do) substantially overestimates how quickly they deposit out of the atmosphere. Correcting for real shape reveals fibres are transported much more efficiently. This has implications for global distribution of micro fibers because fibres can remain airborne longer and travel farther. Atmospheric transport is an important pathway that can carry microplastics from urban and coastal sources to remote regions, potentially even across hemispheres. This supports observations of microplastics in remote mountain snow (Allen et al., 2019) and polar environments (Aves et al., 2022).

1.7. Key issues in the microplastic research field

A major difficulty in detecting microplastics accurately is controlling contamination during sampling and analysis. A common source is airborne microplastics, especially synthetic textile fibers that fall into samples as atmospheric deposition. These particles can easily enter environmental samples and distort results unless strict prevention measures are used throughout fieldwork and laboratory procedures (Brander et al., 2020). Without careful protocols, cross-contamination can cause researchers to overestimate microplastic abundance and concentration, which can misrepresent their real ecological impact (Bogdanowicz et al., 2021). Another central challenge is that broad comparisons across space and time are difficult. This is largely because microplastics are measured and identified using many different methods across studies. Methodological differences make it hard to compare results across regions or across different time periods. These inconsistencies limit the development of robust shared datasets and weaken long-term monitoring. For these reasons, standardized methods for assessing microplastic abundance and distribution are critical for reliable comparisons and for long-term monitoring success (Hidalgo-Ruz et al., 2012; Thornton Hampton et al., 2022). Reporting practices also create problems. Microplastic concentrations are reported in many different units, which complicates comparisons between studies. While some units can be converted (Hidalgo-Ruz et al., 2012), others such as items per square meter cannot, limiting how broadly those results can be used (Cunningham and Sigwart, 2019). It is therefore important to report in standardized units that can be directly compared with toxicology studies, which commonly report exposure as particle number per volume or mass (Burns and Boxall, 2018). A further issue is that many laboratory experiments use microplastic concentrations far above those found in nature, often by several orders of magnitude. This raises concerns about how well lab results reflect real environments, since ingestion rates and effects observed under high concentrations may not match natural exposure conditions. At the same time, because field measurements of nano and microplastic concentrations are still very difficult, studies using elevated levels can still be useful for understanding potential future risks. Challenges are even greater for nanoplastics, where appropriate analytical tools for measuring environmental concentrations are still limited (Koelmans et al., 2015). Given society’s dependence on plastics and their functional advantages, it is unlikely that environmental microplastic levels will fall soon. Instead, they may remain steady or increase. This makes it increasingly important to study biological responses to high concentrations, especially given possible risks to marine biodiversity and the stability of the global food web. Finally, many experiments use commercial microplastics that are uniform in polymer type, size, and shape. In contrast, environmental microplastics are highly variable mixtures of polymers with diverse sizes and geometries. This difference makes it difficult to apply laboratory results directly to real ecosystems, since organisms may respond differently to complex mixtures than to single, uniform particles. Single polymer studies are still useful because they help isolate biological effects, but they can miss interactions that occur under natural conditions. It is also important that environmental microplastics are rarely “clean”: they are often weathered and carry absorbed pollutants, which changes their physical and chemical properties. Reproducing these real-world conditions in the laboratory is very difficult, which strengthens the need for more environmentally realistic experimental designs to better understand how microplastics behave and affect organisms in natural settings (Phuong et al., 2017; Thornton Hampton et al., 2022).

1.8. Thesis outline

The widespread contamination of diverse environments by microplastic fibers like blue carbon ecosystems, coastal and urban regions, and remote, uninhabited depositional sinks represent a serious threat to environmental health. Its consequences extend beyond impacts on flora and fauna to affect entire ecosystems and the human populations that depend on these systems. Research on microplastic fibers continues to face major challenges, including methodological inconsistencies and difficulties in reliably evaluating their ecological impacts.

This thesis aims to tackle several of these pressing challenges by providing new insights. We begin with documenting and understanding the blue carbon mangrove and associated flora and fauna ecosystem in the mangroves of The Republic of Maldives and how microplastic contamination impacts the island nation. It is important to acknowledge that algal ecosystems work in conjunction with mangrove biota and play a very important role in carbon fixation. This led us to a pilot investigation touching upon thermal stress response of algal ecosystems in the face of rising sea temperature induced by climate change and global warming. Then our investigations focus on the central element regarding aeolian dispersion of micro fiber contaminants. Therefore, we commenced on an extensive and meticulous approach to track down the environmental pathways of microplastic fiber which is an emerging contaminant in the UAE, starting from the coastal and urban areas to the remote deserts.

In order to address the research objectives methodically, the thesis is organized into four distinct chapters, beginning with **Chapter 1** introducing relevant background information addressed by a comprehensive review of the relevant literature which sets the research objectives that guide the investigation. The introductory chapter is followed by **Chapter 2 and 3** comprising of investigation of plastic contamination led by poor waste management in The Republic of Maldives and a review paper published in a reputable journal, which offers essential background information through a thorough literature review clearly outlining the importance of blue carbon ecosystems. **Chapter 4** focuses on investigating the responses of algal and coral ecosystems to thermal stress under rising sea temperatures driven by climate change. **Chapter 5** comprises of the core of the thesis, discussing the emerging and often ignored threats of micro plastic fiber contamination led by wind driven transport and it consists of a research paper which is currently in preparation for submission. Finally, **Chapter 6** concludes the thesis by synthesizing the main findings. In the end an appendix follows summarizing conferences attended during the Ph.D. program.

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Chapter 2

Concern of plastic contamination in the mangrove ecosystem of Maldives and review and analysis of mangrove ecosystem in The Republic of Maldives

1. Introduction -

To date, studies focusing on marine sponges within Maldivian coral reef ecosystems have demonstrated contamination by microplastics (MPs) (Saliu et al., 2022). Even marine environments previously regarded as pristine are not exempt from MP pollution. Research conducted in the Faafu Atoll of the Maldives has provided clear evidence of MP contamination in seawater, along with the presence of phthalic acid esters (PAEs), indicating associated chemical pollution (Saliu et al., 2019). Despite these findings, the effects of microplastics on mangrove habitats in the Maldives remain largely unexplored. Earlier work by Saliu et al. (2018) reported MP contamination in Faafu Atoll, a region considered relatively remote. Although Faafu Atoll lies approximately 140 km south of the capital, Malé, and experiences minimal tourism and low population density, measurable levels of MP pollution were still detected. This suggests that microplastic contamination in the area is likely influenced by long-range oceanic transport, facilitated by the durability and buoyancy of MP particles. The study further indicates that the inner rim of the atoll is also affected by contamination originating from local, land-based sources. Notably, open air burning of plastic waste along island shorelines remains a common practice in the Maldives, representing an additional and ongoing source of plastic pollution. Poor waste management in the Maldives has encouraged widespread open dumping and frequent burning of household waste on inhabited islands, largely because of limited space for disposal and insufficient waste infrastructure. Open burning of mixed waste, especially plastics, creates substantial local air and soil pollution and can generate pyroplastics, meaning plastics that are partially combusted and physically reshaped by heat. Under uncontrolled burning conditions, plastics that begin as macro-, meso, or micro-sized items can undergo thermal degradation, charring, and fragmentation, increase brittleness and promote the release of secondary microplastics into nearby terrestrial and marine environments (Turner et al., 2019; Hess et al., 2025). The practice of burning waste can be particularly harmful as they contain heavy metals like cadmium and lead within the well-rounded clasts, originating from unused electrical and packaging plastics (Turner et al., 2019). That is a major concern from pyro plastic pollution as the likelihood of heavy metals entering the food chain can impact environmental health, biota and also human health (Turner et al., 2019). Work from the Maldives, particularly in the Faafu Atoll, has reported both microplastics and charred microplastic particles in coastal sediments. The most frequently observed polymers include polyethylene (PE), polypropylene (PP), polyethylene terephthalate (PET), and polystyrene (PS), with concentrations in the same general range as those reported from other coastlines influenced by open burning and poorly managed waste (Saliu et al., 2018; Utami et al., 2023). Comparable polymer mixtures and pollution signatures have also been described on Indonesian beaches and along South African shores, where uncontrolled burning is linked to plastiglomerates and pyroplastics that can be enriched in organic contaminants suggesting this pathway is relevant well beyond the Maldives (Utami et al., 2023; Zardi et al., 2025). Although reported concentrations differ across locations and sampling periods, these studies collectively point to open dumping and burning as under-recognized yet important contributors of microplastics to both terrestrial and marine systems. This evidence complicates the assumption that incineration or thermal processing necessarily removes plastic pollution, since incomplete combustion can instead transform plastic waste into altered fragments that persist and disperse (Yang et al., 2021; Turner et al., 2019).

2. Materials and Methods -

Sampling -

Sampling for the pilot study was performed in the Maldives in June 2023, using the Marine Research and High Education (MaRHE) Center as a logistics base (https://marhe.unimib.it/), which is located on Magoodhoo Island, Faafu Atoll (3°4’49.08”N, 72°57’57.19”E), the Republic of Maldives. Magoodhoo is an inhabited island (~ 900 inhabitants) that measures 900 × 450 m, located in the south-eastern part of the atoll rim, about 140 km south of the capital Malè. Samples were collected from the landfill area where unsegregated waste is disposed of, and where open burning of mixed waste is a common practice. Sampling of plastic debris was designed to capture a representative range of macroplastics (>25 mm), and mesoplastics (5–25 mm) following widely adopted size classifications in microplastic research (Hidalgo-Ruz et al., 2012; Frias et al., 2018). Given the heterogeneous composition of landfill waste, including bottles, wrappers, fishing nets, and partially burnt plastics, a stratified and systematic sampling approach was adopted.

The landfill area was first divided into visually distinct zones based on waste characteristics, including (i) recently deposited waste, (ii) aged and weathered waste, and (iii) burnt or partially combusted waste. This approach accounts for differences in degradation state and fragmentation processes, which are known to influence microplastic generation (Hidalgo-Ruz et al., 2012). Within each zone, sampling points were selected using a grid-based method to ensure spatial coverage across the landfill.
Macro- and mesoplastic items were collected manually from the surface layer (0–5 cm) using 1 m × 1 m quadrats placed at each sampling point. All visible plastic items, including PET bottles, bottle caps, wrappers, fishing-related materials, and deformed or burnt plastic fragments, were collected using stainless steel forceps or gloved hands. Items were visually classified according to morphology and apparent use (e.g., packaging, domestic, fishing-related), following established protocols (Hidalgo-Ruz et al., 2012; Claessens et al., 2013). Samples were stored in pre-cleaned glass or aluminium containers to prevent contamination.

Strict contamination control measures were applied throughout sampling and laboratory procedures. Cotton laboratory coats were worn, synthetic clothing was avoided, and all tools were rinsed with filtered distilled water prior to use. Procedural blanks were included to account for potential airborne or laboratory contamination, consistent with best practices in microplastic research (Hidalgo-Ruz et al., 2012; Frias et al., 2018).

Macro FTIR Analysis of plastic samples -

Analyses were carried out using a Nicolet™ iS™ 10 FTIR spectrometer (Thermo Fisher Scientific) equipped with a deuterated triglycine sulfate (DTGS) detector operating at room temperature. Spectra were acquired in Attenuated Total Reflectance (ATR) mode, with 32 co-added scans per measurement, over a spectral range of 4600–600 cm^{-1} and at a resolution of 4 cm^{-1}. A background spectrum was collected prior to each analysis. Polymer identification was confirmed using the patented COMPARE™ spectral matching algorithm included in the OMNIC software library, by comparing sample spectra with reference library spectra

3. Analysis and Results –

We found plastic filaments, unidentifiable components made of plastic, melted deformed plastics, PET bottles and bottle caps, soft drinks packaging wrappers.
From Macro FTIR analysis (Considered 45% or higher matches) we found polyvinyl chloride (63% match), polypropylene atactic (47% match), polyethylene high density (94% match), polyethylene propylene (47% match), cellulose nitrate (74% match), polyethylene (85% match), polyethylene terephthalate (89% match), polypropylene isotactic (47% match) (see table 1 in chapter 2 for details). The results of the FTIR scans produced peaks which were also matched with the library confirming the type of polymers (Figure 2 a, b, c, d, e, f, g and h in chapter 2). Earlier investigation (Saliu et al., 2022) confirmed the presence of that polyethylene (PE) and polypropylene (PP) in the coral reef surrounding Magoodhoo, and also occurrence of microplastics and charred microplastic particles in coastal sediments of the island. Presence of contaminants in coral reefs around the island indicates that landfills and practice of burning waste deteriorate the ecosystem, posing risk to the ecosystem and biota.

Table 1

Sample	Type of polymer	Percentage match with FTIR library
polyethylene terephthalate	PET water bottle	89
polyvinyl chloride	filament	63
cellulose nitrate	cellophane wrapper from soft drinks bottle	74
polyethylene propylene	food wrapper	47
polypropylene atactic	food packaging wrapper	47
Polypropylene isotactic	plastic bottle cap	89
polyethylene	burnt plastic	85
polyethylene high density	unburnt plastic	94

Figure 1 -
Specimens of plastic collected from landfill site in Magoodhoo Island in Maldives

Figure 2a, 2b, 2c, 2d, 2e, 2f, 2g and 2h (please turn over to next page) – Macro FTIR spectra of different polymers found within the samples.

Figure 2 a –

Macro FTIR spectrum polyvinyl chloride

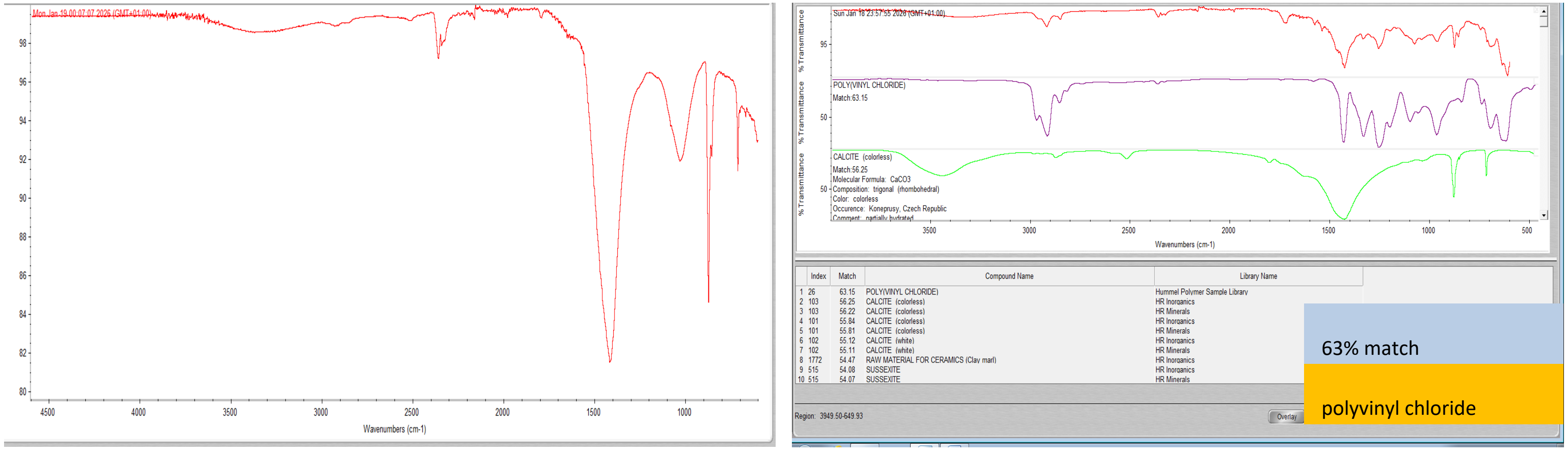


Figure 2 b –

Macro FTIR spectrum polypropylene atactic

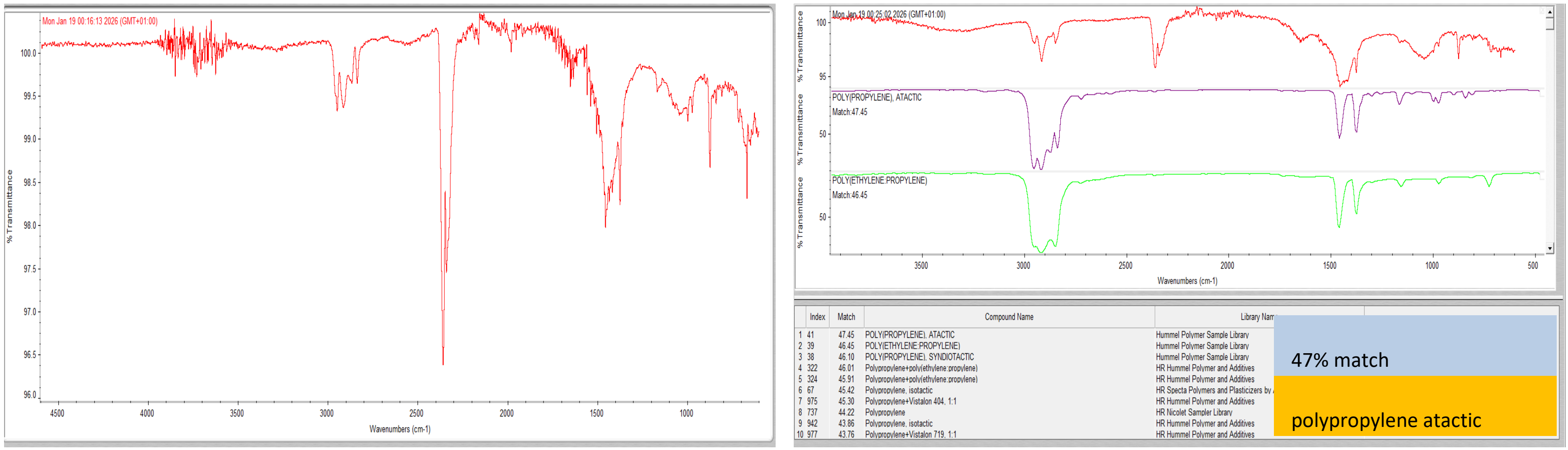


Figure 2 c –

Macro FTIR spectrum polyethylene (high density)

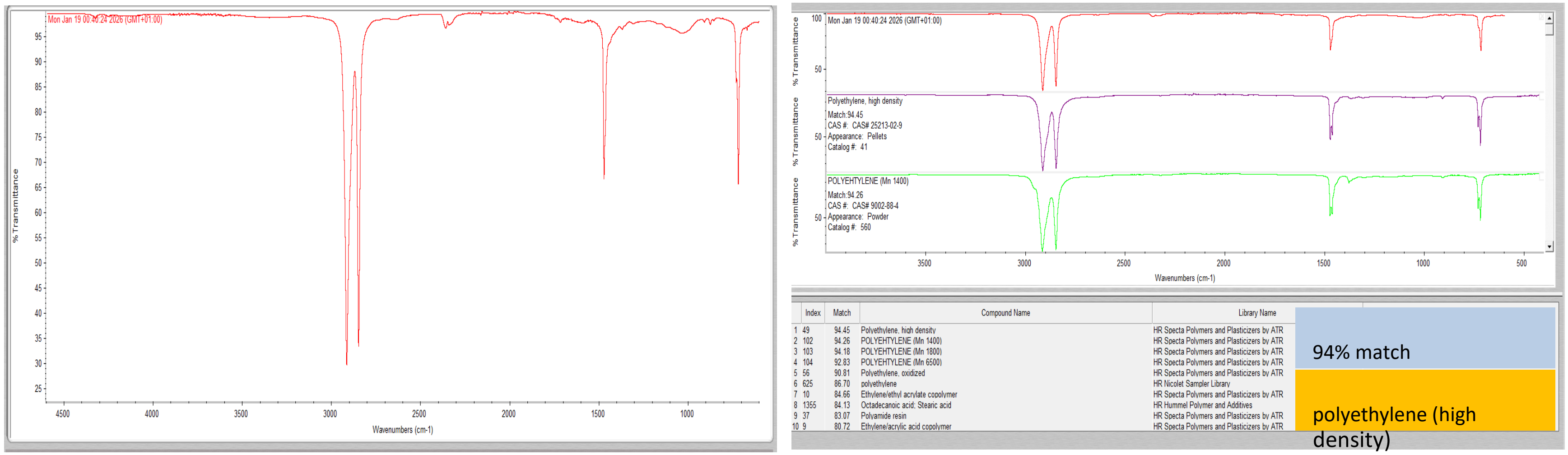


Figure 2 d –

Macro FTIR spectrum polyethylene propylene

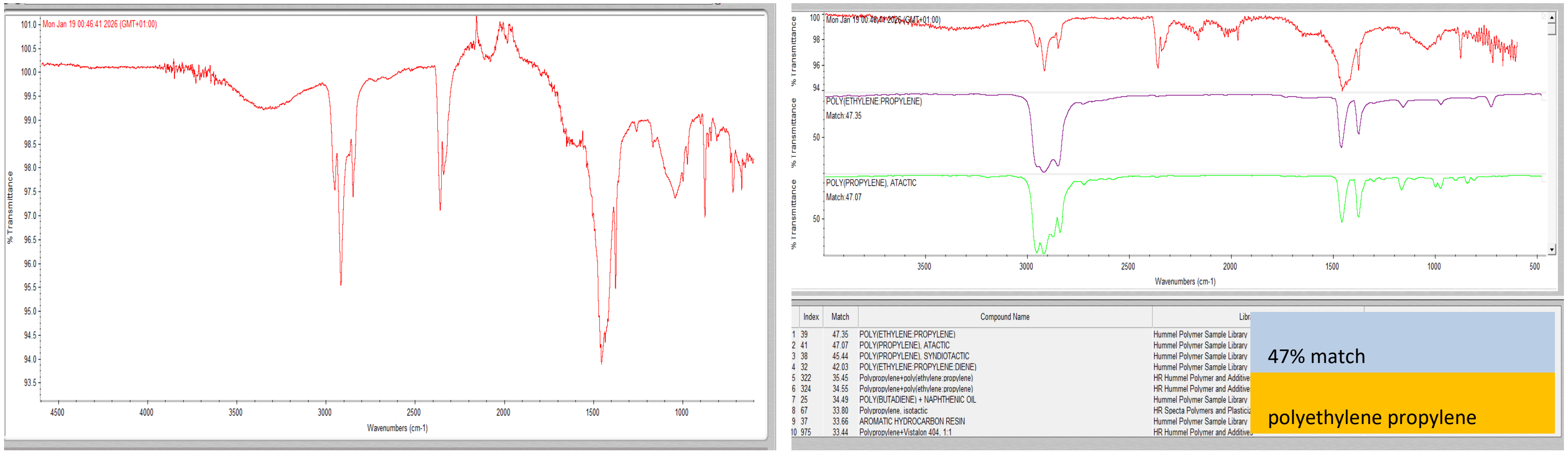


Figure 2 e –

Macro FTIR spectrum cellulose nitrate

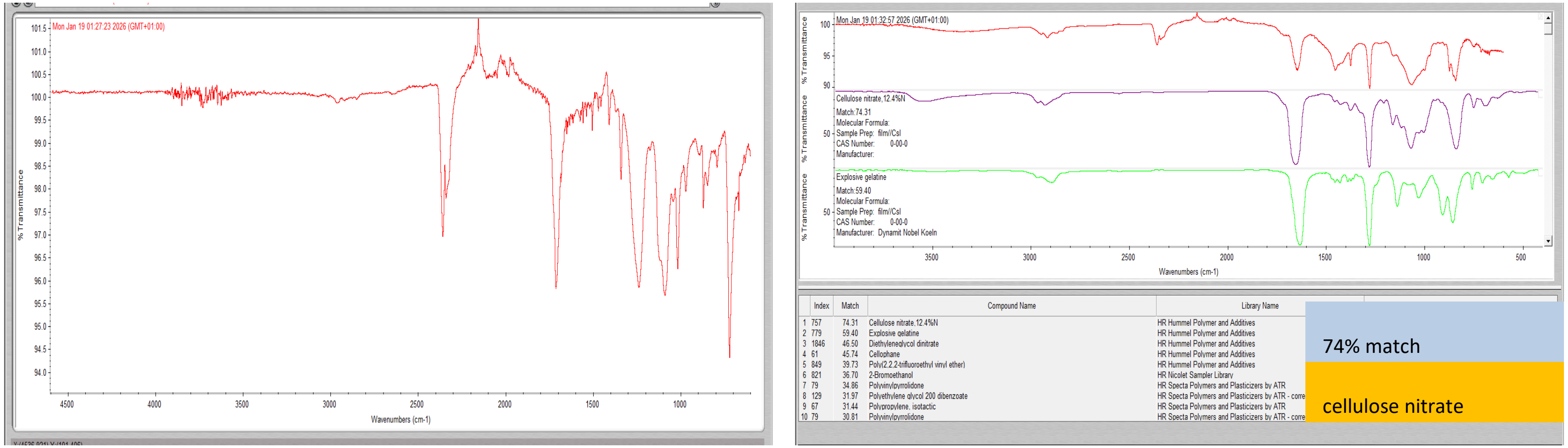


Figure 2 f –

Macro FTIR spectrum polyethylene

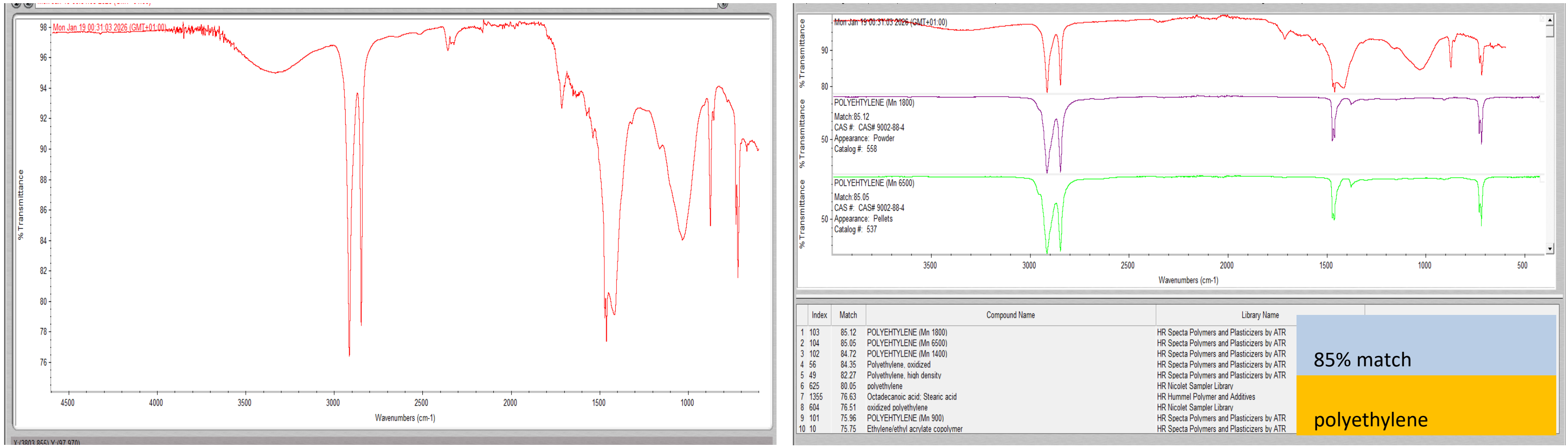


Figure 2 g –

Macro FTIR spectrum polyethylene terephthalate

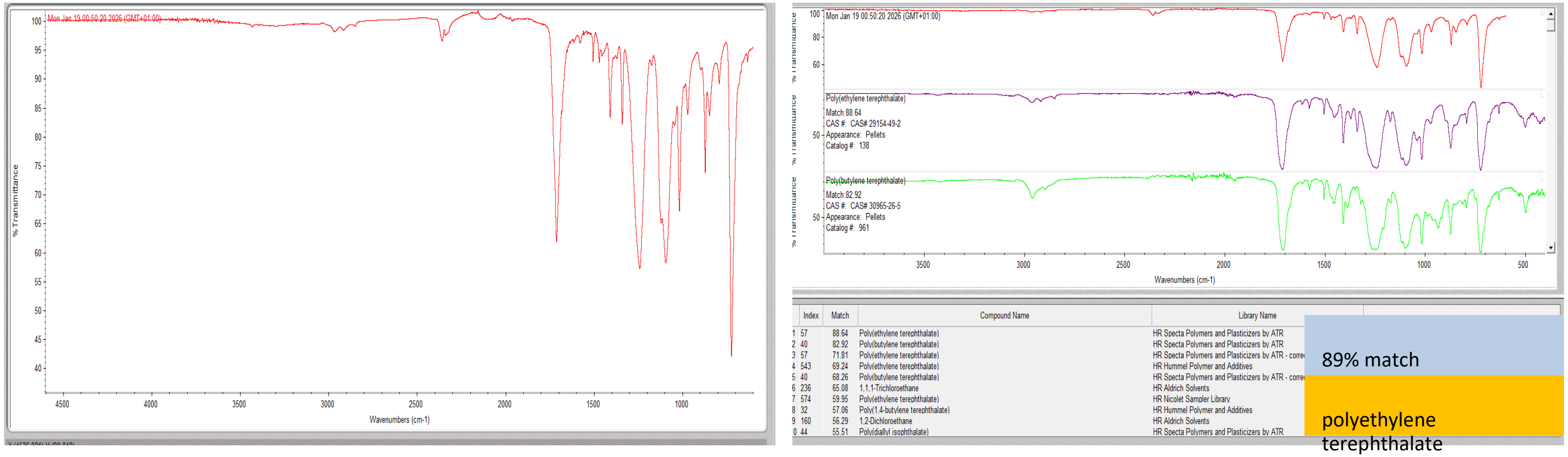
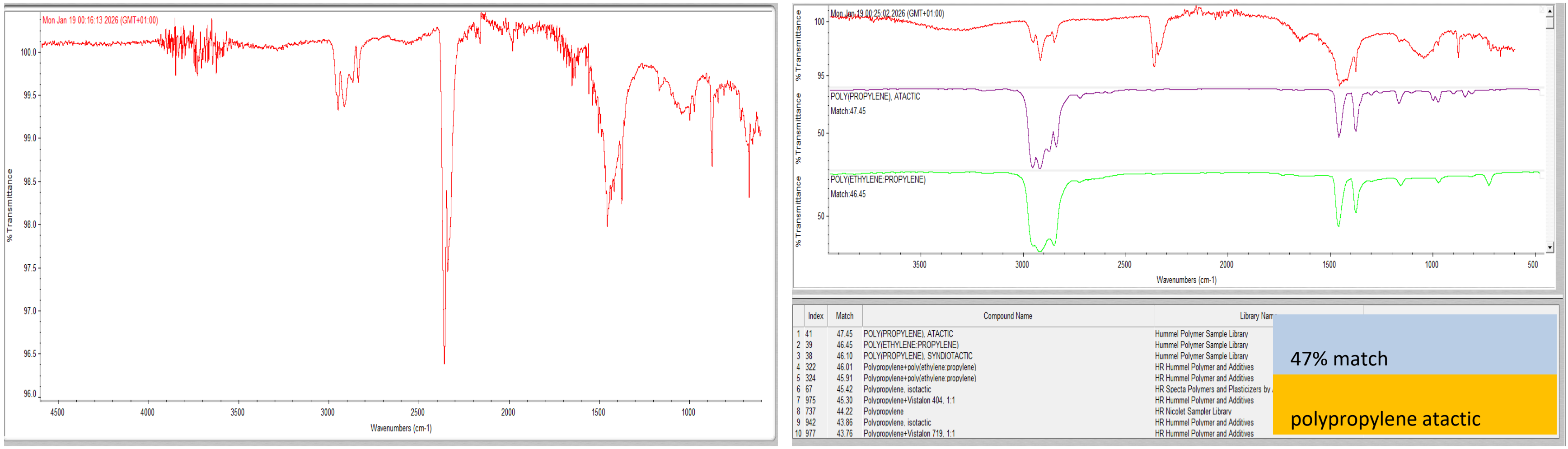


Figure 2 h –

Macro FTIR spectrum polypropylene atactic



Conclusion –

There is direct evidence revealing the issues of waste management in Maldives; essentially no proper waste recycling protocol is followed. Maldives being a collection multiple archipelagos lack any central waste management planning and so far, we did not observe practices encouraging waste sorting and differential treatment of waste as per types. Even previous studies have confirmed that there is contamination of charred and unburnt micro plastic penetrating coastal sediments. In order to manage micro plastic contamination, we would suggest adopting policies and framework discouraging open burning of waste and also creating awareness regarding health hazard and pollution arising from micro plastic contamination, among the locals. The scarcity of targeted studies highlights a critical knowledge gap and underscores the need for multidisciplinary, location specific research which can gauge microplastic sources, distribution, and ecological impacts within Maldivian blue carbon ecosystems.

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Chapter 3
Mangroves of the Maldives: a review of their distribution, diversity, ecological importance and biodiversity of associated flora and fauna

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Abstract

Mangrove forests are one of the most important biological, ecological and economic ecosystems in the world. In the Maldives, they play a crucial role in maintaining coastal biodiversity, providing ecosystem services, such as coastal protection, and supporting livelihoods by providing income and food. Overall, 23 Maldivian islands have at least 1 protected mangrove area. However, knowledge of the mangroves of the Maldives is scarce, scattered and sometimes conflicting. There is a lack of information on a national scale regarding their distribution, diversity, ecological importance and associated biodiversity. The aim of this review is to analyse scientific publications, reports, and online documents on mangroves for the entire Maldivian archipelago to provide the first comprehensive summary of the current state of knowledge of mangroves from a national perspective. This includes the geographical location of mangrove forests, the identity and distribution of mangrove species, ecosystem services, ecological importance and diversity of mangrove-associated flora and fauna. We analysed available information from both the grey literature and scientific publications and found that 14 mangrove species have been documented on 108 islands (9% of all Maldivian islands). Mangroves are mainly concentrated in northern atolls and are associated with diverse flora and fauna. Furthermore, we identified inconsistencies and gaps in the literature and proposed future directions for research. This is crucial for informed decision-making, developing effective conservation strategies and long-term sustainability of mangrove ecosystems.

Introduction

This chapter regarding ecological importance and biodiversity of associated flora and fauna of Maldives is part of a review publication of us and functions as a foundation step. Mangroves are a group of salt-tolerant woody plants that occupy the intertidal zone where seawater and freshwater converge. Their distribution is predominantly restricted to tropical and subtropical latitudes, between 30° north and 30° south (Kandasamy and Bingham 2001; Selvam 2007). These plants are capable of withstanding a range of dynamic and extreme environmental conditions, including high and variable salt concentrations resulting from tidal and precipitation fluctuations, elevated temperatures, strong winds and anaerobic soils, owing to a suite of morphological, biological, ecological, and physiological adaptations (Kandasamy and Bingham 2001; Srikanth et al. 2016). Such adaptations encompass specialised above-ground roots, namely breathing roots or pneumatophores and stilt roots, as well as vivipary and mechanisms of salt exclusion and secretion (Scholander 1968; Shi et al. 2005; Selvam 2007; Srikanth et al. 2016).

Mangrove ecosystems rank among the most productive ecosystems on the planet. They are of considerable biological and ecological importance, supporting numerous terrestrial, estuarine and marine species through habitat provision and food supply. Furthermore, they play a pivotal role in sustaining adjacent coastal marine environments, including coral reefs and seagrass beds, by functioning both as a source and a reservoir of nutrients and sediments (Polidoro et al. 2010; Agardy et al. 2017). Mangrove forests also make a substantial contribution to ecosystem services (Zhang et al. 2018), the economic value of which has been estimated at a minimum of US$1.6 billion annually (Costanza et al. 1997). Among the services they provide are shoreline stabilisation and the prevention of coastal erosion, mitigation of the effects of natural disasters such as tsunamis, food provision (with approximately 80% of global fish catches dependent on mangroves), sequestration of up to 25.5 million tonnes of carbon per year, regulation of energy and nutrient fluxes in the water column, neutralisation of toxins and heavy metals, salt balance regulation, support of recreation and tourism, and the provision of novel compounds relevant to drug discovery, as well as medicines, edible by-products, fuelwood and building materials (Eong 1993; Dahdouh-Guebas et al. 2015; Das and Vincent 2009; Polidoro et al. 2010; Duke et al. 2014; Agardy et al. 2017; Spalding and Parrett 2019; Cerri et al. 2022).

Notwithstanding their ecological and economic significance, mangrove forests are counted among the most threatened ecosystems on Earth, with the overall rate of mangrove loss estimated to be three to five times greater than that of global forests (Duke et al. 2014). It has been estimated that up to 35% of mangrove coverage has been lost since the 1980s (Curnick et al. 2019), with the principal drivers being the overexploitation of resources and the conversion of land for aquaculture, agriculture and urban coastal development (Valiela et al. 2001; Upadhyay et al. 2002; Ellison 2008). Moreover, the pace of decline is accelerating, and the functional disappearance of mangrove forests has been predicted to occur within the next 100 years should current trends persist (Duke et al. 2007; Polidoro et al. 2010). In light of this rapid

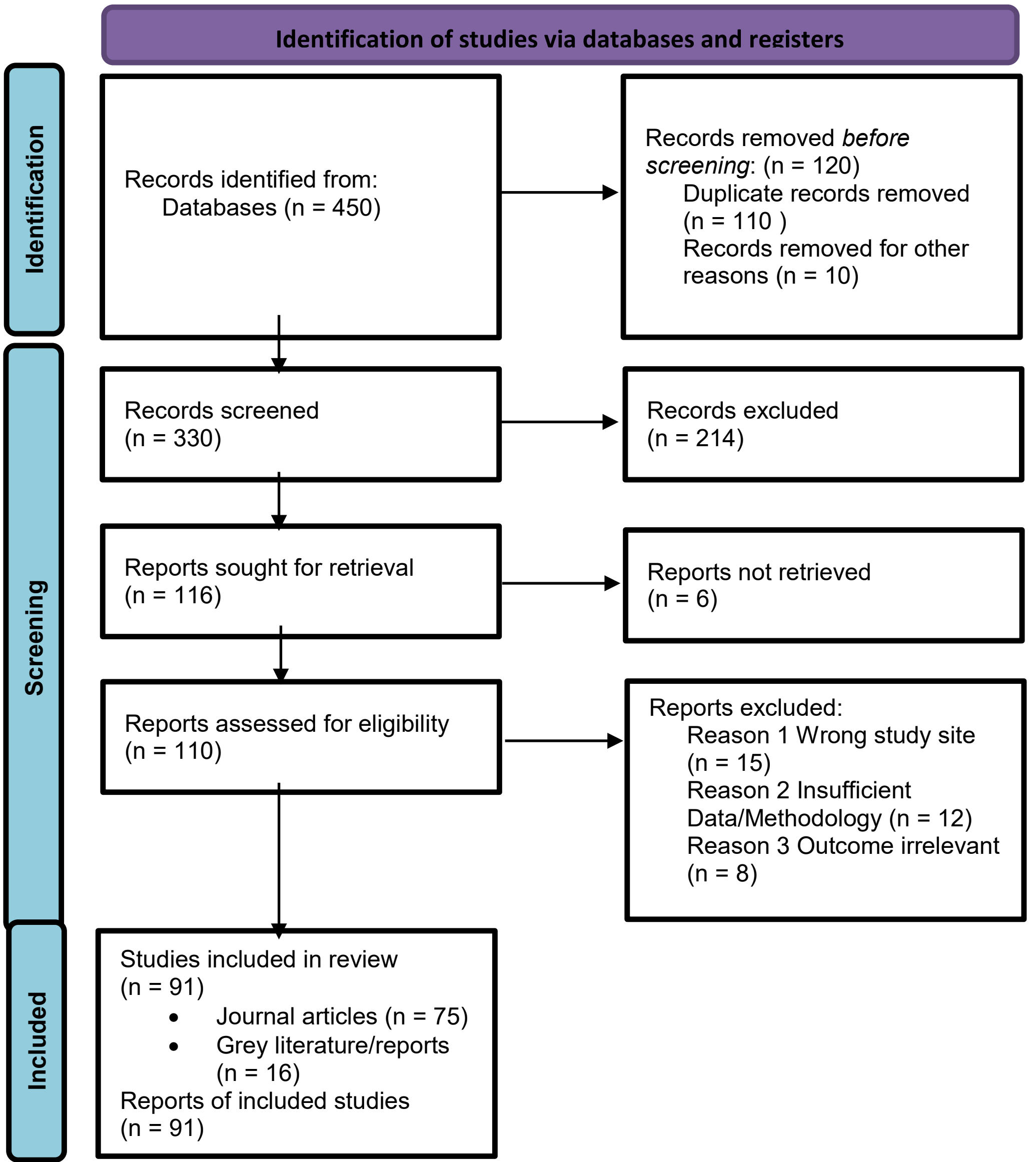
deterioration, and in order to improve efforts at protection and restoration, it is imperative to advance scientific understanding of these ecosystems.
Despite decades of research, mangrove forests remain poorly understood in many parts of the world. In the Maldives, for instance, mangrove ecosystems face mounting anthropogenic pressures, including infrastructure development, improper waste disposal, plastic pollution and land reclamation (Dryden et al. 2020a, b). Yet available information on Maldivian mangroves is both limited and fragmented. A deficit or complete absence of information on an ecosystem can impair scientific understanding, hinder management and conservation efforts, compromise decision-making processes, constrain assessments, reduce stakeholder engagement and result in missed opportunities for sustainable development and conservation. The aim of this thesis chapter is therefore to analyse and synthesise the existing data and information available on the mangroves of the Maldives. The objective is to present, for the first time, a comprehensive summary of the current state of knowledge regarding Maldivian mangrove forests.

Methods

The literature search and selection process for this review followed a systematic approach, as detailed in. To further illustrate the transparency of this process, a PRISMA 2020 flow diagram was developed for this thesis (Figure A).

Figure A

PRISMA 2020 flow diagram illustrating the study selection process. Adapted from The PRISMA 2020 statement: an updated guideline for reporting systematic reviews.



In summary we had the following breakup of literature review.

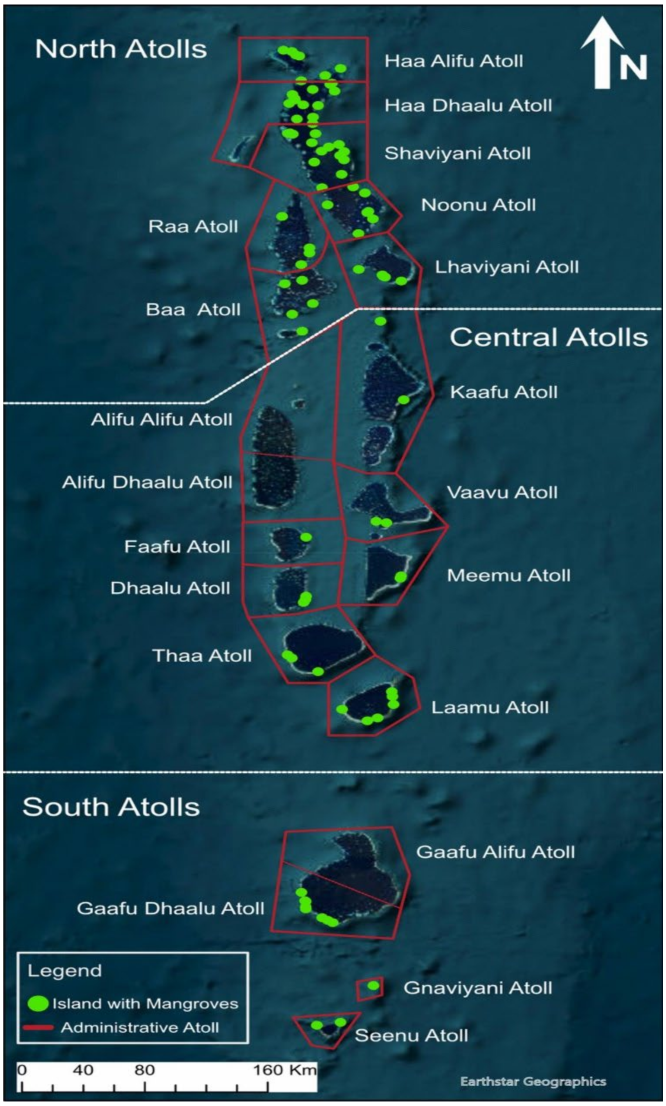
- Final Sources (n=91)
- Scientific Papers: 56
- Reports & Grey Literature: 26 (17 official reports + 9 web/news sources)
- Books/Chapters: 9

We followed a comprehensive review of scientific publications and examination of grey literature, unpublished materials and relevant information from the websites of Maldivian non-governmental organisations (NGOs), as described above. Scientific articles and books were retrieved from online databases, including Google Scholar, Scopus and Web of Science, during the literature review process. The search drew upon combinations of the following terms: 'mangrove', 'Maldives', 'species', 'geographic distribution', 'flora', 'fauna' and 'algae'. The grey literature sources consulted for this study were diverse, encompassing commonly used online platforms such as Google, as well as conventional repositories of information including government documents, NGO websites, policy literature and white papers. Published and unpublished materials were also considered, with care taken to ensure thorough coverage. Grey literature is not easily searchable through conventional databases, which posed a barrier to obtaining up-to-date and comprehensive information about the ecosystem under study. To address this challenge, local government officials and university representatives were contacted for guidance towards relevant documents.

Geomorphology and classification of mangroves in the Maldives

The mangrove forests of the Maldives are distinctive owing to the geomorphological character of the country itself. The Maldives is an archipelago comprising 1,192 islands, divided into 26 geographical atolls, extending 870 km from 7° north to 0.5° south of the equator in the Indian Ocean (Fig. 1 in chapter 3) (Stevens and Froman 2019). Although mangroves are commonly found in estuaries, there are no rivers in the Maldives. Consequently, the morphology of mangrove forests in the Maldivian islands generally consists of small patches of mangrove plants located within closed or semi-enclosed brackish water bodies, known locally as 'kulhi', or in muddy areas referred to locally as 'chasbin'. In general terms, the classification of mangrove forests in the Maldives distinguishes between closed and open systems. Closed systems are further subdivided into lake-based inland mangroves, in which mangrove vegetation surrounds a brackish pond formed within a topographic depression, and marsh-based inland mangroves, in which the vegetation occupies a muddy area. Open systems, by contrast, are subdivided into coastal fringe mangroves, situated on the shoreline and directly exposed to the sea, and bay mangroves, which are subject to daily tidal flushing (Saleem and Nileysha 2003; Shadiya et al. 2016).

Fig 1



Distribution of islands with a reported presence of mangroves (green points) in the Maldives according to our review of the literature. The different atolls are indicated in red boxes (Sources: Esri, DigitalGlobe, GeoEye, i-cubed, USDA FSA, USGS, AEX, Getmapping, Aerogrid, IGN, IGP, swisstopo, and the GIS User Community)

Species diversity of mangroves in the Maldives

Mangrove species are broadly classified as 'true mangrove' species and 'mangrove associates'. True mangrove species are exclusively found in tropical intertidal habitats, whereas mangrove associates are not restricted to these environments but also occur in terrestrial or aquatic habitats (Lugo and Snedaker 1974; Parani et al. 1998; Polidoro et al. 2010). Tomlinson (1986) established strict criteria for classifying a true mangrove, stipulating that it must occur exclusively within the mangrove environment, possess morphological specialisations including aerial roots, gas exchange mechanisms and viviparity, have a physiological mechanism for salt exclusion and/or excretion, and be taxonomically isolated from terrestrial relatives (Wang et al. 2010; Kandasamy and Bingham 2001). Furthermore, Duke (1992) defined a true mangrove species as a tree, shrub, palm or ground fern exceeding 0.5 m in height, which normally grows above mean sea level in the intertidal zone of coastal or estuarine environments (Polidoro et al. 2010). Kandasamy and Bingham (2001) reviewed the historical classifications and recognised 65 true mangrove species in 22 genera and 16 families on a global basis. However, scientific consensus has not been reached for certain species, and their classification remains subject to debate. Wang et al. (2010) referred to these as 'controversial' species and distinguished them on the basis of leaf traits and osmotic properties, categorising as 'controversial' those species for which no accepted classification consensus exists regarding their inclusion within the group of true mangroves or the group of mangrove associates.

In the Maldives, the precise number of mangrove species present in the archipelago varies across the literature, and limited information is available regarding the methodologies employed for species identification. In 2007, the Ministry of Fisheries, Agriculture and Marine Resources of the Maldives documented 13 mangrove species along with their respective characteristics, including their status, description, uses, ecology, propagation and management (Selvam 2007). The species recorded were: Avicennia marina, Bruguiera cylindrica, Bruguiera gymnorrhiza, Bruguiera sexangula, Ceriops tagal, Excoecaria agallocha, Heritiera littoralis, Lumnitzera racemosa, Pemphis acidula, Rhizophora apiculata, Rhizophora mucronata, Sonneratia caseolaris and Xylocarpus rumphii. In addition to these 13 species, Dryden et al. (2020a, b) subsequently reported the presence of Bruguiera hainesii in the Maldives, a hybrid species arising from B. cylindrica and B. gymnorrhiza (Ono et al. 2016). Nevertheless, other sources in the literature report differing species compositions. For example, in a study by Saleem and Nileysha (2003), 14 mangrove species were reported. In comparison with Selvam (2007), Saleem and Nileysha (2003) specifically documented three species of Rhizophora rather than two, indicated the occurrence of Xylocarpus moluccensis in place of Xylocarpus rumphii, and noted the presence of Derris heterophylla. Furthermore, they classified Pemphis acidula as a mangrove associate rather than a true mangrove. The non-governmental organisation (NGO) Bluepeace (2007) reported 13 mangrove species, including the mangrove fern Acrostichum aureum, while Xylocarpus spp. and P. acidula were not indicated. Additionally, Dhunya et al. (2017) and Sivakumar et al. (2018) reported 14 and 15 mangrove species respectively, although species names were not specified in either study.

Acrostichum aureum, Derris heterophylla and Xylocarpus rumphii are not considered mangrove species according to Kandasamy and Bingham (2001). Xylocarpus rumphii is a rare non-mangrove plant (Guo et al. 2018), while Acrostichum aureum and Derris heterophylla are classified as mangrove associate species (Das et al. 2002; Mukherjee et al. 2006; Wang et al. 2010). On the basis of the above, it is proposed that 14 mangrove species are present in the Maldives: A. marina, B. cylindrica, B. gymnorrhiza, B. hainesii, B. sexangula, C. tagal, E. agallocha, H. littoralis, L. racemosa, P. acidula, R. apiculata, R. mucronata, S. caseolaris and X. moluccensis, of which three are considered controversial species — namely E. agallocha, H. littoralis and P. acidula (Wang et al. 2010). The mangrove species of the Maldives are presented in Table 1 in chapter 3, together with their vulnerability status according to the International Union for Conservation of Nature (IUCN) Red List of Threatened Species (Polidoro et al. 2010), their family name (Polidoro et al. 2010), common name and local name in Dhivehi (Selvam 2007; Save Maldives 2020; Ali 2020).

Several factors may account for the inconsistencies in the number and identity of species reported across studies conducted in the Maldives:

- Ambiguity in the classification of 'controversial' species as either true mangrove or mangrove associate, with the same species receiving different designations in different sources.
- Variation in taxonomic expertise among researchers may affect the precision of species identification and contribute to discrepancies in reported species numbers.
- Different studies may apply different taxonomic frameworks or classification schemes, and advances in species identification techniques or taxonomic revisions may introduce further inconsistencies between earlier and more recent studies.
- Studies may have focused on specific geographical regions or sampling sites, resulting in variation in the species compositions recorded; differences in sampling effort, spatial coverage or seasonal variation in fieldwork may also contribute to such discrepancies.
- Mangrove ecosystems are inherently dynamic and subject to natural variation and ecological succession over time, meaning that studies conducted at different periods may capture different stages of ecosystem development, thereby producing variation in observed species diversity.

Table 1 Mangrove species of the Maldives

Scientific name	Family	Common name	Local name (Dhivehi)	IUCN Red List Category	Classification
Avicennia marina	Acanthaceae	Grey mangrove	Baru	LC	TS
Bruguiera cylindrica	Rhizophoraceae	Small-leafed orange mangrove	Kan’doo	LC	TS
Bruguiera gymnorrhiza	Rhizophoraceae	Oriental mangrove or large-leafed mangrove	Bodukandoo or Bodavaki	LC	TS
Bruguiera hainesii	Rhizophoraceae	Eye of the crocodile	Kelavaki, or bodukandoo or makandoo	CR	TS
Bruguiera sexangula	Rhizophoraceae	Upriver orange mangrove	Bodavaki	LC	TS
Ceriops tagal	Rhizophoraceae	Yellow mangrove	Karamana	LC	TS
Excoecaria agallocha	Euphorbiaceae	Blind-your eye mangrove or milky mangrove	Thela	LC	CS
Heritiera littoralis	Malvaceae	Looking-glass mangrove	Kaharuvah	LC	CS
Lumnitzera racemosa	Combretaceae	Black mangrove	Burevi	LC	TS
Rhizophora apiculata	Rhizophoraceae	Tall-stilt mangrove	Thakafathi	LC	TS
Rhizophora mucronata	Rhizophoraceae	Red mangrove	Ran’doo	LC	TS
Sonneratia caseolaris	Lythraceae	Mangrove apple	Kuhlhavah	LC	TS
Pemphis acidula	Lythraceae	Ironwood	Kuredhi	LC	CS
Xylocarpus moluccensis	Meliaceae	Cannonball tree or puzzlenut tree	Marugas	LC	TS

LC least concern, CR critically endangered, CS controversial species, TS true mangrove species

The possible reasons for the inconsistencies in the number and identity of species reported in the different studies conducted in the Maldives, include:

(i) unclear classification between true or associate mangrove species for so-called ‘controversial’ species. The same species can be considered true mangrove or mangrove associate according to different sources.

(ii) Differences in taxonomic expertise among researchers can influence the precision of species identification and lead to variations in reported species numbers.

(iii) Different studies may employ different taxonomic approaches or classifications, leading to variations in species identification and categorisation. Taxonomic revisions or advances in species identification techniques can also result in discrepancies between studies.

(iv) Studies might have focused on specific regions or sampling sites, resulting in variations in the observed species composition. Differences in sampling methodologies, such as sampling effort, spatial coverage or seasonal variations, can also contribute to discrepancies.

(v) Mangrove ecosystems are dynamic and can undergo natural variations and ecological succession over time. Different studies carried out at different periods may capture different stages of ecosystem development, leading to variations in observed species diversity.

Distribution of mangroves in the Maldives

Drawing on data from multiple sources, mangroves have been reported on numerous islands throughout the Maldivian archipelago (Fig. 1). The total area of wetlands and mangroves in the Maldives is estimated to be approximately 7.39 km² (Ministry of Environment and Energy [MEE] 2015). Available reports indicate that not all mangrove species can be found on a single island, which may reflect the relatively small size of both the mangrove forests and the Maldivian islands themselves, as these may be insufficient to sustain high species diversity. The distribution of individual species is also subject to environmental variation; from north to south across the archipelago, environmental conditions differ markedly and restrict certain species to specific geographical localities (Shazra et al. 2008). The dominant mangrove species are B. cylindrica, B. gymnorrhiza, L. racemosa and P. acidula, while others, including H. littoralis and B. hainesii, are occasional or rare (Selvam 2007). Saleem and Nileysha (2003) reported the occurrence of mangroves on at least 150 islands. Based on the available literature and the website of the Maldivian Environmental Protection Agency (EPA) (https://en.epa.gov.mv/), the following sections summarise the mangrove species present in each of the 20 administrative atolls, along with their health and conservation status. The nomenclature of atolls and islands, and the division of the archipelago into northern, central and southern atolls, follows that of Fallati et al. (2017).

Haa Alifu Atoll

Haa Alifu Atoll, officially designated as Thiladhunmathi Uthuruburi (or Northern Thiladhunmathi Atoll), is the third largest atoll by land area in the northernmost administrative division of the Maldives. The mangrove forests on the islands of Baarah, Kelaa and Gallandhoo were declared protected areas by the Ministry of Environment of the Maldives (Protected Area Announcements (IUL) 438-ENV-438-2018-322 and (IUL) 438-ENV-438-2019-150). On Baarah Island, several wetland areas exist, though only some are afforded protection. Bluepeace conducted a comprehensive baseline study of the ten wetlands identified on the island, evaluating the presence and distribution of mangroves within each water body. The wetlands situated within the protected area harbour the greatest mangrove diversity on the island, while the two unprotected wetlands support no mangroves. The most abundant species recorded was B. cylindrica, with additional species including C. tagal, E. agallocha, L. racemosa, R. apiculata and R. mucronata also documented (Bluepeace 2007). Shazra et al. (2008) subsequently identified P. acidula in the atoll. The Kelaa mangrove forest is of considerable ecological importance, owing to the diversity and abundance of its mangrove trees, which support various species of birds and invertebrates (Dryden et al. 2020a, b). The Kelaa ecosystem encompasses one of the most extensive stands of B. cylindrica (locally known as 'kandoofaa') in the Maldives (IDEAS 2017). Additional species documented on the island include B. gymnorrhiza, L. racemosa, S. caseolaris, Acrostichum aureum (Dryden et al. 2020a, b) and R. mucronata (IDEAS 2017). Dryden et al. (2020a, b) also reported the presence of four Bruguiera hainesii trees on Kelaa, representing the first confirmed record of this species in the Maldives. B. hainesii is classified as 'critically endangered' according to the IUCN Red List of Threatened Species, with fewer than 250 mature trees estimated to remain globally (Polidoro et al. 2010). Mangroves have also been reported on the islands of Berinmadhoo, Maafahi, Madulu, Maarandhoo, Muraidhoo, Naridhoo, Uligan and Vangaaru (EPA Sensitive

Areas 2015), and a strip of mangrove vegetation has been documented on the south-eastern side of Thakandhoo Island (Naeem 2008). Bluepeace (2015) additionally reported five small wetland areas on Filladhoo Island, with mangroves present at the perimeter of three of these water bodies. Species recorded included B. cylindrica, L. racemosa and R. apiculata (Bluepeace 2015), and MEE (2015) further reported the presence of B. gymnorrhiza on the island. Dhapparu Island, which is physically connected to Filladhoo by a strip of land and has recently been placed under its administrative jurisdiction, has a water body bordered by R. mucronata and R. apiculata (Bluepeace 2015). In summary, based on the information from the different sources, mangroves have been reported on 13 islands of Haa Alifu Atoll (Fig. 2). In total, ten species have been reported, including *A. aureum*, *B. cylindrica*, *B. gymnorrhiza*, *B. hainesii*, *C. tagal*, *E. agallocha*, *L. racemosa*, *R. apiculata*, *R. mucronata* and *S. caseolaris* (Table S1 in chapter 3, Supplementary Information).

Fig 2

Geographic location of islands with a reported presence of mangroves in Haa Alifu Atoll and Haa Dhaalu Atoll (Sources: Esri, DigitalGlobe, GeoEye, i-cubed, USDA FSA, USGS, AEX, Getmapping, Aerogrid, IGN, IGP, swisstopo, and the GIS User Community)

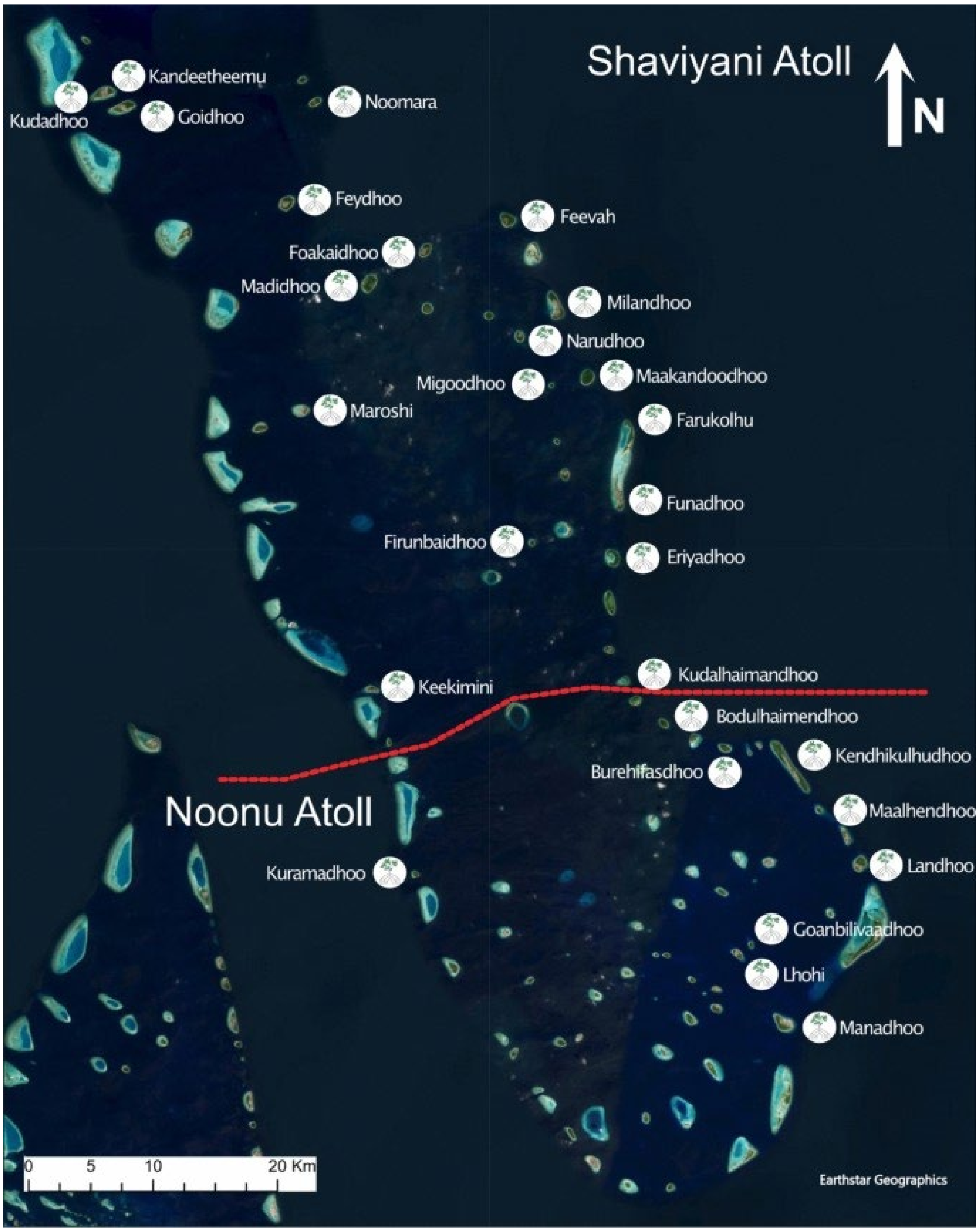
Haa Dhaalu Atoll

Haa Dhaalu Atoll, officially known as Thiladhunmathi Dhekunuburi (or Southern Thiladhunmathi Atoll), encompasses several islands with documented mangrove presence. Among these, the mangrove forests of Keylakunu and Neykurendhoo have been designated as protected areas (Protected Area Announcement (IUL) 438-ENV-438-2018-322). The Island of Keylakunu is protected in its entirety, with the mangrove forest situated within a wetland area in the south-eastern part of the island, comprising two separate water bodies. Available reports exhibit discrepancies with respect to species diversity on this island. Bluepeace (2015) reported that the forest is dominated by B. cylindrica, with additional species including R. apiculata, E. agallocha and A. marina also recorded (Bluepeace 2015). In contrast, the IUCN (2020) reported the presence of B. cylindrica, R. mucronata (documented in the area of greatest water depth), A. marina, P. acidula and a single individual of S. caseolaris (located at the northern end of the forest). The second protected mangrove area comprises the wetland with two water bodies on the eastern side of Neykurendhoo. Bluepeace (2015) reported B. cylindrica to be abundant in this area, with additional occurrences of E. agallocha, L. racemosa and X. moluccensis observed. The EPA has documented mangroves on the islands of Faridhoo, Hanimaadhoo, Hirinaidhoo, Kulhudhuffushi, Kumundhoo, Kurinbee, Maavaidhoo, Naagoashi, Naivaadhoo, Nellaidhoo, Nolhivaran and Vaikaradhoo (EPA Sensitive Areas 2015), though for the majority of these islands, no further details are available in the literature. Kulhudhuffushi, the capital island of Haa Dhaalu Atoll, is the most thoroughly documented. The island derives its name from its mangrove forest (Jaufar 2021), and the dominant species reported are L. racemosa, B. cylindrica and R. mucronata (IUCN 2019), with S. caseolaris also observed (MEE 2015). Curnick et al. (2019) estimated that airport construction may have resulted in the destruction of up to 70% of the island's mangrove forests. The presence of a mangrove forest has furthermore been reported in a wetland on Nolhivaranfaru (IDEAS 2018a). This wetland is characterised by B. cylindrica trees along the perimeter of a water body, accompanied by patches of P. acidula, forming a belt between the lake and the sea that is subject to tidal inundation. Two X. moluccensis trees have been recorded near the beach, and R. mucronata has been documented in high abundance around a smaller adjacent water body (IDEAS 2018a). On Hirimaradhoo, a wetland area has been documented and listed as a sensitive area by the EPA (EPA Sensitive Areas 2015), with mangrove presence confirmed by Saleem and Nileysha (2003). This forest is classified as an embayment mangrove. In total, mangroves have been reported on 16 islands within Haa Dhaalu Atoll (Fig. 2), with nine species documented: A. marina, B. cylindrica, E. agallocha, L. racemosa, P. acidula, R. apiculata, R. mucronata, S. caseolaris and X. moluccensis (Table S1 in chapter 3, Supplementary Information).

Shaviyani Atoll

Shaviyani Atoll, officially designated as Miladhunmadulu Uthuruburi (or Northern Miladhunmadulu Atoll), contains the entire Island of Farukolhu as a protected area (Protected Area Announcement (IUL)438-ENV-438-2018-262). In the south of the island, there is an extensive mangrove bay dominated by C. tagal, with additional species including P. acidula and R. mucronata (IUCN 2020). C. tagal is a relatively rare species in the Maldives, found only along the margins of this large lagoon and on the island immediately to its south (Selvam 2007). In the northern part of Farukolhu Island, there are several brackish-water ponds with patches of R. mucronata. Evidence reported by the IUCN (2020) indicates that the mangroves in this area are experiencing a decline in health. Other islands with a documented presence of mangroves include Eriyadhoo, Feevah, Feydhoo, Firunbaidhoo, Foakaidhoo, Funadhoo, Gooidhoo, Kandeetheemu, Kudadhoo, Kudalhaimendhoo, Keekimini, Maakandoodhoo, Madidhoo, Maroshi, Migoodhoo, Milandhoo, Narudhoo and Noomaraa (EPA Sensitive Areas 2015). Detailed information from the literature has been found only for Milandhoo and Maakandoodhoo. On Milandhoo, a water body is present with R. mucronata, L. racemosa and P. acidula documented along its perimeter (IDEAS 2018b). Maakandoodhoo has a large central pond and wetland areas in the north. Although no channels directly connect the sea to these water bodies, the marshy terrain is indicative of a high water table. The mangrove species L. racemosa and B. cylindrica are dominant at both sites, with the mangrove forest surrounding the central pond in notably better condition than that found in the north. Additional species documented include A. aureum, E. agallocha and R. apiculata (IUCN 2020). In summary, mangroves have been reported on 19 islands of Shaviyani Atoll (Fig. 3). In total, seven species have been documented, namely A. aureum, B. cylindrica, C. tagal, E. agallocha, L. racemosa, P. acidula and R. mucronata (Table S1 in chapter 3, Supplementary Information).

Fig 3



Geographic location of islands with a reported presence of mangroves in Shaviyani Atoll and Noonu Atoll (Sources: Esri, DigitalGlobe, GeoEye, i-cubed, USDA FSA, USGS, AEX, Getmapping, Aerogrid, IGN, IGP, swisstopo, and the GIS User Community)

Noonu Atoll

Noonu Atoll, officially known as the Southern Miladhunmadulu Atoll (or Miladhunmadulu Dhekunuburi), encompasses two protected mangrove ecosystems: Bodulhaimendhoo and Kendhikulhudhoo (Protected Area Announcement (IUL)438-ENV-438-2019-150). Bodulhaimendhoo is a small circular island with vegetation distributed around the perimeter of a large brackish water body. The mangrove species R. apiculata and P. acidula have been recorded there (IUCN 2020). Kendhikulhudhoo has an extensive mangrove forest along the eastern edge of the island, consisting of water bodies that have been artificially connected to the sea to create a flow of water for aquaculture activities. Bruguiera spp., Rhizophora spp. and P. acidula have been reported on the island (IUCN 2020). Mangroves have additionally been documented on the islands of Burehifasdhoo, Goanbilivaadhoo, Kandoodhoo, Kuramaadhoo, Landhoo, Lhohi and Manadhoo (EPA Sensitive Areas 2015). In the literature, a further record of mangroves on Landhoo was documented by Kathiresan and Rajendran (2005). Additionally, Nishan (2010) reported the presence of C. tagal, E. agallocha and R. apiculata on the Island of Maalhendhoo. In total, mangroves have been reported on ten islands of Noonu Atoll (Fig. 3). Seven species have been documented, including Bruguiera spp., C. tagal, E. agallocha, P. acidula, R. apiculata and Rhizophora sp. (Table S1 in chapter 3, Supplementary Information).

Lhaviyani Atoll

Lhaviyani Atoll, officially known as Faadhippolhu Atoll, contains two protected areas on Dhiffushimaadhoo and Maakoa, respectively (Protected Area Announcement (IUL)438-ENV-438-2020-179). Dhiffushimaadhoo is an uninhabited island that represents one of the most significant natural heritage sites in the Maldives. It was originally formed from four separate islands — Dhiffushi, Maidhoo, Sehlhifushi and Hiriyadhoo — and continues to evolve geomorphologically (Bluepeace 2008). The island includes an extensive bay area in the Lhohifushi-Hithaadhoo region, where P. acidula is the dominant vegetation and H. littoralis has also been reported (IUCN 2020). Maakoa is likewise uninhabited and falls under the jurisdiction of the Maldives National Defence Force (MNDF) owing to its proximity to Maafilaafushi Island, which hosts the northern MNDF military base. P. acidula is reported as the dominant species, and B. cylindrica has been documented around the large pond found on the island (IUCN 2020). The islands of Faadhoo, Kanifushi, Lhohi, Lhossalafushi, Thilamaafushi and Varihuraa are listed as sensitive areas owing to the presence of mangroves (EPA Sensitive Areas 2015). On Lholi, a small patch of mangroves has been reported on the south-western side of the island, comprising B. cylindrica (Jameel and Faiz 2016). In summary, mangroves have been reported on eight islands of Lhaviyani Atoll (Fig. 4). Three species have been documented, including B. cylindrica, H. littoralis and P. acidula (Table S1 in chapter 3, Supplementary Information).

Fig 4

Geographic location of islands with a reported presence of mangroves in Lhaviyani Atoll (Sources: Esri, DigitalGlobe, GeoEye, i-cubed, USDA FSA, USGS, AEX, Getmapping, Aerogrid, IGN, IGP, swisstopo, and the GIS User Community)

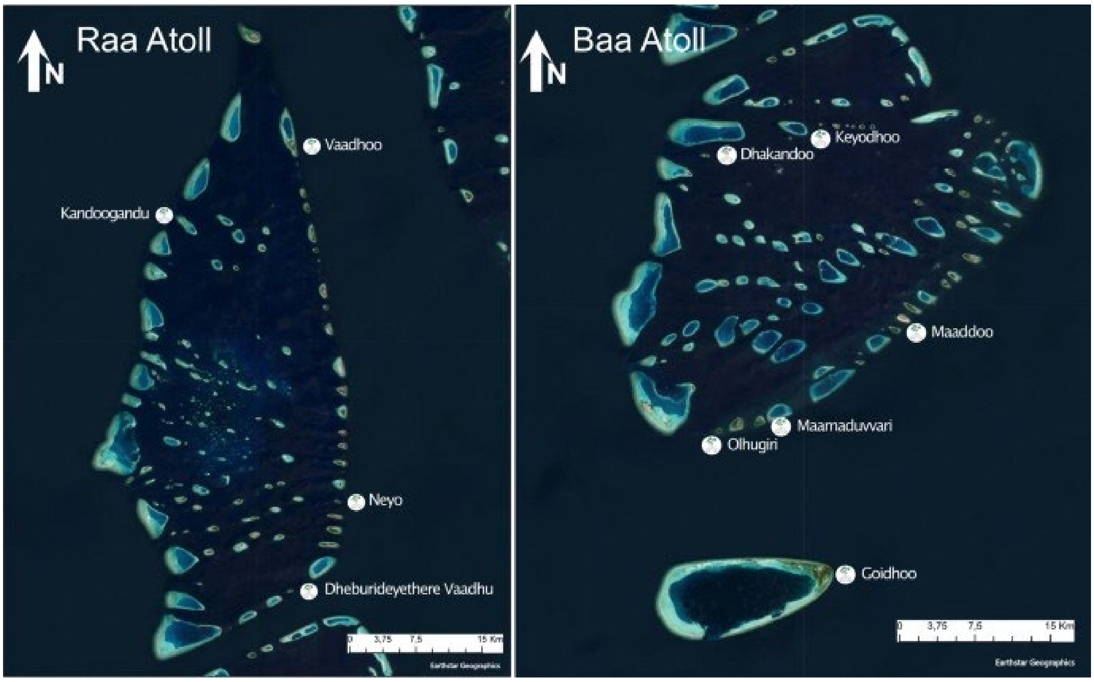
Raa Atoll

Raa Atoll, officially known as Northern Maalhosmadulu Atoll (or Maalhosmadulu Uthuruburi), has four islands with a reported presence of mangroves: Dheburidheythereyvaadhoo, Kan'doogan'du, Neyo and Vaadhoo (EPA Sensitive Areas 2015) (Fig. 5). No information on the identity of the species present in this atoll has been found in the literature.

Baa Atoll

Baa Atoll, officially known as Southern Maalhosmadulu Atoll (or Maalhosmadulu Dhekunuburi), has been designated a UNESCO World Biosphere Reserve since 2011. The presence of mangroves has been documented on the islands of Goidhoo, Olhugiri, Maamaduvvari and Keyodhoo (Bers 2005). The mangrove forests of Goidhoo and Olhugiri are classified as protected areas (Protected Area Announcements (IUL)138-FS2-1-2011-35 and Environment Law 174-AB1/2006/13, respectively). The most common mangrove species in Baa Atoll are B. cylindrica, R. mucronata and P. acidula, though A. marina, E. agallocha and S. caseolaris have also been reported (Bers 2005). The EPA has additionally documented the presence of mangroves on Dhakandoo, Keyodhoo, Maamaduvvari and Maddoo (EPA Sensitive Areas 2015). In summary, mangroves have been reported on six islands of Baa Atoll (Fig. 5). Six species have been recorded, namely A. marina, B. cylindrica, R. mucronata, P. acidula, E. agallocha and S. caseolaris.

Fig 5

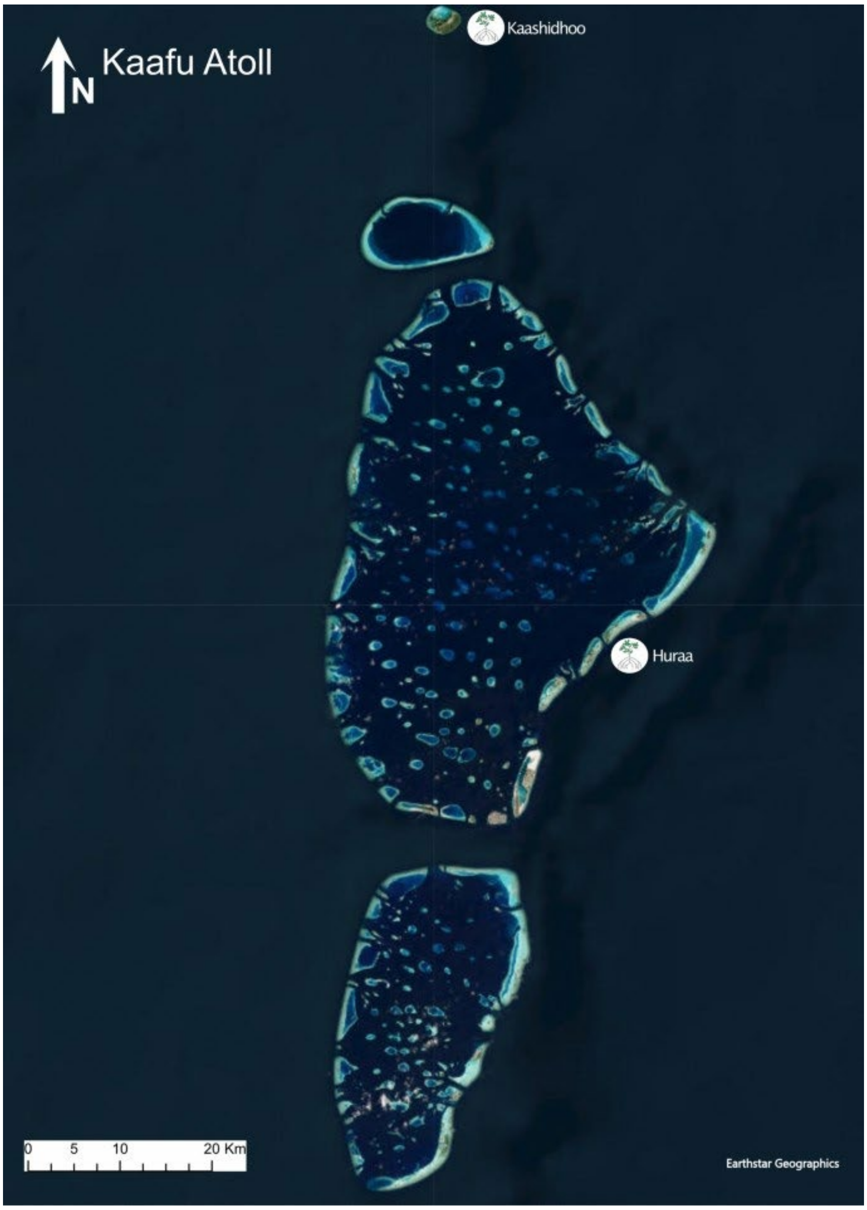


Geographic location of islands with a reported presence of mangroves in Raa Atoll and Baa Atoll (Sources: Esri, DigitalGlobe, GeoEye, i-cubed, USDA FSA, USGS, AEX, Getmapping, Aerogrid, IGN, IGP, swisstopo, and the GIS User Community)

Kaafu Atoll

Kaafu Atoll, officially known as Malé Atoll or Malé Atolhu, comprises four geographical atolls: Kaashidhoo, Gaafaru, North Malé Atoll and South Malé Atoll. Only two mangrove forests have been documented within Kaafu Atoll, located on the islands of Kaashidhoo and Huraa (Protected Area Announcements (IUL)438-ENV-438-2021-24 and Environment Law 174-AB1/2006/13, respectively), both of which are protected areas. The presence of mangroves on Kaashidhoo Island was reported by Kathiresan and Rajendran (2005). The Huraa Mangrove Nature Reserve covers approximately 9 ha on the western side of the island, and 4,000 mature mangrove trees belonging to four species have been documented: B. cylindrica, which is the most abundant, B. gymnorrhiza, R. apiculata, and the less common R. mucronata (MEE 2015; Shadiya et al. 2016). In summary, mangroves have been reported on two islands of Kaafu Atoll (Fig. 6). Four species have been documented, namely B. cylindrica, B. gymnorrhiza, R. apiculata and R. mucronata (Table S1 in chapter 3, Supplementary Information).

Fig 6



Geographic location of islands with a reported presence of mangroves in Kaafu Atoll (Sources: Esri, DigitalGlobe, GeoEye, i-cubed, USDA FSA, USGS, AEX, Getmapping, Aerogrid, IGN, IGP, swisstopo, and the GIS User Community)

Northern Ari Atoll and Southern Ari Atoll

In the Northern Ari Atoll, officially known as Alifu Alifu Atoll (or Ari Atholhu Uthuruburi), no mangroves have been reported. Similarly, in the Southern Ari Atoll, officially known as Alifu Dhaalu Atoll (or Atholhu Dhekunuburi), no mangroves have been documented.

Vaavu Atoll

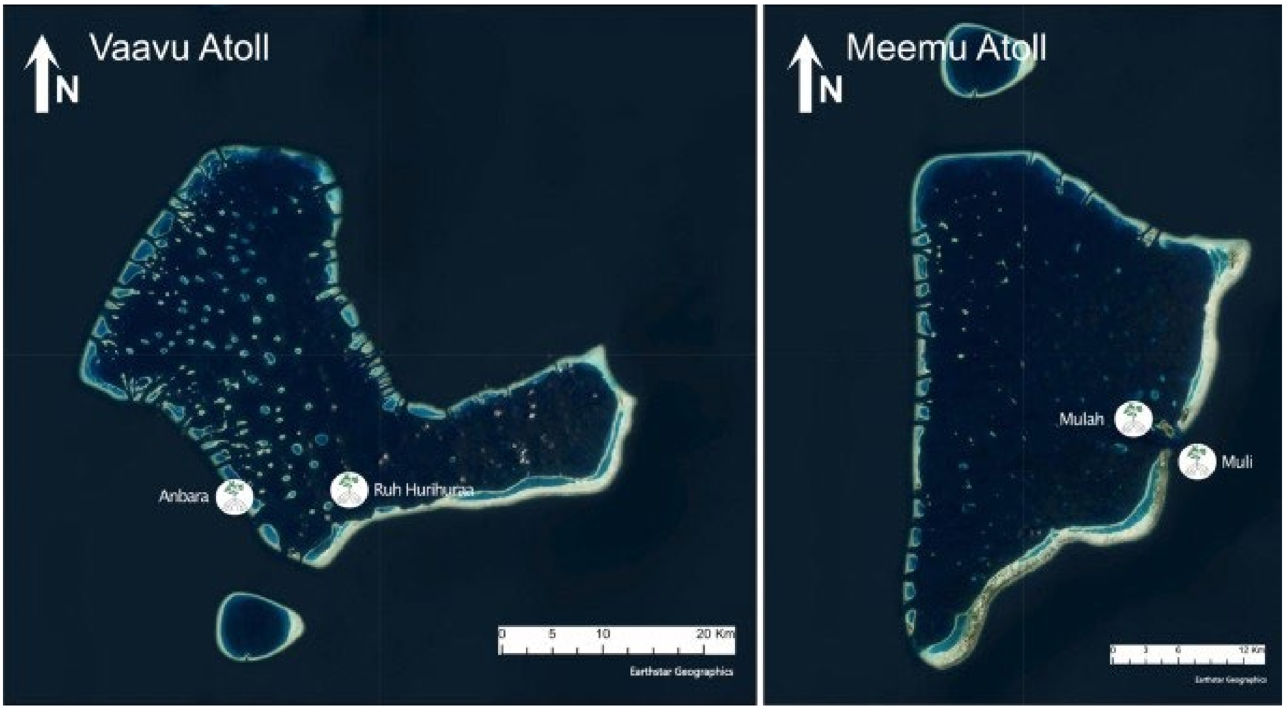
Vaavu Atoll, officially known as Felidhu Atoll, includes only two islands with recorded mangroves: Run Hurihuraa and Anbaraa (EPA Sensitive Areas 2015) (Fig. 7). No study has reported the identity of the species present.

Meemu Atoll

In Meemu Atoll, officially known as Mulaku Atoll, mangroves have been documented on only two islands:

Mulah and Muli (EPA Sensitive Areas 2015) (Fig. 7). No study has reported the identity of the species present.

Fig 7



Geographic location of islands with a reported presence of mangroves in Vaavu Atoll and Meemu Atoll (Sources: Esri, DigitalGlobe, GeoEye, i-cubed, USDA FSA, USGS, AEX, Getmapping, Aerogrid, IGN, IGP, swisstopo, and the GIS User Community)

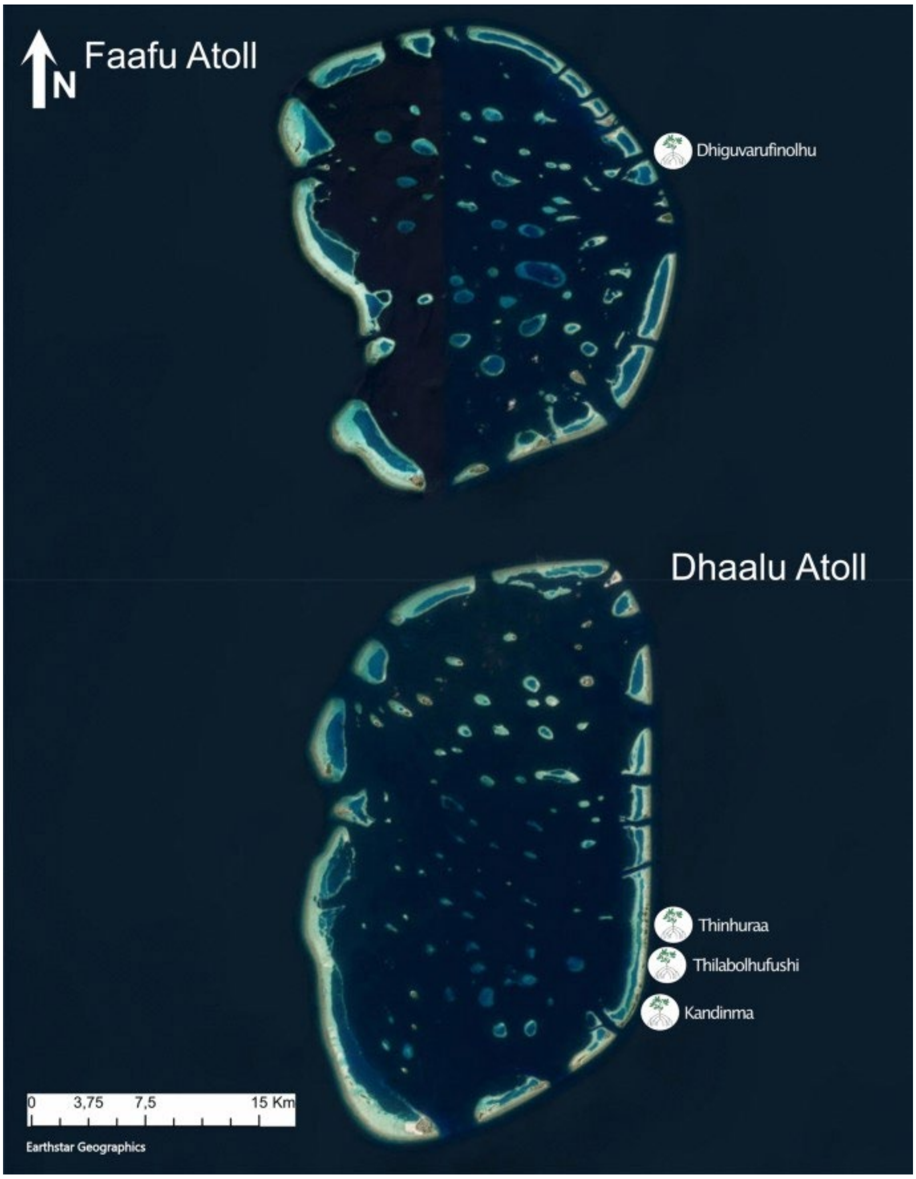
Faafu Atoll

In Faafu Atoll, officially known as Nilandhe Atholhu Uthuruburi (or Northern Nilandhe Atoll), mangroves have been reported only on the Island of Dhiguvarufinolhu (EPA Sensitive Areas 2015) (Fig. 8). The mangrove species recorded on the island are Bruguiera cylindrica and Pemphis acidula (Dryden et al. 2021).

Dhaalu Atoll

Dhaalu Atoll, officially known as Nilandhe Atholhu Dhekunuburi (or Southern Nilandhe Atoll), has three islands with a reported presence of mangroves: Kan'dinma, Thilabolhufushi and Thinhuraa (EPA Sensitive Areas 2015) (Fig. 8). No study has reported the identity of the species present.

Fig 8

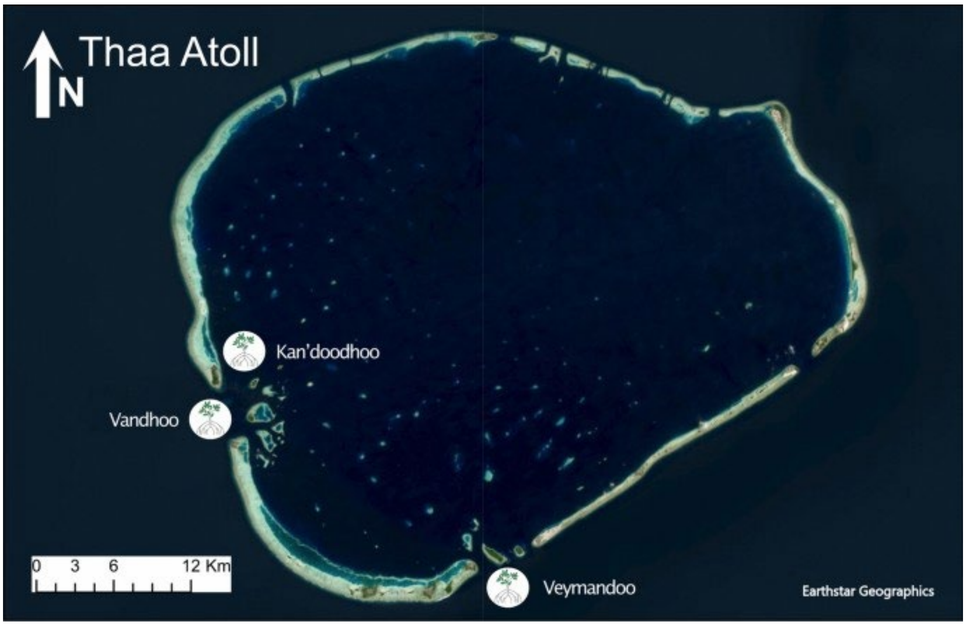


Geographic location of islands with a reported presence of mangroves in Faafu Atoll and Dhaalu Atoll (Sources: Esri, DigitalGlobe, GeoEye, i-cubed, USDA FSA, USGS, AEX, Getmapping, Aerogrid, IGN, IGP, swisstopo, and the GIS User Community)

Thaa Atoll

Thaa Atoll, officially known as Kolhumadulu Atoll, has three islands with a reported presence of mangroves: Kan'doodhoo, Vandhoo and Veymandoo (EPA Sensitive Areas 2015) (Fig. 9). No study has reported the identity of the species present.

Fig 9



Geographic location of islands with a reported presence of mangroves in Thaa Atoll (Sources: Esri, DigitalGlobe, GeoEye, i-cubed, USDA FSA, USGS, AEX, Getmapping, Aerogrid, IGN, IGP, swisstopo, and the GIS User Community)

Laamu Atoll

In Laamu Atoll, officially known as Haddhunmathi Atoll, the islands of Gaadhoo, Gan, Hithadhoo and Maabaidhoo contain protected areas in which mangroves have been documented. C. tagal and R. mucronata have been recorded in the mangrove forests of Maabaidhoo and Hithadhoo, and L. racemosa has additionally been reported on Hithadhoo. On Gan, B. gymnorrhiza has been documented (Protected Area Announcement (IUL)438-ENV-438-2021-371). Mangroves have further been reported on Kalhaidhoo and Maavah islands (EPA Sensitive Areas 2015). On Maavah, plants of B. cylindrica and S. caseolaris have been observed (Rifqa 2022). In summary, mangroves have been reported on six islands of Laamu Atoll (Fig. 10). Five species have been documented, namely B. cylindrica, C. tagal, L. racemosa, R. mucronata and S. caseolaris (Table S1 in chapter 3, Supplementary Information).

Fig 10

Geographic location of islands with a reported presence of mangroves in Laamu Atoll (Sources: Esri, DigitalGlobe, GeoEye, i-cubed, USDA FSA, USGS, AEX, Getmapping, Aerogrid, IGN, IGP, swisstopo, and the GIS User Community)

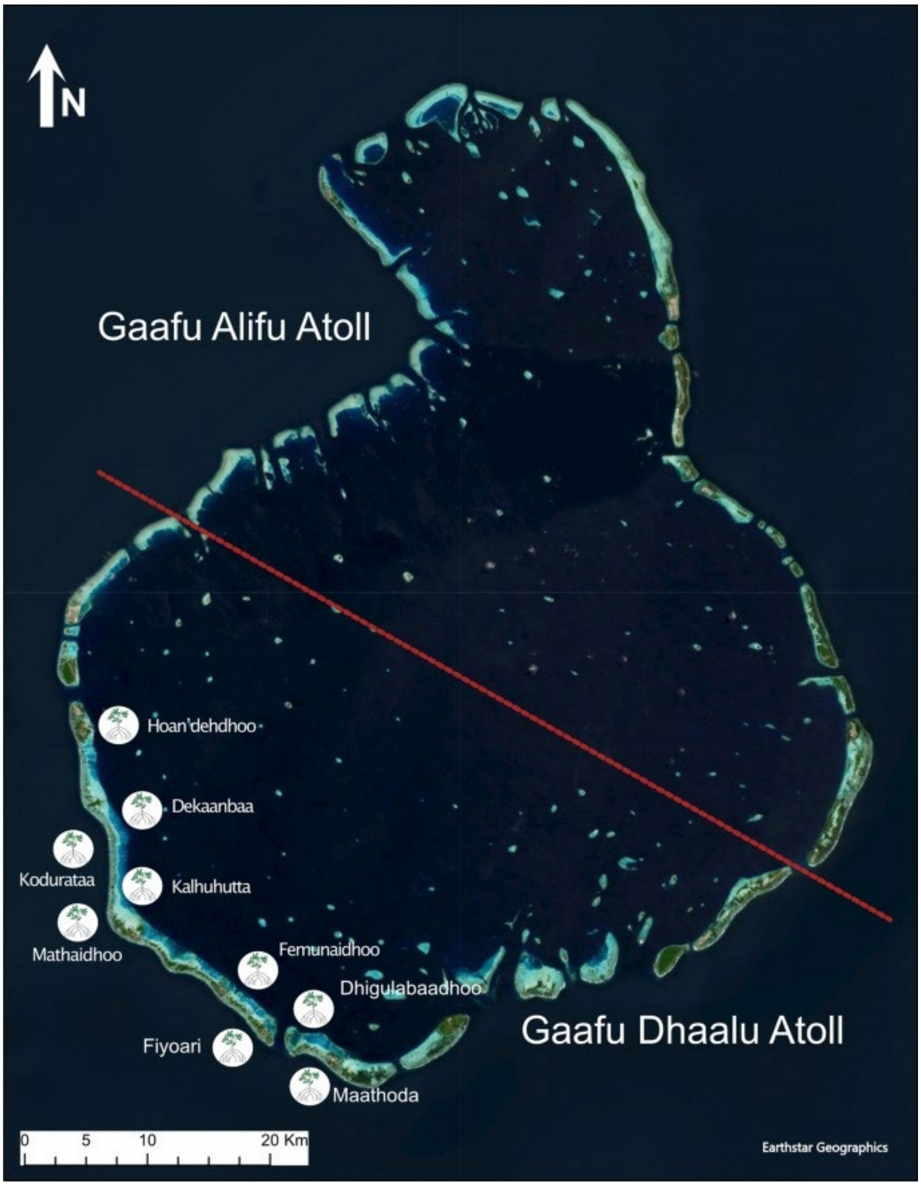
Gaafu Alifu Atoll

In Gaafu Atoll, officially known as Huvadhu Atholhu Uthuruburi (or Northern Huvadhu Atoll), no mangroves have been reported.

Gaafu Dhaalu Atoll

Gaafu Dhaalu Atoll, officially known as Huvadhu Atholhu Dhekunuburi (or Southern Huvadhu Atoll), comprises 153 islands. Among these, Dhigulaabadhoo was declared a protected area on 7 October 2018 (Protected Area Announcement (IUL)438-ENV-438-2018-262). An embayment of mangrove forest has been reported on this island, dominated by P. acidula and C. tagal, with scattered patches of R. mucronata (IUCN 2020). The EPA has documented the presence of mangroves in sensitive areas of Dekaanba, Femunaidhoo, Fiyoari, Hoan'dehdhoo, Kalhuhutta, Koduratаa, Maathoda and Mathaidhoo (EPA Sensitive Areas 2015). On Hoan'dehdhoo, B. cylindrica, B. gymnorrhiza and L. racemosa have been specifically reported (MEE 2015). In total, mangroves have been reported on nine islands of Gaafu Dhaalu Atoll (Fig. 11). Five species have been documented, namely B. cylindrica, B. gymnorrhiza, C. tagal, P. acidula and R. mucronata (Table S1 in chapter 3, Supplementary Information).

Fig 11



Geographic location of islands with reported presence of mangroves in Gaafu Atoll (Sources: Esri, DigitalGlobe, GeoEye, i-cubed, USDA FSA, USGS, AEX, Getmapping, Aerogrid, IGN, IGP, swisstopo, and the GIS User Community)

Gnaviyani Atoll

Gnaviyani Atoll consists solely of the inhabited Island of Fuvahmulah (Fig. 12), which is situated below the equator. Fuvahmulah, a UNESCO Biosphere Reserve, encompasses two protected areas, Dhandimagu Kilhi and Bandaara Kilhi (Protected Area Announcement (IUL)438-PPIRS-438-2012-2). Mangroves have been documented in both protected areas; however, no studies have reported the identity of the species present.

Seenu Atoll
Geographically, the southernmost atoll of the Maldives is Seenu Atoll, also known as Addu Atoll, which is situated south of the equator and is a UNESCO Biosphere Reserve. The atoll contains two islands with mangroves: Hithadhoo Island and Hulhumeedhoo Island (comprising the administratively separate islands of Hulhudhoo and Meedhoo; Fig. 12). On Hithadhoo Island, there are two distinct protected areas: one is located at the northern edge of the island and has been protected since 13 September 2018 (Protected Area Announcement (2018-R-105)), and the other is situated within the urbanised area and has been protected since 22 September 2020 (Protected Area Announcement (IUL)438-ENV-438-2020-162). The mangrove forest comprises C. tagal, L. racemosa, P. acidula and R. mucronata (Latheefa et al. 2009). Mangroves are also present in Hulhumeedhoo, which is geographically a single island comprising the two administratively distinct areas of Hulhudhoo and Meedhoo, and which contains two mangrove/wetland ecosystems that have been protected since 22 September 2020 (Protected Area Announcement (IUL)438-ENV-438-2020-162).

Fig 12

Geographical location of islands with a reported presence of mangroves of Gnaviyani Atoll and Seenu Atoll (Sources: Esri, DigitalGlobe, GeoEye, i-cubed, USDA FSA, USGS, AEX, Getmapping, Aerogrid, IGN, IGP, swisstopo, and the GIS User Community)

Archipelago-wide Distribution

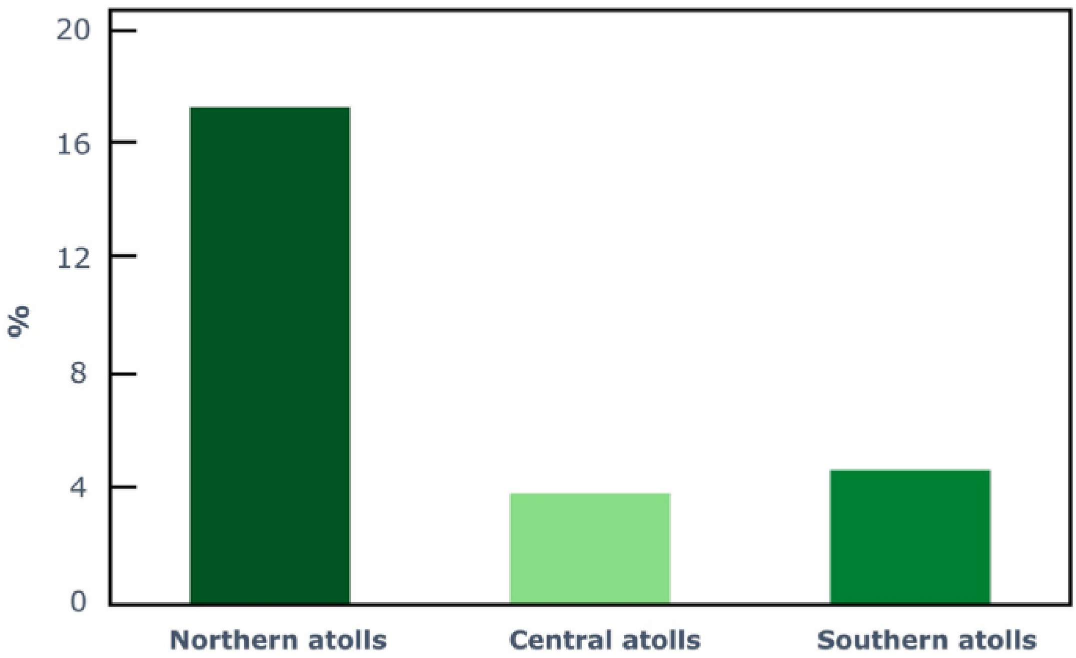
Following an extensive review of the literature, documented evidence was found for mangroves on 108 islands, representing approximately 9% of all islands in the Maldivian archipelago (Table S1 in chapter 3, Supplementary Information). This figure contrasts with the estimate of 150 islands provided by Saleem and Nileysha (2003), though the methodology employed in that study was not specified. Of the 108 islands with mangroves, only 23 (21.3%) have at least one protected mangrove area. Mangroves are unevenly distributed from north to south across the archipelago, with the highest occurrence in the northern atolls. On a national scale, 70.4% of islands with mangroves are located in the northern atolls (Fig. 1), comprising 76 islands, compared to 19 in the central atolls and 13 in the southern atolls. Among the northern atolls, Shaviyani Atoll has 19 islands with mangroves, closely followed by Haa Dhaalu Atoll (16 islands) and Haa Alifu Atoll (13 islands), making these three atolls the regions with the most significant occurrence of mangrove ecosystems in the Maldives. Northern Ari Atoll, Southern Ari Atoll and Gaafu Alifu Atoll are the only atolls with no reported presence of mangroves.

To better characterise the distribution pattern, it is important to consider the proportion of islands with mangroves relative to the total number of islands in each atoll, as a higher total number of islands inherently increases the probability of mangrove occurrence. Accordingly, Fig. 13 provides a visual representation of the percentage of islands with a reported presence of mangroves in the northern, central and southern regions of the archipelago (generated on the basis of the present review). The northern atolls have the highest proportion of islands with mangroves, exceeding 17.4%, compared to 3.9% in the central atolls and 4.7% in the southern atolls. This uneven distribution may be related to the proximity of mangrove seed and propagule sources, particularly from India and Sri Lanka to the northeast, as well as to climatic conditions that strongly influence the structure of the archipelago. Monsoon dynamics have substantially shaped the morphology of atolls and islands, producing a notable gradient from north to south. The monsoon intensifies towards the north of the archipelago, while wave energy diminishes in magnitude, and annual rainfall decreases from south to north (Kench 2012).

The northern atolls not only host the greatest number of islands with mangroves but also support the highest species diversity, which is likewise likely attributable to closer proximity to regional sources of seeds and propagules. A total of 14 species have been recorded in the northern atolls, compared to 7 in the central atolls and 6 in the southern atolls. Of the 14 species documented in the northern atolls, A. marina, A. aureum, B. hainesii, E. agallocha, H. littoralis and X. moluccensis are reported exclusively from this region. In contrast, B. cylindrica, B. gymnorrhiza, C. tagal, L. racemosa and R. mucronata are distributed throughout all regions of the Maldives without apparent geographical restriction (Table S1 in chapter 3, Supplementary Information).

It is noteworthy that approximately half of the islands with mangroves — specifically 55 of the 108 — are inhabited. Given that the Maldives has 188 inhabited islands in total (National Bureau of Statistics 2021), this indicates that nearly one third of the inhabited islands in the country support mangroves. On one hand, this raises concerns about the potential pressures these ecosystems face from human activities. On the other hand, it presents a valuable opportunity for sustainable co-management of these ecosystems through protection and conservation in partnership with local communities.

Fig 13



Percentage of islands with documented presence of mangroves out of the total number of islands in the northern, central and southern atolls

Ecosystem services of mangroves in the Maldives

Mangroves provide several ecosystem services of particular relevance to the Maldives. Historically, fruits of B. cylindrica ('Kandhoo') served as a critical subsistence food source during the national famine, locally known as 'bodu thadhu', which occurred during World War II (IDEAS 2017; Dryden et al. 2020a, b). Islands with Kandhoo plants were among the most food-secure during this period, and strict rules governing harvest periods were imposed on the population. Today, owing to the ready availability of rice, the Kandhoo fruit is no longer a dietary staple, though it continues to be consumed as an occasional delicacy (IDEAS 2017).

Mangroves provide another critical ecosystem service through coastal protection, mitigating the impact of tidal events and reducing the risk of catastrophic flooding, particularly during the monsoon season (Guannel et al. 2016; Agardy et al. 2017). On Kendhikulhudhoo Island, for example, during the 2004 tsunami, mangroves in conjunction with coral reefs and coastal vegetation absorbed and attenuated the destructive force of the waves, thereby reducing loss of human life and damage to buildings (Bluepeace 2010; CHEC 2020). During the rainy season, mangrove roots absorb a portion of rainfall and transfer it to the island's freshwater lens. Furthermore, mangroves assist in the removal of toxic substances and contaminants from water bodies and freshwater lenses (CHEC 2020).

Mangrove forests are carbon-fixing habitats that contribute to the reduction of anthropogenic carbon dioxide in the atmosphere. However, to date no research has been published on rates of carbon sequestration in the Maldives (Agardy et al. 2017).

Mangrove plants also constitute a source of timber, fuelwood and thatching materials, particularly B. cylindrica, B. gymnorrhiza and R. mucronata (Shadiya et al. 2016; Dryden et al. 2020a, b). Fish from mangrove ecosystems have historically been, and continue to be, an important source of protein for local communities (Shadiya et al. 2016).

Fauna Associated with Mangroves in the Maldives

Mangroves in the Maldives play a critical role in sustaining the biodiversity of coastal ecosystems, providing protection, food and habitat for a wide variety of animal species (Shadiya et al. 2016; Agardy et al. 2017). Numerous bird species utilise mangrove areas for feeding, breeding and shelter. Mangrove trees frequently serve as roosting sites for migratory birds, and the nests of resident species are commonly found within the canopy (IDEAS 2018a, 2018b; IUCN 2020).

Fish seek refuge and forage within mangrove forests. The complex root systems of many species, including Rhizophora spp., provide important nursery habitats for various fish including Carcharhinus melanopterus (the blacktip reef shark), as well as herbivorous species that feed on algae growing in the wetland. Numerous other fish species (Table S2 in chapter 3, Supplementary Information) have been recorded within mangrove bays, including members of the families Mullidae, Ostraciidae and Gerreidae (Shadiya et al. 2016; IUCN 2020). Fish species commonly found in lake-based mangrove systems include Chanos chanos (milkfish) and various species of the families Chanidae, Cichlidae, Congridae, Gobiidae, Poeciliidae and Sphyraenidae (IUCN 2020). A large number of upside-down jellyfish (Cassiopea xamachana) have been recorded in the shallow mangrove bay of Goidhoo (Bers 2005). Turtle nests have also been found on beaches adjacent to certain mangrove areas (IUCN 2020).

A diverse assemblage of crabs inhabits mangrove sediments, particularly visible during low tides. These crabs play a crucial role in the sustainability of mangrove ecosystems by influencing forest structure, zonation patterns, nutrient cycling, sediment dynamics and overall productivity. Crabs in the family Sesarmidae collect fallen leaves, seeds and propagules and transport them into burrows as a food source. This feeding behaviour contributes to soil enrichment and accelerates the incorporation of mangrove biomass into the food chain. Their burrowing activities aerate the substrate and promote water infiltration into sediments, facilitating sediment mixing and creating microenvironments for bacteria and fungi that supply essential nutrients for primary production. These crabs also serve as a vital food source for numerous fish and bird species (Ewel et al. 1998; Bluepeace 2015; Shadiya et al. 2016; Dryden et al. 2020a, b). Other crab species typically associated with mangrove ecosystems include fiddler crabs (Family Ocypodidae, e.g. Uca rapax), which process sediment to consume plant detritus, microbes and algae, and whose presence has been associated with enhanced mangrove health (Smith et al. 2009), as well as land crabs (Family Geocarcinidae — Cardisoma carnifex) and hermit crabs (Bluepeace 2015; Shadiya et al. 2016).

Limited research exists on the meiofauna and macrofauna inhabiting mangrove sediments (infauna) in the Maldives, yet these communities play crucial ecological roles in maintaining ecosystem function, nutrient cycling and biodiversity (Aller 1988; Snelgrove 1997). Macrofauna — larger visible organisms ranging from 250 µm to 1 cm in size (Snelgrove 1998) — participate in burrowing and bioturbation activities that improve oxygenation, nutrient cycling and overall ecosystem health. They decompose organic matter, release nutrients, and function as both prey and predators, thereby regulating population dynamics. Macrofauna also construct complex burrow systems that provide habitat and shelter. The macrofaunal community comprises a diverse array of taxa including isopods (sea slaters), amphipods, small gastropods (snails), polychaete worms and bivalves (clams, oysters, cockles, mussels and scallops). Meiofauna — smaller organisms ranging from 32 µm to 1 mm in size (Giere 2009) — accelerate decomposition, promote nutrient recycling and function as a food source for higher trophic levels. They include nematodes, copepods, ostracods and small worms (Lalli and Parsons 1997). The movements and feeding activities of both groups influence the structure, oxygenation and stability of the sediment, and both serve as indicators of environmental health and water quality. Monitoring their composition and diversity therefore provides valuable information for assessing and managing environmental impacts on mangrove habitats.

In the Maldives, the majority of studies on mangrove infauna have focused on nematodes. In Noonu Atoll, a study by Gerlach (1962) reported the presence of the free-living nematode Halalaimus filum. Schulz (1935) documented the occurrence of Nudora thorakista in the atolls of Addu and Noonu. Meiofaunal and macrofaunal communities in the Maldives have been studied primarily in coral reef and lagoonal sediment habitats (Semprucci et al. 2018; Grassi et al. 2022). Further investigation of macrofaunal and meiofaunal communities in Maldivian mangroves is both necessary and important. Such studies would generate valuable data on biodiversity, community dynamics and the functional contributions of these organisms to mangrove sediment ecosystems. In the Maldives, as in the wider Indian Ocean region (Hollander et al. 2020), very few studies have examined the capacity of marine organisms and ecosystems to adapt to climate change, further underscoring the importance of such research for comprehensive ecosystem management and conservation.

Flora associated with mangroves in the Maldives

Mangrove forests are globally recognised for supporting considerable floral diversity (Kathiresan and Rajendran 2005). In the Maldives, Saleem and Nileysha (2003) documented the occurrence of Tournefortia argentea (tree heliotrope, locally 'Boashi'), Thespesia populnea (milo, 'Hirundhu'), Scaevola taccada ('Magoo'), Pandanus sp. ('Kashikeyo'), Hibiscus tiliaceus (beach hibiscus, 'Dhiggaa') and Barringtonia asiatica (fish poison tree, 'Kinbi') in association with mangroves. Seagrasses (locally 'Moodhu vina') have also been reported within Maldivian mangrove ecosystems (Glen et al. 2008). Five seagrass species are known to occur in the Maldives — Syringodium isoetifolium, Thalassia hemprichii, Thalassodendron ciliatum, Cymodocea rotundata and Cymodocea sp. — of which T. hemprichii is the most widespread (Hameed 2022). However, limited information is available on which of these species are typically found in mangrove habitats or which mangrove species they are associated with.

Mangrove forests provide habitat for macroalgal species that grow epiphytically on pneumatophores, prop roots, basal trunks and surrounding sediments (Zuccarello et al. 2001; Mendonça and Lana 2021). Macroalgae (seaweeds) are a primary source of energy and provide habitat for diverse communities of small benthic animals, including nematode worms, bivalves, copepods, polychaetes and crabs, thereby enhancing the faunal diversity of mangrove systems (Mendonça and Lana 2021; Neba et al. 2021). Despite their important ecological functions, macroalgae are frequently excluded from ecological surveys of mangrove flora (West et al. 2013), primarily owing to their inconspicuous size and the difficulty of accurate identification. Studies reporting on algal diversity in the Maldives are extremely scarce and tend to focus on taxonomic inventories and species richness, either without molecular validation of identified species or with molecular validation conducted only for a limited number of selected specimens (e.g. Dictyota J.V. Lamouroux, Padina Adanson and Halimeda J.V. Lamouroux; Payri et al. 2012). In reports from the Maldivian government, algae are often described broadly as 'algal turf' (e.g. MEE 2015; Dryden et al. 2020a, b). Dhunya et al. (2017) reviewed ecosystem function services and threats to coral, mangrove and seagrass habitats but made only brief mention of algae in these habitats, without discussing mangrove-algal associations. The survey by Hackett (1977) is frequently cited as a reference for algal diversity in the Maldives; however, the author only briefly documented a small number of algal species associated with Rhizophora mucronata. Although Payri et al. (2012) produced the most comprehensive census of algal diversity in the Maldives to date, encompassing 321 species, these records are limited to Baa Atoll and make no reference to habitat associations. While very little is known about mangrove-associated macroalgae in the Maldives, mangrove habitats are globally recognised for supporting diverse macroalgal communities (Post 1936). The representative assemblage of such algae is commonly referred to as the 'bostrychietum complex' (Post 1936), and includes red algal genera such as Bostrychia, Caloglossa and Catenella. Although the bostrychietum complex has been reported from various locations in the Indian Ocean (Lambert et al. 1987; Steinke and Naidoo 1990; Phillips et al. 1996; Ganesan et al. 2018), it has not yet been described in the Maldives. Furthermore, no studies to date have comprehensively characterised Maldivian algae at the molecular level, which is crucial for adequate ecosystem management and conservation.

Knowledge gaps

This extensive review highlights several critical knowledge gaps regarding mangrove ecosystems in the Maldives, notably as follows.

- Lack of information on the precise location and boundaries of mangrove forests, and an absence of satellite or aerial mapping of these areas, which limits comprehensive assessments of mangrove health, ecosystem services and biodiversity.
- Lack of accurate data on the identity and abundance of mangrove species within individual mangrove areas, a gap which extends to the associated fauna and flora. Species identification is documented for only a limited number of mangrove areas (Table S1 in chapter 3, Supplementary Information) and the methods used for identification are frequently unclear. Identification of different taxa should be undertaken using standard taxonomic references in combination with

DNA barcoding to confirm the identity of cryptic species. Information on species identity and abundance is fundamental both to conservation efforts and to assessments of the general health of mangrove ecosystems, and there is therefore an urgent need to address these gaps.

- Lack of long-term monitoring of environmental parameters within mangrove areas, particularly air and water temperature, pH, water salinity and light intensity. These parameters are of critical importance given that they are undergoing significant changes as a result of climate change, altered precipitation patterns and rising sea levels. Integrating environmental parameter monitoring with biodiversity assessments and species distribution surveys would provide a more holistic understanding of how environmental change is affecting mangrove flora and fauna.
- Absence of studies on soil and sediment analysis in mangrove forests. Soil composition, grain size distribution, clay content, organic content, cation exchange capacity and physicochemical parameters including pH, salinity and temperature of mangrove sediment are essential for comprehensive ecological characterisation and for understanding potential relationships between species distribution and soil characteristics.

Conclusions and future research

The existing literature on mangrove forests in the Maldives reveals their unique character, comprising patches of trees in closed or semi-enclosed brackish water bodies and muddy regions. These mangroves have been documented on 108 islands, representing approximately 9% of all islands in the country. The Maldives supports 11 true mangrove species — A. marina, B. cylindrica, B. gymnorrhiza, B. hainesii, B. sexangula, C. tagal, L. racemosa, R. apiculata, R. mucronata, S. caseolaris and X. moluccensis — and 4 controversial species: A. aureum, E. agallocha, H. littoralis and P. acidula. The majority of mangrove forests are concentrated in the northern atolls, particularly in Haa Alifu, Haa Dhaalu and Shaviyani atolls, which together account for 76 islands with mangroves. Of these, 23 islands have at least one protected mangrove area, underscoring their conservation significance. These ecosystems provide vital ecosystem services that sustain livelihoods — including food provision, coastal protection and water purification — and also constitute a valuable source of timber, fuelwood and thatching materials. Furthermore, mangroves are instrumental in supporting coastal biodiversity by providing essential protection, food and habitat for diverse organisms, including numerous bird species, fish, crabs and marine invertebrates such as upside-down jellyfish (Cassiopea xamachana), free-living nematodes (Halalaimus filum and Nudora thorakista), fiddler crabs (Uca rapax), land crabs (Cardisoma carnifex) and hermit crabs. Several plant species are also associated with Maldivian mangroves, including T. argentea, T. populnea, S. taccada, Pandanus sp., H. tiliaceus and B. asiatica, contributing to the overall diversity and health of these ecosystems.

Nonetheless, this review has identified significant inconsistencies and gaps in the literature relating to various aspects of mangrove ecosystems in the archipelago, particularly in uninhabited islands. These include the absence of precise information on mangrove species locations and boundaries, the lack of satellite or aerial mapping, insufficient long-term monitoring of environmental parameters such as temperature, pH and salinity, and a lack of soil and sediment analyses. These knowledge gaps impede a comprehensive understanding of mangrove distribution, and the impacts of environmental change on mangrove health and resilience.

To address these gaps, a nationwide biodiversity survey of all mangrove forests in the Maldives is considered essential. Such a survey would provide invaluable data on the precise locations and boundaries of mangrove areas throughout the archipelago. Through systematic documentation and mapping of mangroves and their associated fauna and flora, researchers and conservationists could establish a comprehensive baseline from which to monitor change and assess ecosystem health. Field surveys would additionally enable the collection of critical data on environmental parameters, including temperature, salinity and sea level change, as well as soil and sediment physicochemical characteristics. A thorough and inclusive survey effort would make a substantial contribution to informed decision-making, policy development, improved scientific knowledge, and evidence-based management and conservation strategies, ultimately supporting the long-term sustainability of the invaluable mangrove ecosystems of the Maldives.

Supplementary materials

Table S1. Islands with reported presence of mangroves in the Maldives. I = inhabited, U = uninhabited, P = protected, S = sensitive, nr = not reported

Island	Atoll	Human habitation	Status	Species of mangroves
Baarah	Haa Alifu	I	P	*B. cylindrica, C. tagal, E. agallocha, L. racemosa, R. apiculata,* and *R. mucronata* (Bluepeace, 2015)
Berinmadhoo	Haa Alifu	U	S	nr
Filladhoo	Haa Alifu	I	–	*B. cylindrica, B. gymnorrhiza, L. racemosa, R. apiculata,* and *R. mucronata* (Bluepeace, 2015)
Gallandhoo	Haa Alifu	U	P	nr
Kelaa	Haa Alifu	I	P	*A. aureum, B. cylindrica, B. gymnorrhiza, B. hainesii, L. racemosa, S. caseolaris, R. mucronata* (Dryden et al. 2020; IDEAS, 2017)
Maafahi	Haa Alifu	U	S	nr
Maarandhoo	Haa Alifu	I	S	nr
Madulu	Haa Alifu	U	S	nr
Muraidhoo	Haa Alifu	I	S	nr
Naridhoo	Haa Alifu	U	S	nr
Thakandhoo	Haa Alifu	I	S	nr
Uligan	Haa Alifu	I	S	nr
Vangaaru	Haa Alifu	U	S	nr
Faridhoo	Haa Dhaalu	U	S	nr
Hanimaadhoo	Haa Dhaalu	I	S	nr
Hirimaradhoo	Haa Dhaalu	I	S	nr
Hirinaidhoo	Haa Dhaalu	U	S	nr
Keylakunu	Haa Dhaalu	U	P	*A. marina, B. cylindrica, E. agallocha, P. acidula, R. apiculata, R. mucronata, S. caseolaris* (Bluepeace Maldives; IUCN, 2020)
Kulhudhuffushi	Haa Dhaalu	I	S	*B. cylindrica, L. racemosa, R. mucronata, S. caseolaris* (IUCN, 2019; MEE, 2015)
Kumundhoo	Haa Dhaalu	I	S	nr
Kurinbee	Haa Dhaalu	I	S	nr
Maavaidhoo	Haa Dhaalu	I	S	nr
Naagoashi	Haa Dhaalu	U	S	nr
Naivadhoo	Haa Dhaalu	I	S	nr
Nellaidhoo	Haa Dhaalu	I	S	nr

Neykurendhoo	Haa Dhaalu	I	P	*B. cylindrica, E. agallocha, L. racemosa,* and *X. moluccensis* (Bluepeace, 2015)
Nolhivaran	Haa Dhaalu	I	S	nr
Nolhivaranfaru	Haa Dhaalu	I	–	*B. cylindrica, P. acidula, R. mucronata,* and *X. moluccensis* (IDEAS, 2018a)
Vaikaradhoo	Haa Dhaalu	I	S	nr
Eriyadhoo	Shaviyani	U	S	nr
Farukolhu	Shaviyani	U	P	*C. tagal, P. acidula,* and *R. mucronata* (IUCN, 2020)
Feevah	Shaviyani	I	S	nr
Feydhoo	Shaviyani	I	S	nr
Firunbaidhoo	Shaviyani	U	S	nr
Foakaidhoo	Shaviyani	I	S	nr
Funadhoo	Shaviyani	I	S	nr
Goidhoo	Shaviyani	I	S	nr
Kandeetheemu	Shaviyani	I	S	nr
Keekimini	Shaviyani	U	S	nr
Kudadhoo	Shaviyani	U	S	nr
Kudalhaimandhoo	Shaviyani	U	S	nr
Maakandoodhoo	Shaviyani	U	S	*A. aureum, B. cylindrica, E. agallocha,* and *R. apiculata* (IUCN, 2020)
Madidhoo	Shaviyani	U	S	nr
Maroshi	Shaviyani	I	S	nr
Migoodhoo	Shaviyani	U	S	nr
Milandhoo	Shaviyani	I	S	*L. racemosa, P. acidula,* and *R. mucronata* (IDEAS, 2018b)
Narudhoo	Shaviyani	I	S	nr
Noomara	Shaviyani	I	S	nr
Bodulhaimendhoo	Noonu	U	P	*P. acidula* and *R. apiculata* (IUCN, 2020)
Burehifasdhoo	Noonu	U	S	nr
Goanbilivaadhoo	Noonu	U	S	nr
Kandoodhoo	Noonu	U	S	nr
Kendhikulhudhoo	Noonu	I	P	*Bruguiera* spp., *Rhizophora* spp., and *P. acidula* (IUCN, 2020)
Kuramadhoo	Noonu	U	S	nr
Landhoo	Noonu	I	S	nr
Lhohi	Noonu	I	S	nr
Maalhendhoo	Noonu	I	–	*C. tagal, E. agallocha,* and *R. apiculata* (Nishan, 2010)
Manadhoo	Noonu	I	S	nr
Dhiffushimaidhoo	Lhaviyani	U	P	*H. littoralis* and *P. acidula* (IUCN, 2020)
Faadhoo	Lhaviyani	U	S	nr
Kanifushi	Lhaviyani	U	S	nr
Lhohi	Lhaviyani	U	S	*B. cylindrica* (Jameel & Faiz, 2016)
Lhossalafushi	Lhaviyani	U	S	nr
Maakoa	Lhaviyani	U	P	*B. cylindrica* and *P. acidula* (IUCN, 2020)

Thilamafushi	Lhaviyani	U	S	nr
Varihuraa	Lhaviyani	U	S	nr
Dheburideyetherevaadhoo	Raa	U	S	nr
Kan'doogan'du	Raa	U	S	nr
Neyo	Raa	U	S	nr
Vaadhoo	Raa	I	S	nr
Dhakandoo	Baa	U	S	nr
Goidhoo	Baa	I	P	nr
Keyodhoo	Baa	U	S	nr
Maamaduvvari	Baa	U	S	nr
Maddoo	Baa	U	S	nr
Olhugiri	Baa	U	P	nr
Huraa	Kaafu	I	P	*B. cylindrica, B. gymnorrhiza, R. apiculata,* and *R. mucronata* (MEE, 2015)
Kaashidhoo	Kaafu	I	P	nr
Anbaraa	Vaavu	U	S	nr
Ruh Hurihuraa	Vaavu	U	S	nr
Mulah	Meemu	I	S	nr
Muli	Meemu	I	S	nr
Dhiguvarufinolhu	Faafu	U	S	*B. cylindrica, P. acidula* (Dryden et al. 2021)
Kan'dinma	Dhaalu	U	S	nr
Thilabolhufushi	Dhaalu	U	S	nr
Thinhuraa	Dhaalu	U	S	nr
Kan'doodhoo	Thaa	I	S	nr
Vandhoo	Thaa	I	S	nr
Veymandoo	Thaa	I	S	nr
Gaadhoo	Laamu	I	P	nr
Gan	Laamu	I	P	nr
Hithadhoo	Laamu	I	P	*C. tagal, L. racemosa,* and *R. mucronata* (Protected Area Announcement IUL-438-ENV-438-2021-371)
Kalhaidhoo	Laamu	I	S	nr
Maabaidhoo	Laamu	I	P	*C. tagal* and *R. mucronata* (Protected Area Announcement IUL-438-ENV-438-2021-371)
Maavah	Laamu	I	S	*B. cylindrica* and *S. caseolaris* (Rifqa, 2022)
Dekaanbaa	Gaafu Dhaalu	U	S	nr
Dhigulaabadhoo	Gaafu Dhaalu	U	P	*C. tagal, P. acidula,* and *R. mucronata* (IUCN, 2020)
Femunaidhoo	Gaafu Dhaalu	U	S	nr
Fiyoari	Gaafu Dhaalu	I	S	nr
Hoan'dehdhoo	Gaafu Dhaalu	I	S	*B. cylindrica* and *B. gymnorrhiza* (MEE, 2015)
Kalhuhutta	Gaafu Dhaalu	U	S	nr
Kodurataa	Gaafu	U	S	nr

	Dhaalu			
Maathoda	Gaafu Dhaalu	U	S	nr
Mathaidhoo	Gaafu Dhaalu	U	S	nr
Fuvahmulah	Gnaviyani	I	P	nr
Hithadhoo	Seenu	I	P	*C. tagal, L. racemosa, P. acidula,* and *R. mucronata* (Latheefa et al. 2019)
Hulhumeedhoo (Hulhudhoo and Meedhoo)	Seenu	I	P	nr

Table S2. Fishes found in the embayment mangroves of the Maldive

Scientific name	Common name
Abudefduf septemfasciatus	Nine-band sergeant
Acanthurus striostegus	Convict surgeonfish
Caranx melampygus	Blue-fin jack
Carcharhinus melanopterus	Black-tip reef shark
Chaetodon auriga	Threadfin butterflyfish
Chaetodon lunula	Racoon butterflyfish
Chanos sp.	Milkfish
Chromis viridis	Blue-green chromis
Corythoichthys haematopterus	Piperfish
Epinephelus caeruleopunctatus	Small-spotted grouper
Gerres longirostris	Longtail silverbiddy
Gerres oyena	Blacktip pursemouth
Himantura fai	Whiptail stingray
Himantura granulata	Mangrove whipray
Lethrinus harak	Blackspot emperor
Lutjanus argentimaculatus	Mangrove red snapper
Lutjanus fulviflamma	Blackspot snapper
Negaprion acutidens	Sicklefin lemon shark
Ostracion cubicus	Yellow boxfish
Parupeneus barberinus	Dash-dot goatfish
Pastinachus sephen	Cowtail stingray
Plectorhinchus chaetodonoides	Sweetlips
Siderea picta	Peppered moray
Siganus argenteus	Rabbit fish
Sphyraena barracuda	Great barracuda
Urogymnus asperrimus	Porcupine ray

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Chapter 4

Analysis of algae and coral under thermal stress conditions

4.1 Introduction –

Algae are crucial and integral flora within mangrove ecosystem. We can broadly classify algae into three ecological groups, namely phytoplankton, benthic microalgae, and macroalgae in mangrove ecosystems (Gao and Lin., 2018). Undoubtedly algae ecosystems are key contributors in primary production, organic carbon production, nutrient cycle, carbon sequestration, mitigating water pollution and element cycle (Gao and Lin., 2018). Diatoms are among the most abundant photosynthetic eukaryotes in the oceans, indicating their immense global ecological impact; recent evidence points to contribution of diatoms in biogeochemical cycles, both as primary producers of organic material and as conduits facilitating the export of carbon and silicon to the ocean interior (Benoiston et al., 2017).
Li & Yao (2024) in their work distinguished between micro and macro algae, microalgae as small, unicellular organisms found within diverse aquatic environments, exhibiting rapid growth rates and high photosynthetic efficiency. In contrast macroalgae are large multicellular organisms inhabiting coastal ecosystems and requiring attachment to substrates. Although macroalgae has slower growth rate compared to microalgae; their lager biomass can compensate for growth rate and contribute significantly to carbon capture (Li & Yao 2024). Raven (2018) elaborates on evidence of macroalgal ecosystem role in marine carbon sequestration. There is huge potential for macroalgae (seaweeds and kelps) to be applied for nature-based solutions for climate change mitigation, although there is IPCC's recognition of Blue Carbon ecosystems, seaweed ecosystems are largely neglected from carbon accounting and policy frameworks (Samadder et al., 2025). Most importantly both macroalgae and microalgae are significant contributors in phenomenon of capture, utilization, and storage of carbon (Li & Yao 2024). Algae, especially diatoms are crucial elements within the blue carbon ecosystem like mangroves in terms of contribution to organic carbon pools in coastal sediment; studies provide evidence of mangrove plants being the largest contributor to organic carbon pools (Arina et al., 2023). Quite recent estimates indicate that diatoms contribute to ~40% of ocean primary production (Harvey et al., 2019). Furthermore, eDNA analysis suggests diatoms form the dominant share of species within the mangrove sediments (Arina et al., 2023). It is well documented that sea level mean temperature is increasing due to climate change, studies indicate that heat stress (e.g., up to 40 °C) for microalgae like Tetraselmis sp. can evoke stress response resulting in altered carotenoid and pigment profiles (Santos et al., 2025). Studies demonstrate that high thermal stress inhibited carotenoid production and altered overall pigment metabolism compared with control conditions (Santos et al., 2025; Patrinou et al., 2022). Transmission electron microscopy (TEM) analysis shows that the cells get bleached within the thylakoid membrane structure, which in turn impacts photosynthetic pigment content (Ko et al., 2019). Further investigation on microalgae exhibits role of various sub lethal temperature exposures (26–39°C). It can impact growth, biomass, and chlorophyll-a levels over multiple days which validates biochemical stress responses to temperature effects (Zargar et al., 2006). Lee & Hsu., 2013 elaborates on a subjecting S. vacuolatus cells to heat pretreatment (46.5 °C) and monitoring recovery; the authors documented how photosynthetic activity and pigment-related physiology were inhibited during the heat treatment and then partially regained during recovery, indicating thermal stress greatly impacts the photosynthetic apparatus, including chlorophyll-dependent processes. In Spirulina spp. high temperatures can lead to reduced chlorophyll and carotenoid content and increased oxidative stress markers, which highlights effects of thermal stress on pigment composition in a cyanobacterial microalga (Rehman et al., 2025). Therefor it is important to conduct studies on thermal stress induced pigment profile among microalgae. Similarly in case of Scenedesmus sp. there is evidence of loss of chlorophyll when exposed to higher temperatures, heat treated cells become photo bleached (loss of chlorophyll fluorescence) compared with control cells.

Coral reefs provide immense ecological benefits, serving as a cornerstone for marine blue economy ecosystems. They protect coastlines from storms and erosion and can also be a source of food or medicine. For example, the antiviral drug Vir-A (Vidarabine), used to treat herpes simplex infections, was developed from compounds isolated from the Caribbean sponge Tectitethya crypta that lives in coral reefs (Bergmann & Burke, 1955; Bruckner, 2002). Despite occupying a very small portion of area (less than 0.1% of the ocean floor), they host more than 820,000 different species (Fisher et al., 2015). From an economic perspective, the total value of coastal protection provided by reefs is over $4 billion in averted damages during usual storms. Projections suggest that in 25 to 100 years, corals can prevent $36 to $230 billion

worth of damages. They can also be used to define jurisdictional boundaries over marine territories. Coral reefs support fish and seafood species that enrich the global fisheries trade industry, like groupers or lobsters. These activities can be worth around $100 million each year in U.S. recreational fisheries and depend on healthy coral systems to sustain. In general, the net economic value estimated from a coral reef is $10 billion per year. Additionally, tourism is a vital activity; Australia saw more than 26 million visitors in 2016, while Hawaii and Florida generate up to $2 billion annually from reef-related tourism. Heat stress is the primary driver of the breakdown in the nutritional symbiosis between the coral host and its intracellular algae (Symbiodiniaceae). When temperatures rise, the photosynthetic apparatus of the algae becomes impaired, leading to the accumulation of reactive oxygen species (ROS) that cause cellular damage to both the algae and the host (Weis, 2008). As a result, the coral expels the algae, leading to a "bleached" appearance and a state of metabolic starvation. Beyond bleaching, corals under thermal stress show a significant reduction in calcification rates—the process by which they build their calcium carbonate skeletons—and a decrease in their ability to allocate energy toward reproduction (Hoegh-Guldberg et al., 2007). If the thermal stress is prolonged or intense, it leads to widespread tissue necrosis and colony mortality, which ultimately degrades the structural complexity of the entire reef ecosystem (Pratchett et al., 2014).
This chapter pertains to analysis of algae Tetraselmis, Ceramium and corals Pocillopora damicornis and Stylophora pistillata, there were 3 studies which have been divided in 3 subchapters. The next subchapter explains our studies where we exposed Tetraselmis to thermal stress and then did an analysis within control (healthy) specimens vis-à-vis stressed specimens. The research aim was to apply innovative analytical methods for the identification of stress markers induced by anthropogenic factors in marine algae. Following the analysis of Tetraselmis there is the sub chapter elaborating on the period of 6 months abroad where I was working in my host institution Stockholm University, focusing on detecting pigment changes due to thermal stress within Ceramium spp. Finally the last subchapter deals with investigation of thermal stress and detection of metabolites within corals Pistillata and Damicornis.

Subchapter 4.2- Analysis of Tetraselmis under thermal stress

4.2.1 Materials and methods -

4.2.1.1 Samples –
The biological material used in this study consisted of mono-specific cultures of the marine microalga Tetraselmis spp., sourced from the Acquario di Genova (Genoa, Italy). To ensure experimental reproducibility, the samples were categorized into two distinct cohorts based on their physiological state:

- Control Cohort (n = 18): These samples represented "healthy" or baseline physiological conditions. They were maintained under optimal standardized conditions—typically characterized by a stable temperature of 20°C (±1°C), constant aeration, and a 12:12 hour light/dark photoperiod—to ensure exponential growth and high photosynthetic efficiency.

- Thermally Stressed Cohort (n = 20): This group was subjected to controlled thermal stress (32°C for 24 hours) to induce physiological divergence. The stress protocol involved a rapid elevation of the ambient temperature beyond the species' optimal thermal window, intended to trigger cellular defense mechanisms, heat-shock protein expression, or observable changes in lipid and pigment composition.

All 38 samples were processed immediately following the stress induction period to preserve the integrity of the metabolic and transcriptomic signatures. The physical condition of the stressed samples was verified via microscopy, noting initial indicators of chlorosis or reduced motility compared to the high-vitality control group.

4.2.1.2 Chemicals -
All reagents and solvents were of the highest purity grade available. The solvents used for liquid

chromatographic elution were ultra-grade methanol (MeOH) purchased from Merck (Merck KGaA, Darmstadt, Germany) and Ultrapure Water (resistivity, 18.2 MΏ-cm) produced on a Milli-Q Plus apparatus (Millipore, Milan, Italy). Standards of -β-carotene and chlorophyll a were purchased from Sigma-Aldrich (Sigma Aldrich, Darmstadt, Germany). Methanol, ammonium acetate was purchased from Sigma-Aldrich (Sigma Aldrich, Darmstadt, Germany).

4.2.1.3 Dry weight determination –

To determine the total dry weight (DW), the freeze-dried biomass was kept in a desiccator and subsequently weighed on an analytical balance.

4.2.1.4 Pigment extraction -

The carotenoids and chlorophylls were exhaustively extracted from the freeze-dried samples (0.02g) with acetone in a mortar with a pestle followed by centrifugation for 10 min at 5500 rpm (Fernandes et al., 2017).

4.2.1.5 Chromatographic Analysis of Algal Pigments via HPLC-DAD -

The procedure of extraction and analysis of the pigments was carried out following indications from the literature. The pigment analysis was made using an Agilent 1260 Infinity HPLC-DAD instrument, detecting the continuous wavelengths in the visible spectrum in the 350 to 700nm window. For the elution, a binary gradient obtained using a mixture of 80% methanol and 1M 20% ammonium acetate 1M, solvent B 60% methanol and acetone 40% acetone as solvent A. The column used for chromatographic separation was Eclipse Plus C18 3.5µm with the dimensions of 4.6x100mm. For solvent B it was used a solution made by 60% methanol and 40% acetone. The flow rate was 0.550 ml/min. Samples were injected after elution 1 to 1000, with the injection volume of 10 microliters.

Chromatographic separations were carried out at 25 °C, with a flow rate of 500 µL min^{-1} and an injection volume of 10 µL. We employed a gradient elution program to accommodate the wide polarity range of target pigments. The mobile phase consisted of: (i) Solvent B: 80% methanol (MeOH) and 20% 1 M ammonium acetate ($C_2H_7NO_2$), and (ii) Solvent C: 60% MeOH and 40% acetone (C_3H_6O). The inclusion of ammonium acetate as an ionic modifier improves peak shape for chlorophylls and related polar pigments and has been shown to stabilize pigment retention behavior during reverse phase separation, without interfering with DAD detection. Acetone in the strong mobile phase enhances elution strength for non-polar carotenoids such as β-carotene, promoting sharper peaks and better resolution.
Pigment identification was verified by comparing both retention times and UV visible absorbance spectral profiles obtained from the DAD with those of commercial standards of chlorophyll a and β-carotene. We further validated peak identities by comparing our chromatographic profiles with established reference pigment fingerprints reported in the literature. This dual approach (retention time + spectral matching) is recommended in algal pigment studies to ensure correct assignment of overlapping or structurally similar pigment peaks, reducing ambiguity inherent to complex environmental extracts.

4.2.1.6 Pigment quantification and statistical analysis -

Pigment composition was quantified from chromatographic peak areas. To account for differences in total chlorophyll content, selected carotenoids were normalized to beta carotene (Brunet et al. 2011; Lavaud & Goss 2014; Roy et al. 2011). Comparisons between healthy and stressed samples were performed using two-sided Mann–Whitney U tests due to non-normal distributions of pigment values (Legendre & Legendre 2012; Litchman et al. 2004; Rasconi et al. 2015). Data are presented as mean plots with error bars. All statistical analyses were performed using JASP (version 0.9.5.4; JASP Team, 2019)

4.2.2 Results and discussion –

We found clear differences among healthy and stressed Tetraselmis algae. The thermal stress elicits photo protective pigment accumulation response. We found significantly higher concentrations of the pigment Lutein (Mann–Whitney U test, $p = 0.013$) among in stressed Tetraselmis samples compared to healthy controls. While no significant differences were observed for peridinin or chlorophyll a1 and a2; however, there is clear evidence of stress-related changes in lutein allocation. The increase in lutein under stress suggests enhanced photo protective pigment accumulation. The observed variability in our data, reflected in the large error bars (interquartile range), is a result of the extensive sampling design which included 18 healthy controls and 20 thermal stress replicates. These replicates were not gathered from a single batch but were performed across five different dates spanning March to July 2024 (27/03, 04/04, 14/05, 13/06, and 16/07), thereby capturing the natural biological and physiological breadth of Tetraselmis spp. throughout different growth phases. Despite this high variance, the increase in Lutein remained statistically significant ($p = 0.013$), underscoring the robustness of this photoprotective response. Analysis of pigments indicates the effect of physiological stress (temperature increase) and demonstrates photo acclimation in photosynthetic algae. Chlorophylls are primary pigments for light harvesting and photosynthesis while carotenoids likeβ-carotene, lutein, peridinin function as Accessory pigments with photo protective and antioxidant roles (Gitelson et al., 2020; Zhou et al., 2019; He et al., 2023). There was no significant decrease in the chlorophyll content which suggests the algae were undergoing active photo-protective remodelling rather than irreversible pigment bleaching, consistent with the broader literature on carotenoid-mediated stress responses in microalgae (Cazzonelli, C. I. 2011; Pérez-Pérez et al. 2012 ; Xu & Harvey 2019).

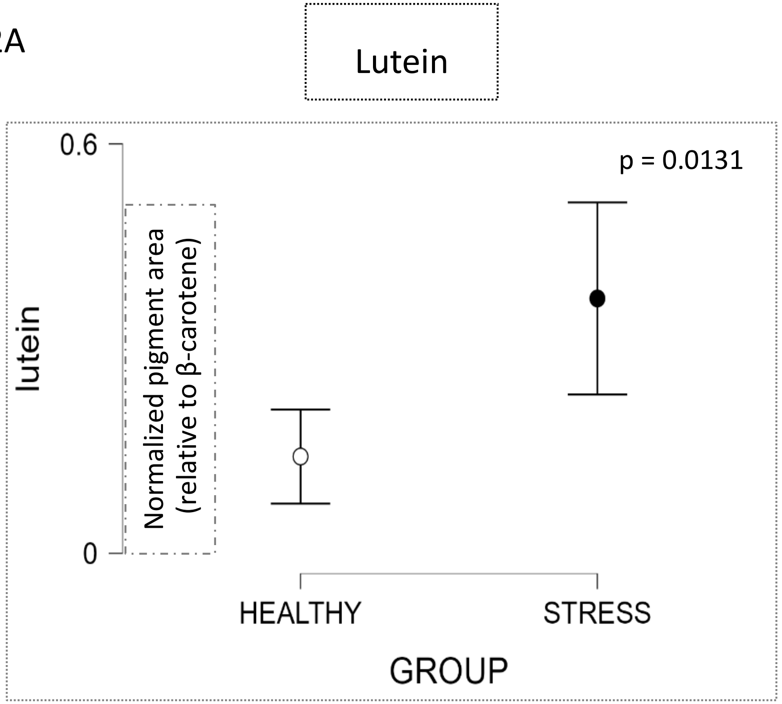
Table 4.2.2.1 – P values obtained from Mann Whitney test. Under stress protective response significantly increases levels of Lutein

Pigments	U	p	Hodges-Lehmann Estimate
Peridinin unknown	147	0.342	-0.189
Diatoxanthin	180	1	-34.62
lutein	95	0.013*	-0.183
Chlorophyll a-2	215	0.313	0.914
Chlorophyll a1	221	0.236	1.568

* **p < 0.05**

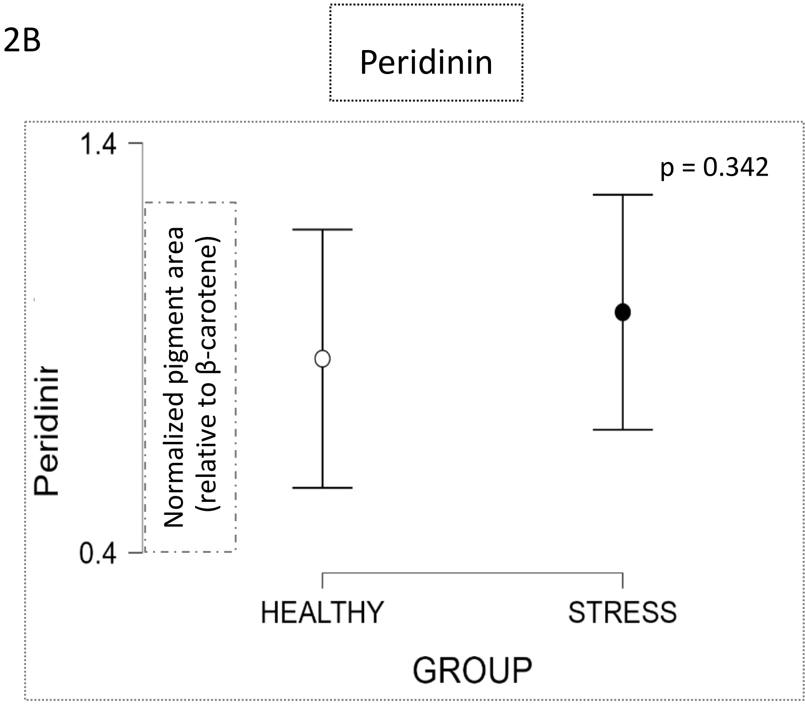
Figure 2 A, B, C, D, E, - Mean plots and error bars of pigment abundances in healthy and stressed algae Tetraselmis spp... Pigment peak areas were normalized to β-carotene. Statistical significance was assessed using the Mann–Whitney U test. Figures show (A) lutein, (B) peridinin, (C) chlorophyll a1, (D) chlorophyll a2 and (E) diatoxanthin. Boxplots represent the median and interquartile range. Statistical differences between groups were evaluated using a two-sided Mann–Whitney U test.

2A



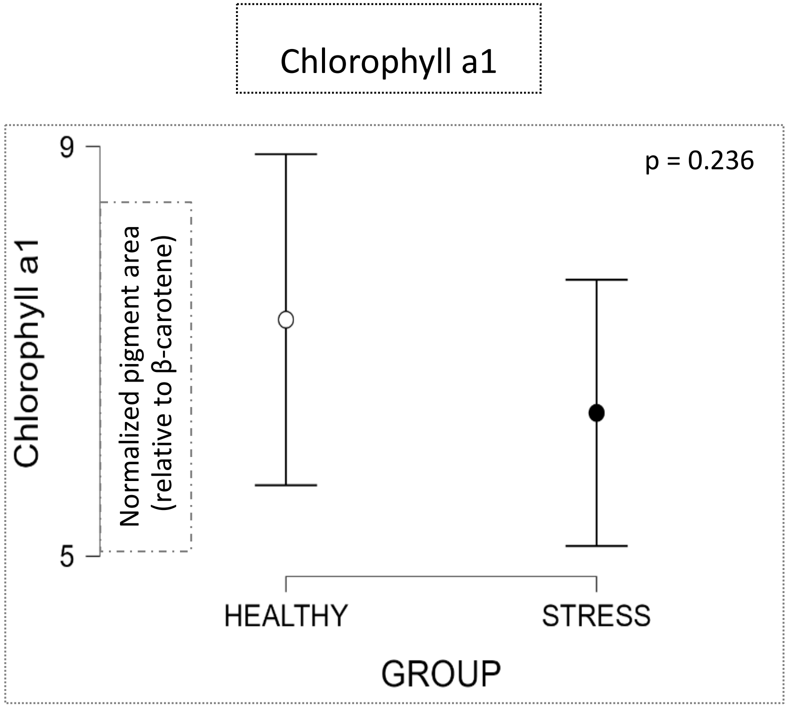
Lutein
0.6
0
p = 0.0131
lutein
Normalized pigment area
(relative to β-carotene)
HEALTHY
STRESS
GROUP

2B

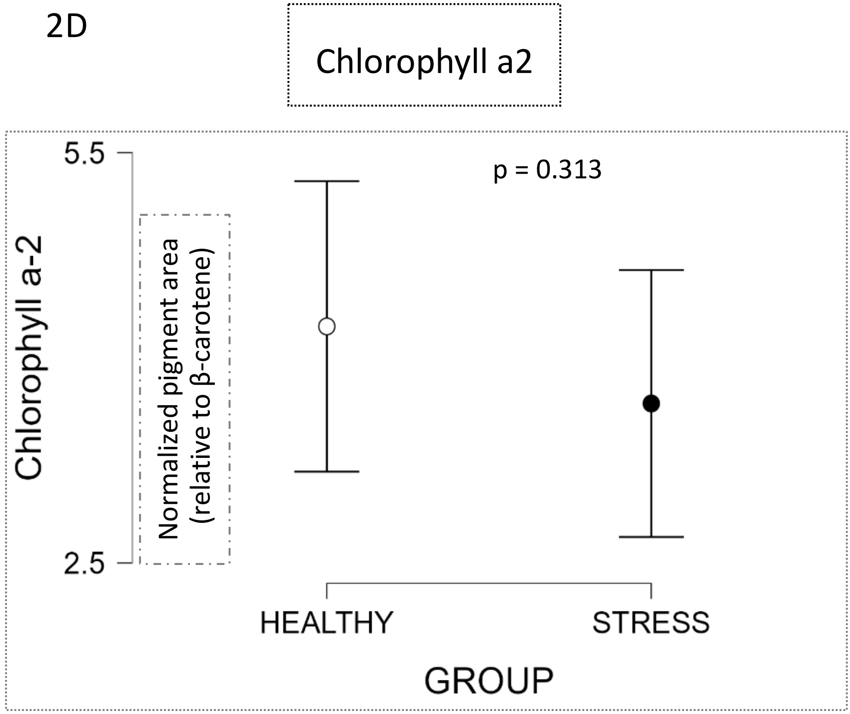


Peridinin
1.4
0.4
p = 0.342
Peridinir
Normalized pigment area
(relative to β-carotene)
HEALTHY
STRESS
GROUP

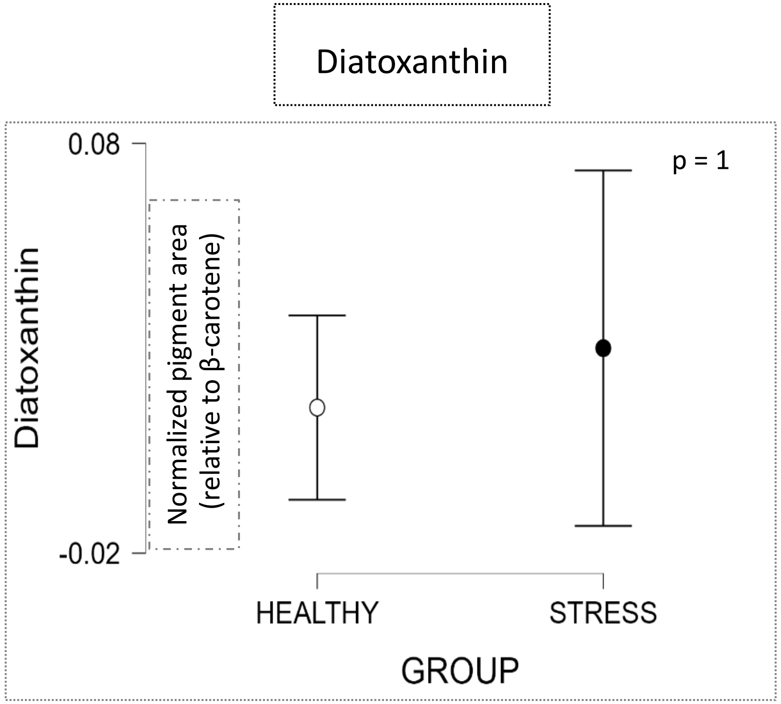
2C


Chlorophyll a1
p = 0.236
9
5
Chlorophyll a1
Normalized pigment area
(relative to β-carotene)
HEALTHY
STRESS
GROUP

2D


Chlorophyll a2
p = 0.313
5.5
2.5
Chlorophyll a-2
Normalized pigment area
(relative to β-carotene)
HEALTHY
STRESS
GROUP

2E



Diatoxanthin
0.08
-0.02
p = 1
Diatoxanthin
Normalized pigment area
(relative to β-carotene)
HEALTHY
STRESS
GROUP

Conclusion –

Climate change is impacting blue economy algal ecosystems, with drastic changes in sea temperature it would alter the physiological response among these phytoplankton fauna. Algal communities extensively support aiding blue carbon ecosystem like mangroves. The pilot study demonstrates enhanced photo protective pigment accumulation among Tetraselmis; confirming the effect of physiological stress (temperature increase) and demonstrates photo acclimation. This is crucial in terms of understanding our knowledge about changing dynamics within blue carbon ecosystems. Algae, especially diatoms are crucial elements within the blue carbon ecosystem like mangroves due to major contribution to organic carbon pools in coastal sediment (Arina et al., 2023). Recent evidence suggests diatoms contribute to ~40% of ocean primary production (Harvey et al., 2019). Furthermore, eDNA analysis suggests diatoms form the dominant share of species within the mangrove sediments (Arina et al., 2023). This brings forward the need for further scientific studies which could be translated to policies centered around protection of blue carbon ecosystem and mitigate the concerns arising from sea temperature increase.

of the red filamentous macroalga Ceramium spp. (Rhodophyta), provided by the Department of Environmental Science (ACESX), Stockholm University. A total of 22 samples were analyzed to evaluate physiological divergence under thermal stress. To ensure experimental reproducibility, the samples were categorized into two distinct cohorts:

- Control Cohort (n = 12): These samples represented the baseline physiological state. They were maintained under standardized Stockholm archipelago-simulated conditions, typically characterized by a stable temperature of 12.5°C (±0.5°C), a salinity of 6.5 PSU, and a 16:8 hour light/dark photoperiod to support healthy metabolic activity (Ek et al., 2019).

- Thermally Stressed Cohort (n = 10): This group was subjected to an elevated temperature protocol (e.g., 25–30°C) to induce a stress response. Such thermal shifts in Ceramium are intended to trigger changes in the concentration of accessory pigments, specifically affecting the ratio of phycobiliproteins to chlorophylls and inducing potential chlorosis (Valverde et al., 2022).

4.3.1.2 Chemicals -

All reagents and solvents were of the highest purity grade available. The solvents used for liquid chromatographic elution were ultra-grade methanol (MeOH) purchased from Merck (Merck KGaA, Darmstadt, Germany) and Ultrapure Water (resistivity, 18.2 MΏ-cm) produced on a Milli-Q Plus apparatus (Millipore, Milan, Italy). Standards of -β-carotene and chlorophyll a were purchased from Sigma-Aldrich (Sigma Aldrich, Darmstadt, Germany). Methanol, ammonium acetate was purchased from Sigma-Aldrich (Sigma Aldrich, Darmstadt, Germany).

4.3.1.3 Dry weight determination –

To determine total dry weight (DW), the Ceramium biomass was carefully blotted to remove excess seawater, freeze-dried, and kept in a desiccator. Samples were subsequently weighed on an analytical balance to the nearest 0.0001 g.

4.3.1.4 Pigment extraction -

The carotenoids and chlorophylls were exhaustively extracted from the freeze-dried samples (0.02g) with acetone in a mortar with a pestle followed by centrifugation for 10 min at 5500 rpm (Fernandes et al., 2017).

4.3.1.5 Chromatographic Analysis of Algal Pigments via HPLC-DAD -

The procedure of extraction and analysis of the pigments was carried out following indications from the literature. The pigment analysis was made using an Agilent 1260 Infinity HPLC-DAD instrument, detecting the continuous wavelengths in the visible spectrum in the 350 to 700nm window. For the elution, a binary gradient obtained using a mixture of 80% methanol and 1M 20% ammonium acetate 1M, solvent B 60% methanol and acetone 40% acetone as solvent A. The column used for chromatographic separation was

Eclipse Plus C18 3.5µm with the dimensions of 4.6x100mm. For solvent B it was used a solution made by 60% methanol and 40% acetone. The flow rate was 0.550 ml/min. Samples were injected after elution 1 to 1000, with the injection volume of 10 microliters.

Chromatographic separations were carried out at 25 °C, with a flow rate of 500 µL min^{-1} and an injection volume of 10 µL. We employed a gradient elution program to accommodate the wide polarity range of target pigments. The mobile phase consisted of: (i) Solvent B: 80% methanol (MeOH) and 20% 1 M ammonium acetate ($C_2H_7NO_2$), and (ii) Solvent C: 60% MeOH and 40% acetone (C_3H_6O). The inclusion of ammonium acetate as an ionic modifier improves peak shape for chlorophylls and related polar pigments and has been shown to stabilize pigment retention behavior during reverse phase separation, without interfering with DAD detection. Acetone in the strong mobile phase enhances elution strength for non-polar carotenoids such as β-carotene, promoting sharper peaks and better resolution.
Pigment identification was verified by comparing both retention times and UV visible absorbance spectral profiles obtained from the DAD with those of commercial standards of chlorophyll a and β-carotene. We further validated peak identities by comparing our chromatographic profiles with established reference pigment fingerprints reported in the literature. This dual approach (retention time + spectral matching) is recommended in algal pigment studies to ensure correct assignment of overlapping or structurally similar pigment peaks, reducing ambiguity inherent to complex environmental extracts.

4.3.1.6 Pigment quantification and statistical analysis -

Pigment composition was quantified from chromatographic peak areas. To account for differences in total chlorophyll content, selected carotenoids were normalized to beta carotene (Brunet et al. 2011; Lavaud & Goss 2014; Roy et al. 2011). Comparisons between healthy and stressed samples were performed using two-sided Mann–Whitney U tests due to non-normal distributions of pigment values (Legendre & Legendre 2012; Litchman et al. 2004; Rasconi et al. 2015). Data are presented as boxplots (median and interquartile range). All statistical analyses were performed using JASP (version 0.9.5.4; JASP Team, 2019)

4.3.2 Results and discussion –

The results of the HPLC-DAD analysis conducted on Ceramium spp. revealed a pronounced physiological response to thermal stress. Based on the comparison between the control group (n=12) and the thermally stressed cohort (n=10), a highly significant reduction in primary photosynthetic pigments was observed. Statistical evaluation using the two-sided Mann-Whitney U test confirmed that the degradation of pigments was not due to random variation. As shown in Table 4.3.2.1 in chapter 4, both Chlorophyll a1 and Chlorophyll a2 exhibited a highly significant decrease under thermal stress conditions ($p<.001$). The quantitative data, normalized to β-carotene to account for variations in total biomass, highlighted the extent of pigment loss:

- Chlorophyll a1: The mean normalized area dropped significantly from approximately 3.5 in the healthy control group to below 2.0 in the stressed group.
- Chlorophyll a2: A similar degradation pattern was observed, with levels falling from a mean of approximately 0.045 in the control samples to roughly 0.020 in the stressed samples.

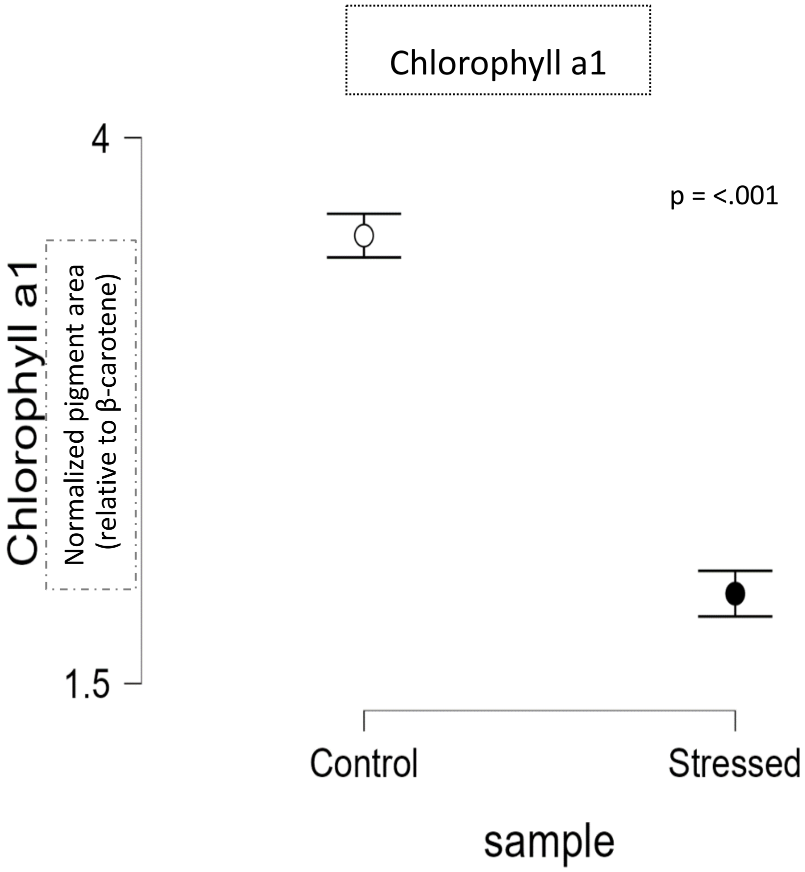
These findings, visualized in Figure 3 (A and B), demonstrate that thermal shifts beyond the species' optimal window trigger a substantial loss of photosynthetic pigments. This reduction in Chlorophyll a variants suggests that heat stress induces potential chlorosis and compromises the metabolic health and photosynthetic efficiency of Ceramium spp..

Table 4.3.2.1 – p values obtained from Mann Whitney test. Under thermal stress chlorophyll a1 and a2 significantly decreases

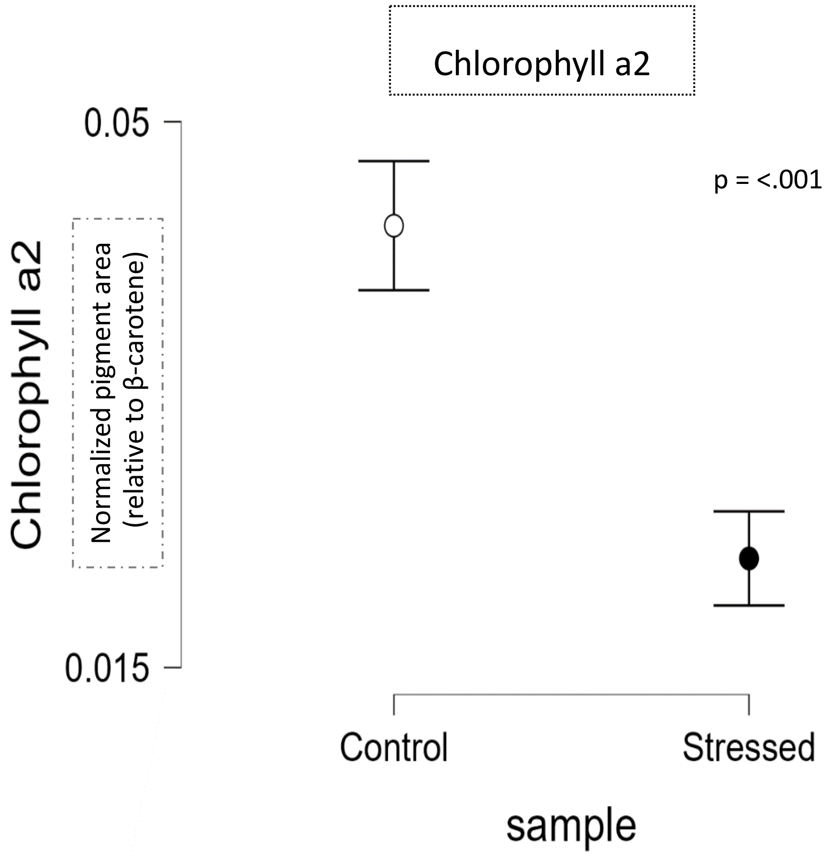
Pigments	p
Chlorophyll a1	< 0.001
Chlorophyll a2	< 0.001

Figure 3 A, B, - Mean plots with error bars of pigment abundances in healthy and stressed algae Ceramium spp... Pigment peak areas were normalized to β-carotene. Statistical significance was assessed using the Mann–Whitney U test. Figures show (A) chlorophyll a1 and (B) chlorophyll a2. Statistical differences between groups were evaluated using a two-sided Mann–Whitney U test.

3A



3B



Conclusions -

Based on the Mann-Whitney U test performed on a total of 22 samples (12 healthy and 10 stressed), there is a highly significant decrease in pigment concentration within Ceramium due to heat stress, as indicated by p-values of less than .001 for both Chlorophyll a1 and Chlorophyll a2. The data, visualized through both mean plots with error bars and grey bar plots, shows that Chlorophyll a1 normalized area dropped from a mean of approximately 3.5 in the control group to below 2.0 in the stressed group. A similar degradation is observed in Chlorophyll a2, where levels fell from a mean of approximately 0.045 to roughly 0.020 relative to β-carotene. This significant reduction in both primary pigments suggests that thermal stress leads to a substantial loss of photosynthetic pigments in Ceramium, potentially compromising the organism's metabolic health.

Subchapter 4.4- Analysis of Pocillopora damicornis and Stylophora pistillata under thermal stress

4.4.1 Materials and methods -

4.4.1.1 Samples and preconditioning–
The samples Tetraselmis spp. were kindly offered by the Aquarium of Genoa. The purpose of our experiments is to analyze how thermal stress can affect two species of corals: Pocillopora damicornis and Stylophora Pistillata. The first, also known as the cauliflower coral, is a colonial coral, belonging to the order Scleractinia that live in water of temperatures around 24-27°C mm and they have a calcareous skeleton. The second Stylophora pistillata is a salt water Cnidaria belonging to the family Pocilloporidae. The study meant to show how specifically these external stressors influence the coral, depending if it's stressed or not, analyzing the the production of non- polar metabolites such as steroids and terpenoids. There were 3 different conditions of the coral: control, which means they were maintained under 25°C, pre-conditioned, warmed up and maintained at 27C for a few days before sampling and non-pre-conditioned, maintained at first at 25°C but then taken at 30°C from the start to the end of the interested period of time.

4.4.1.2 SPME extraction and instrumental analysis non-targeted GCMS-
The analysis of non-polar metabolites utilized Solid Phase Micro Extraction (SPME) followed by Gas Chromatography-Mass Spectrometry (GC-MS) to identify and quantify steroids and terpenes in the coral tissue. SPME analysis was carried out in the following order –

- Tissue Extraction: Coral tissue was removed from the skeleton using compressed air and a saline solution.
- Solvent Purification: The non-polar metabolites were extracted using an organic solvent (acetone) followed by a purification procedure involving sonification to ensure the release of membrane-bound steroids.
- Extraction: The SPME fiber (typically a non-polar coating such as PDMS for these analytes) was exposed to the sample headspace or liquid extract to selectively adsorb non-polar compounds, including steroids and hydrocarbons.
- Thermal Desorption: The fiber was then inserted into the GC injector port at 280°C, where the analytes were thermally desorbed and transferred to the column by the Helium carrier gas.

SPME extraction was followed by non-targeted Gas Chromatography-Mass Spectrometry (GC-MS). Chromatographic separation was performed using an Agilent 8860 system equipped with an Agilent HP5 non-polar capillary column (5% phenyl methyl-polysiloxane; 30 m × 250 µm).

A 1 µL sample volume was introduced via split injection at a temperature of 280°C. The carrier gas flow was maintained at a constant column flow of 1.2 mL/min, with a purge flow set at 3.0 mL/min. To ensure the elution of complex high-molecular-weight metabolites, the oven temperature program utilized a linear ramp starting from 50°C and increasing to 320°C at a rate of 7°C/min.

Mass spectral data were acquired using an MS detector with a scanning range from 40.00 to 500.00 m/z. The acquisition parameters included a scan frequency of 1.6 scans/sec, a cycle time of 613.92 ms, and a step size of 0.1 m/z.

For the interpretative phase, data processing was conducted using Agilent Quantitative Analysis software. Individual chromatographic peaks, representing the detector signal as molecular fractions eluted, were integrated and analyzed. Compound identification was achieved through a dual-verification approach: first, by cross-referencing the relative mass spectra against the NIST MS Search library, and second, by validating these results against the elution times and spectra of reference molecules through the calculation of Kovats Retention Indices.

4.4.2 Results and discussion –

We evaluated 9 samples, 3 controls and 6 stressed and we divided the identified compounds in different classes. Then we compared them, evaluating if a compound is present or not after stressed. The following table illustrates the same.

Table 4.4.2.1 non-polar metabolites identified.

	type of molecule	name	C1	C2	C4	27	2	14	SP5	SP8	SP10
9.426	benzothiazole	benzothiazole	50271	134938	66849	14370	17609	52607			
10.810	keton	2-(2-methyl-propenyl)-cyclohexanone	9983	18848	13460	8261	11702	24333			
11.046	keton	2-Cyclohexen-1-one, 2,4,4-trimethyl-3-(3-methylbutyl)	35948	88845	88483	60417	31105	48181			
14,462	amide	N-Methoxymethyl-N-methylformamide	0	0	0	0	0	279445			
15,046	alkane	Hexadecane, 1,1-bis(dodecycloxy)	0	0	0	351594	0	396033			
15.631	aromatic hydrocarbon	azulene.1-4-dimethyl 7-(1-methylethyl)-	128753	511052	392119	53947	212674	474648			
15.928	fatty acid	tetradecanoic acid	88203	235311	142732	113713	99411	313832			
16.041	ester	7-methyl-Z-tetradecen-1-ol acetate	226453	986296	372692	45055	202969	398285			
16,554	aldehyde	Hexadecanal	0	0	0	0	0	239336			
16.780	alkene	Neophytadiene	604551	904333	673733	408590	708239	1459485	365603	548670	0
16,964	hydrocarbon	1,4-di-iso-propylnaphthalene	0	0	0	0	0	0	0	67559	0
17,026	alcohol	9-Octadecen-1-ol, (Z)-	803577	2244699	1151369	1719299	358754	1316969	364778	183447	0
17.139	ftalic acid	1,2-benzenedicarboxyilic acid , bis(2-methylpropyl) ester	635993	863878	1403082	1181512	708718	468019	496317	335321	851721
17.200	alcohol	1-hexadecanol	1499136	3329830	2110686	2415148	1124554	3199234	991333	760999	1568046
17.805	fatty acid	9-hexadecenoic acid	237291	7625637	462877	449552	302151	1270279	0	0	0
18.010	fatty acid	n-Hexadecanoic acid	1088233	2258934	1740984	940630	1514011	3434553	292597	179028	0
18,072	ester	1,2-Benzenedicarboxylic acid, butyl 2-ethylhexyl ester	0	0	0	0	0	0	0	98239	0
18,174	aromatic tiol	5-(Methylthio)-2-phenylpent-4-en-2-ol	0	0	0	0	0	337275	0	0	0
18.585	alcohol	ethanol,2-(9-octadecenyloxy)-.(Z)-	338507	534976	278570	288091	214180	390309	164313	136729	0
18,759	ester	5,8,11,14,17-Eicosapentaenoic acid, methyl ester, (all-Z)-	0	0	0	0	126033	270433	0	0	0
18.933	fatty acid	doconexent	613615	871294	584460	298742	332459	814135	0	125661	0
19,169	IPA	Fluoranthene	0	0	0	0	0	0	0	124488	0
19.467	alcohol	phytol	76031	229098	169724	50987	99485	397220	0	0	0
19,580	ester	cis-5,8,11,14,17-Eicosapentaenoic acid	0	0	0	0	181922	1342921	0	0	0
19,672	tetracyclic hydrocarbon	Pyrene	0	0	0	0	201650	268831	0	120797	0
19.857	fatty acid	octadecanoic acid	94650	408586	168946	126354	159068	544784	0	0	0
20,195	alkane	docasane	0	0	0	0	216137	273662	0	76978	0
21.057	alkane	triocosane	1343647	1497646	1780752	889174	491257	549948	197722	167583	813291
21,590	ester	Methyl dehydroabietate	0	0	0	0	117002	298881	0	0	0
21.877	alkane	tetracosane	3930252	4603086	5167390	2553883	1409491	1619444	559547	505975	2096266
22,082	ester	Dimethyldithiocarbamic acid, S-(2-cyano-2-methyl-1-phenyl)vinyl ester	0	1813249	1103293	0	475627	1517053	0	0	0
22.133	phenol	phenol, 2.4-bis(1-phenylethyl)-	475103	1380769	764986	84129	383775	1108620	0	81627	915453
22,267		unknown	0	2058284	1255895	0	696782	1409551	0	0	947319
22,380	alkane	Octadecane, 3-ethyl-5-(2-ethylbutyl)-	0	0	0	0	142522	0	0	0	0
22,677	alkane	pentacosane	7413578	8697490	9860833	4587355	2575688	2431423	458189	692365	1981649
22,728		unknown	0	0	0	0	524485	1057205	0	0	0
22,780	ester	Hexadecanoic acid, 1-(hydroxymethyl)-1,2-ethanediyl ester	0	0	0	0	0	265376	0	0	0
23,118	ester	Bis(2-ethylhexyl) phthalate	0	0	0	0	156228	369552	0	0	0
23,159	alkane	Hexatriacontane	0	0	0	0	175847	0	0	0	0
23,241	alkane	3-Methylpentacosane	0	0	0	0	177761	0	0	0	0
23,436	alkane	hexacosane	8021539	8261222	1E+07	5261714	2918564	2841770	546561	609751	2096349
23,908	alkane	Octadecane, 3-ethyl-5-(2-ethylbutyl)- / tetratetracontane	0	0	828962	432404	224742	0	0	0	0
23,990	alkane	Pentacosane, 13-undecyl-	0	0	0	316771	188723	0	0	0	0
24.175	alkane	heptacosane	5849739	6436193	7677782	3682398	2063105	2148437	473922	539033	1804153
24,257	siloxane	Tetracosamethyl-cyclododecasiloxane	0	0	0	0	163963	0	0	0	0
24.575	ester	1,3-benzenedicarboxylic acid, bis(2-ethylhexyl) ester	1246301	4375523	1877376	151388	897355	2094669	309101	118305	2778316
24,626	ester	Docosanoic acid, 1,2,3-propanetriyl ester	0	0	716174	315619	189833	0	0	0	0
24,708	alkane	Hexacosane, 9-octyl-	0	0	0	0	179723	0	0	0	0
24.892	alkane	octacosane	4367436	4907556	6537796	2774926	1656385	1672487	491952	422378	1457728
25,241	siloxane	Cyclononasiloxane, octadecamethyl-	0	0	981555	0	163077	0	0	0	0
25,405	alcano	3-Methyloctacosane	0	0	787184	0	0	0	0	0	0
25,549	steroid	Ergosta-5,22-dien-3-ol, acetate, (3β,22E)-	0	0	0	0	0	0	730419	562980	1229529
25.580	alkane	nonacosane	2617434	3328874	3329486	1959532	1102884	1114606	0	0	0
25,774	ester	1-Hydroxy-2-(2,3,4,6-tetra-O-acetyl-beta-d-glucopyranosyl)-9H-xanthene-3,6,7-triyl triaceta	0	0	0	0	0	0	0	0	0
25,980	steroid	unknown	0	0	0	0	0	0	104861	0	0
26,041	radical	1-Picryl-2,2-bis(p-(1,1,3,3-tetramethylbutyl)phenyl)hydrazyl radical	0	0	0	0	0	0	0	242270	0
26,072		Pessimi match	0	0	0	332385	125964	0	0	0	0
26,154	siloxane	1,1,1,3,3,5,5,7,7,9,9,11,11,13,13,15,15,17,17,19,19,19-docosamethyldecasiloxane	0	0	0	0	0	175916	0	222048	0
26.246	alkane	triacontane	1777997	2114191	2701153	1274717	799752	799609	592281	612491	786466

29,826	ester	6,9,12,15-Docosatetraenoic acid, methyl ester	2224023	4814071		1156692	0	0	0	0	3217358
29,990	ester	9-Octadecenoic acid (Z)-, hexadecyl ester	13772625	17048920	889501	4443620	323607	1791056	4695735	1269036	2,7E+07
30.021	ester	oleic acid, eicosyl ester	1773830	2849179	2768807	1234725	98327	415696	0	0	0
30,113	ester	Cetyl stearate	22077584	31724597	2,2E+07	8151033	1316484	5773915	3211486	1241438	1,8E+07
30,339		unknown	0	0	0	0	0	0	910748	132259	756064
30,564	alcohol	Olean-12-ene-3,15,16,21,22,28-hexol, (3β,15α,16α,21β,22α)-	0	0	0	0	0	0	0	186411	0
30,698	siloxane	1,1,1,3,3,5,5,7,7,9,9,11,11,13,13,15,15,15-octadecamethyloctasiloxane	0	0	0	0	154662	0	0	0	0
31,293		unknown	0	0	0	0	0	0	831345	97039	0
31,754	ester	7,10,13-Eicosatrienoic acid, methyl ester	4834917	9848366	5270072	2361210	0	0	466202	75499	4742058
31,918	ester	Butyl 11-eicosenoate	6448563	15100878	9787118	3443624	123555	861062	1325684	130558	5679764
31,959		unknown	0	0	0	0	0	0	299424	0	0

The chromatograms of the single samples are represented here (figure 4.4.2 A – 4.4.2 I) with the identification of the non-polar metabolites.

Figure 4.4.2A

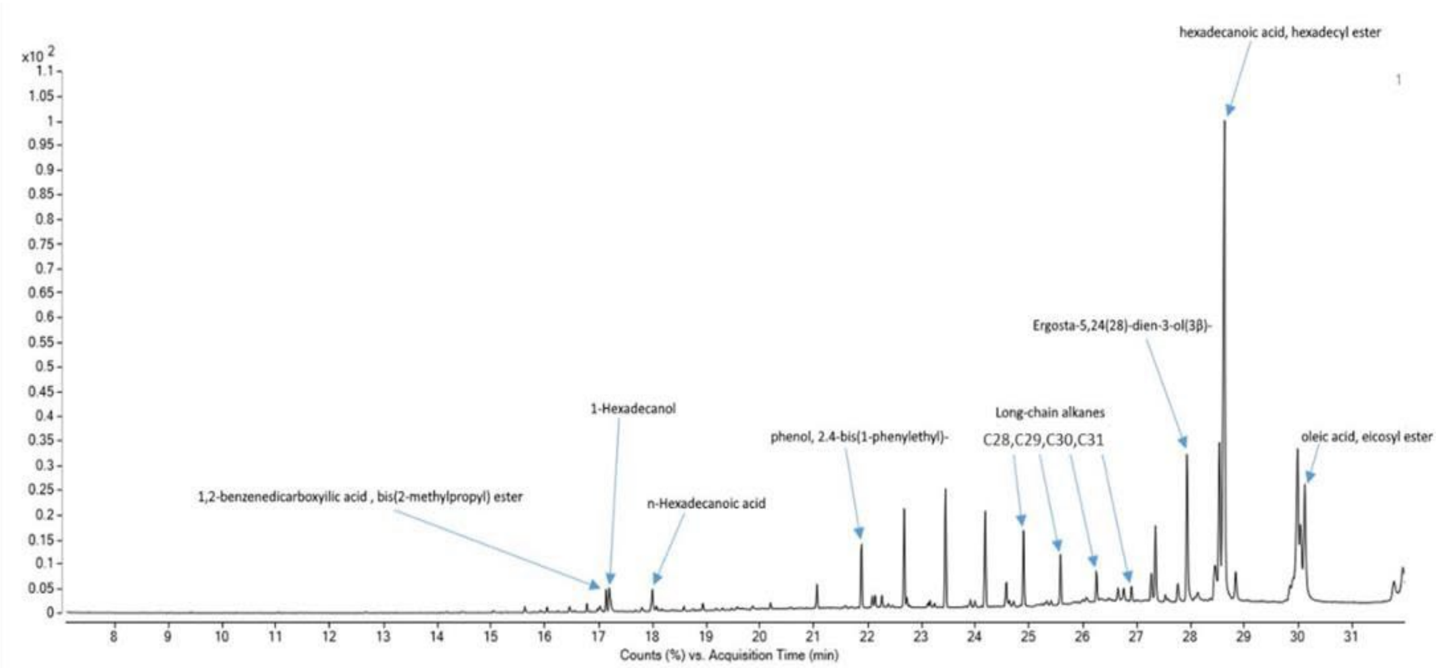


Figure 4.4.2B

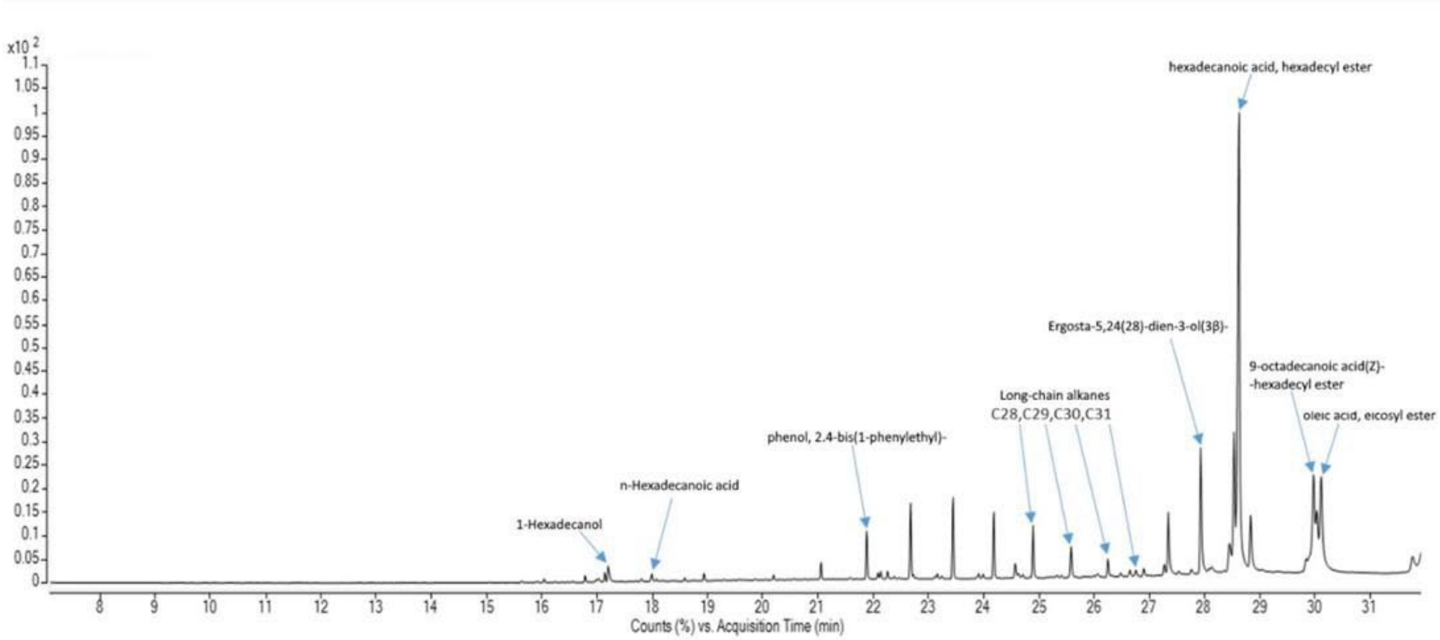


Figure 4.4.2C

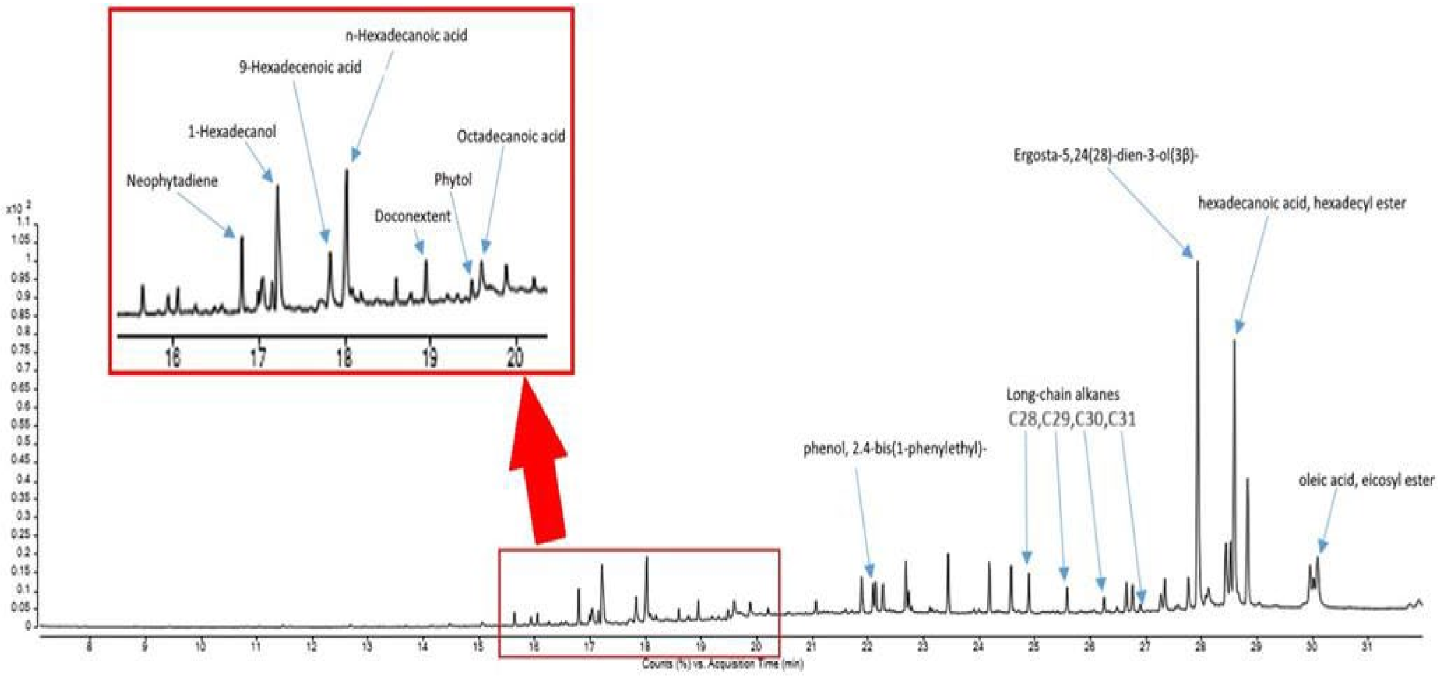


Figure 4.4.2D

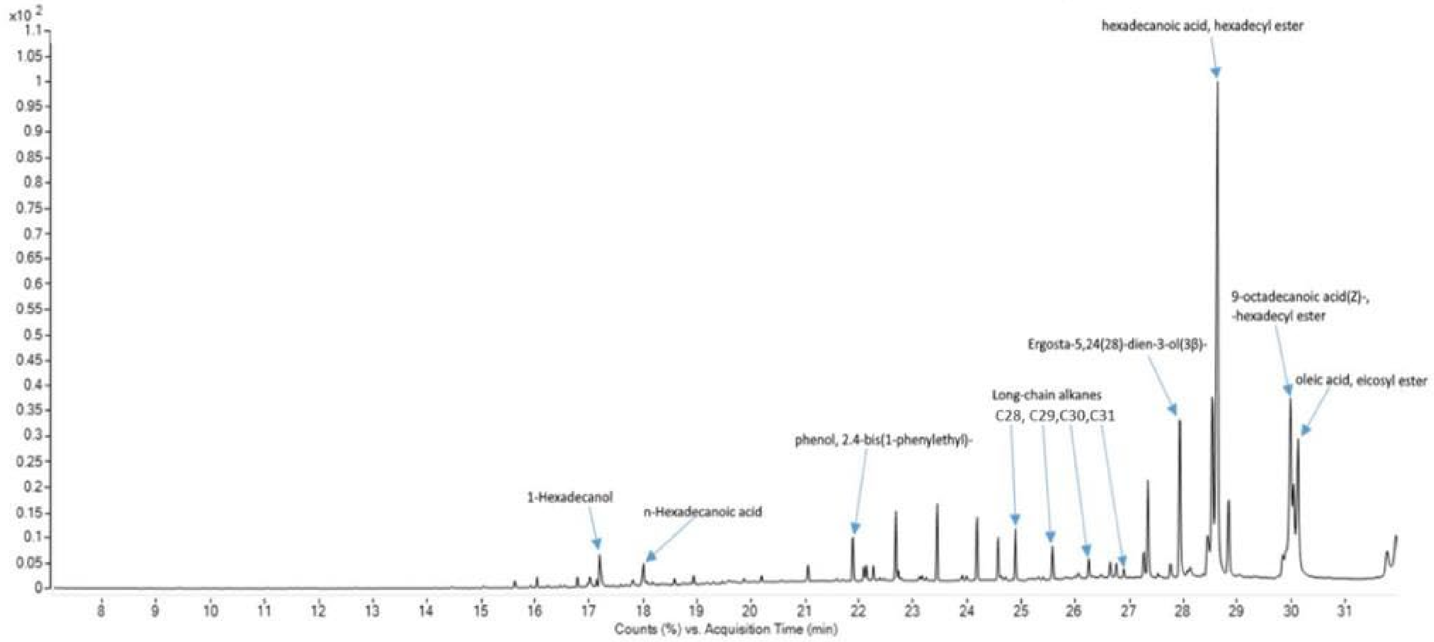


Figure 4.4.2E

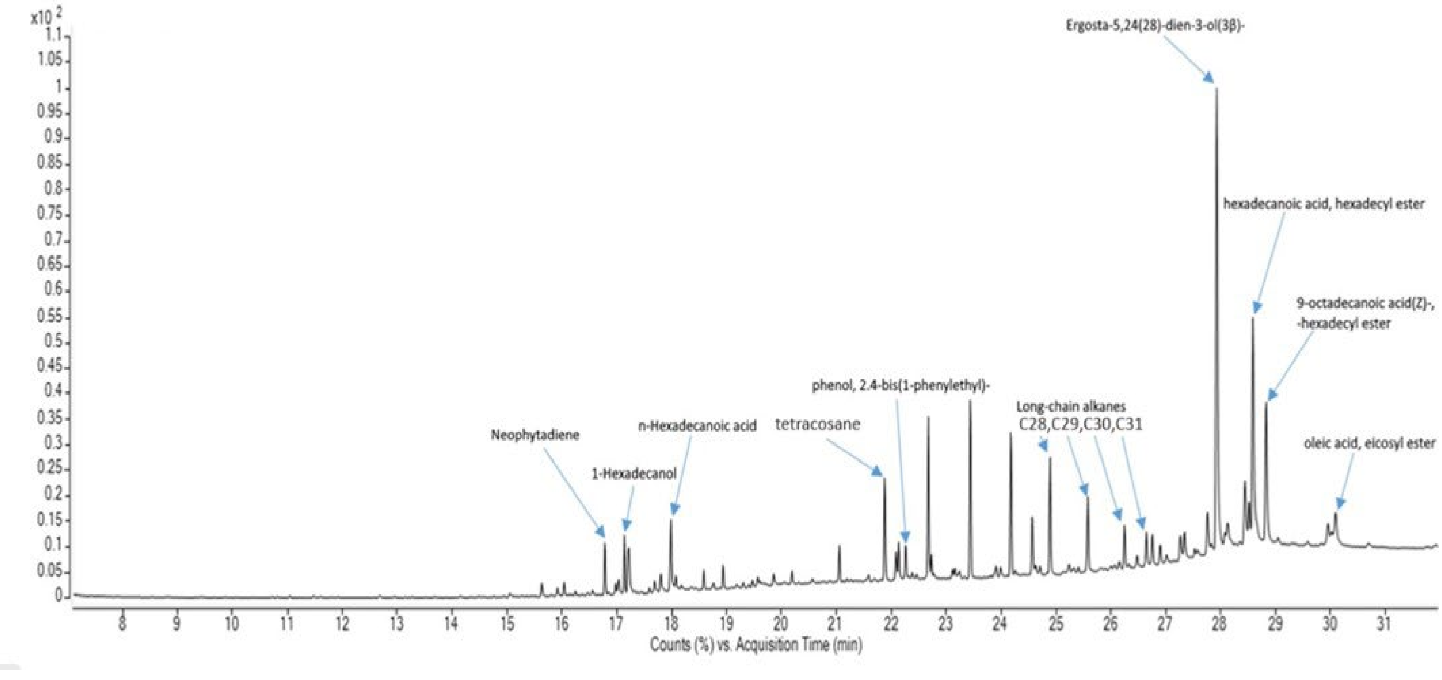


Figure 4.4.2F

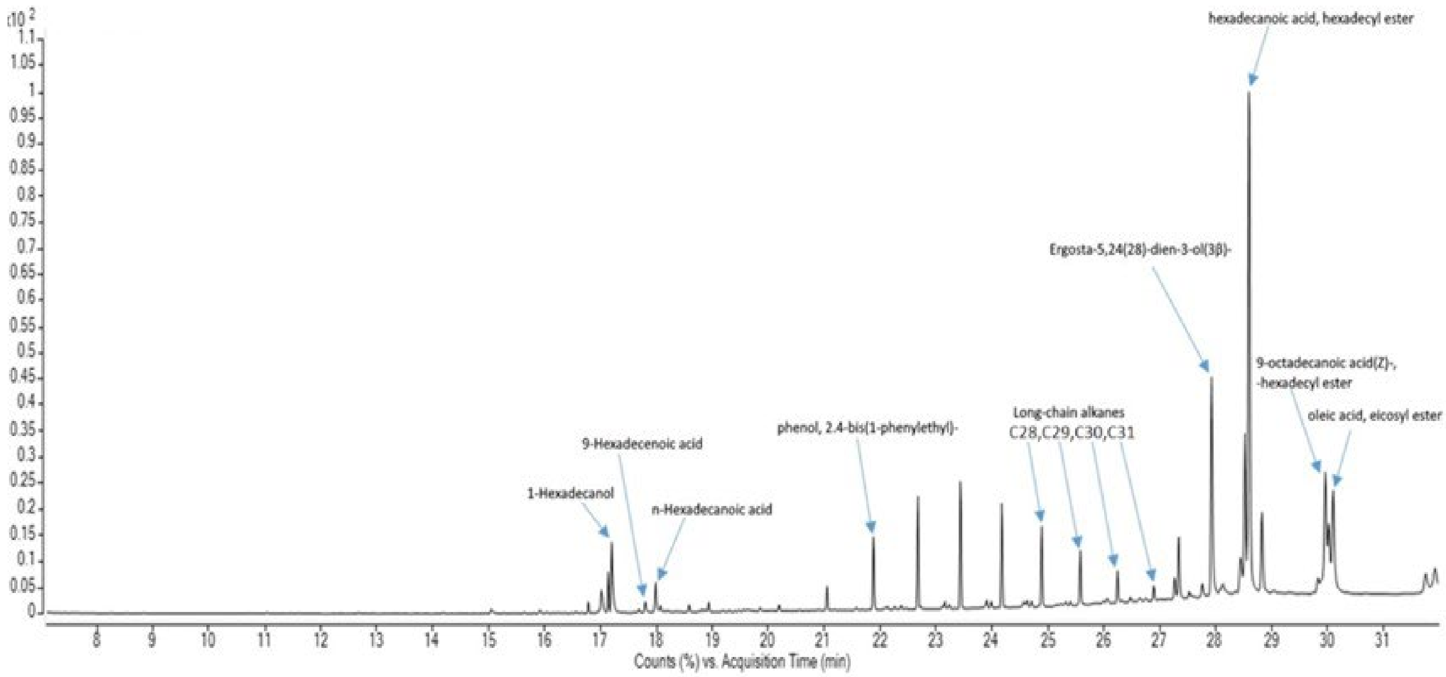


Figure 4.4.2G

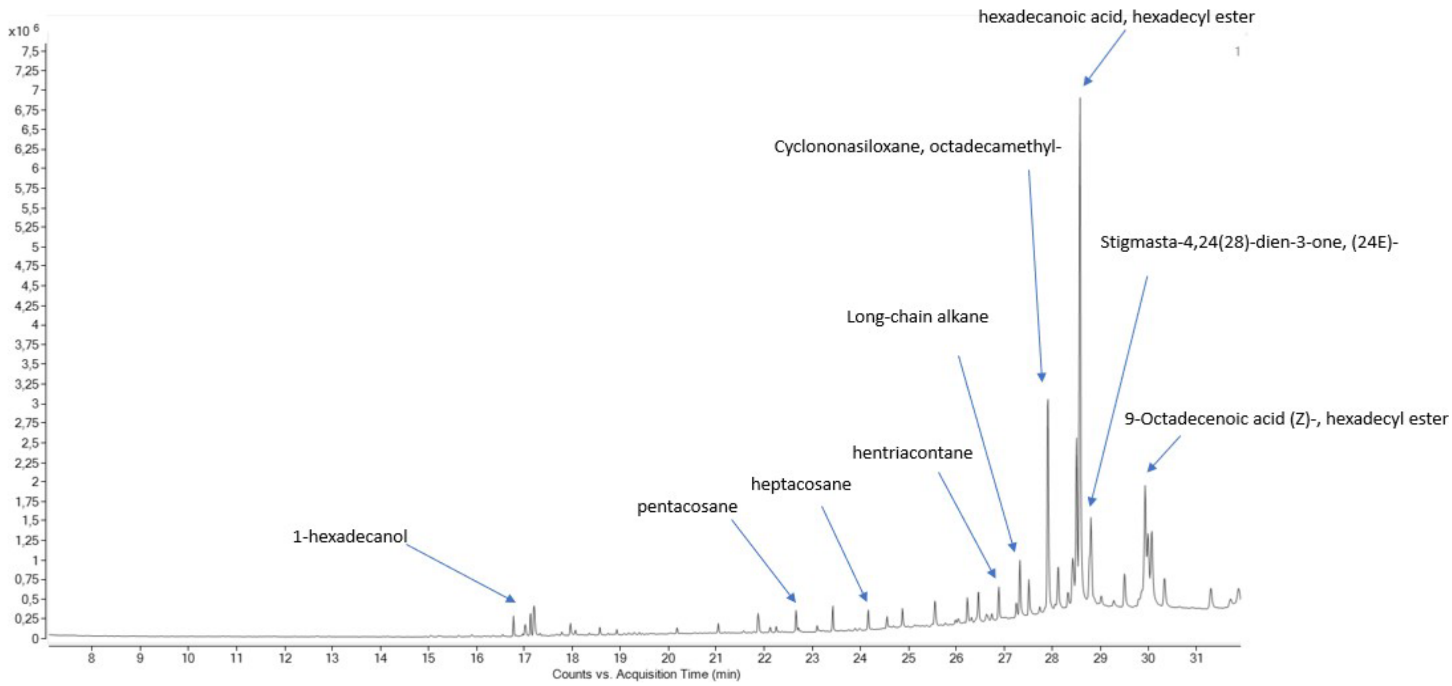


Figure 4.4.2H

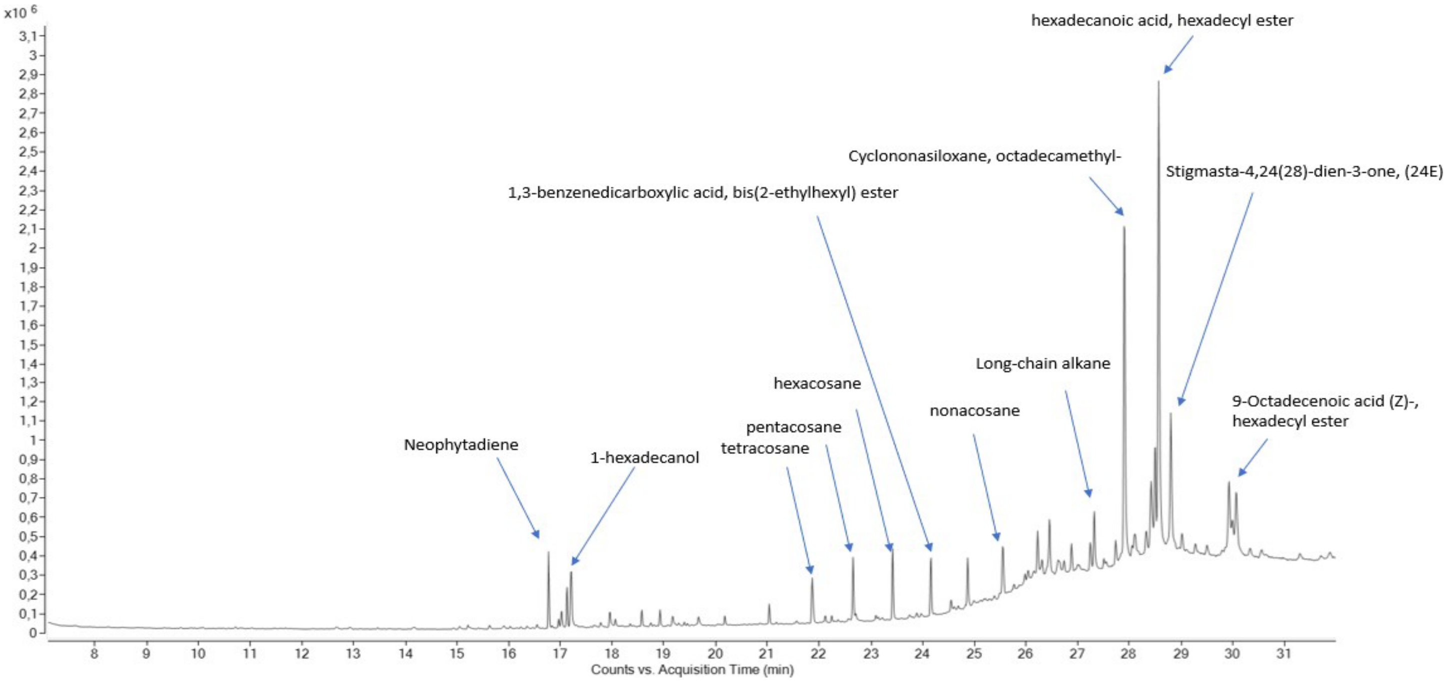
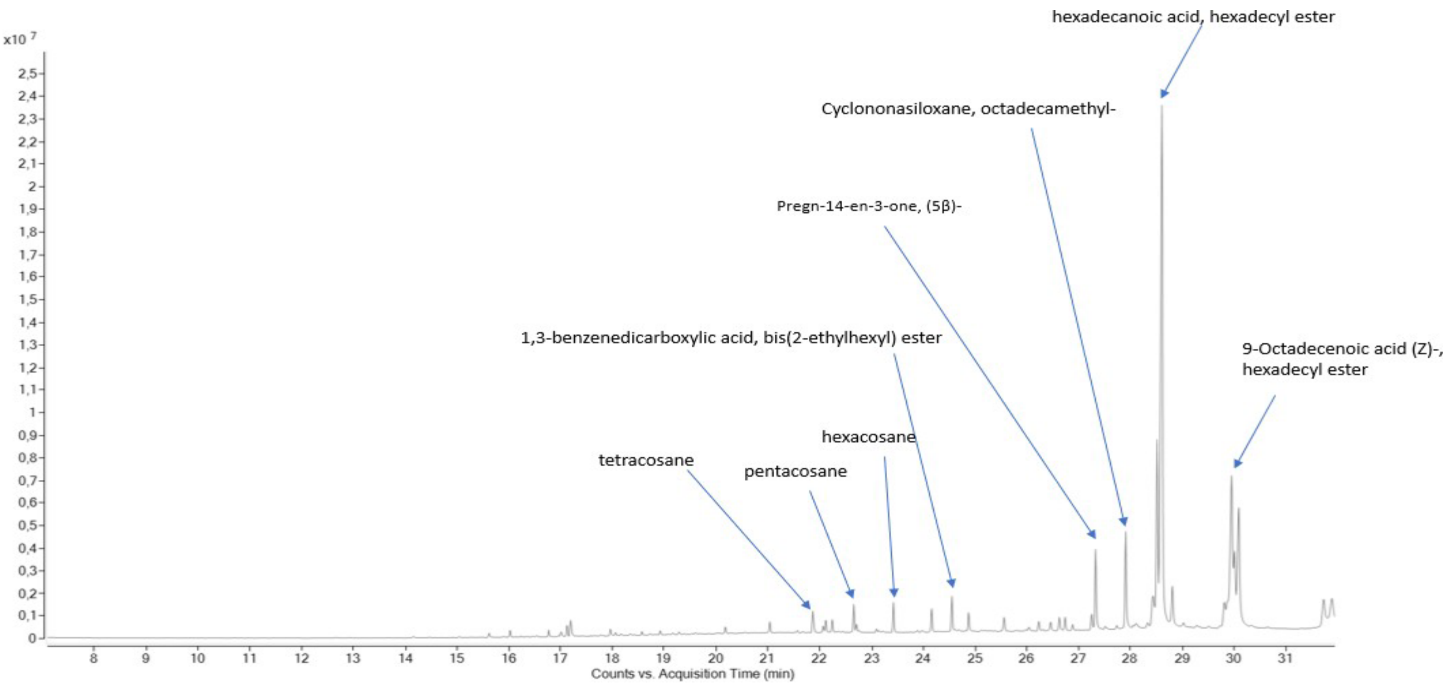
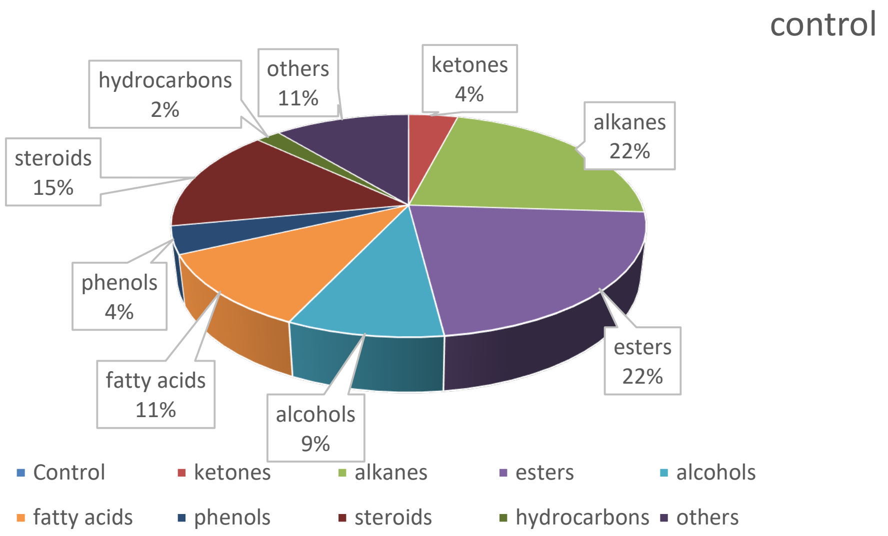


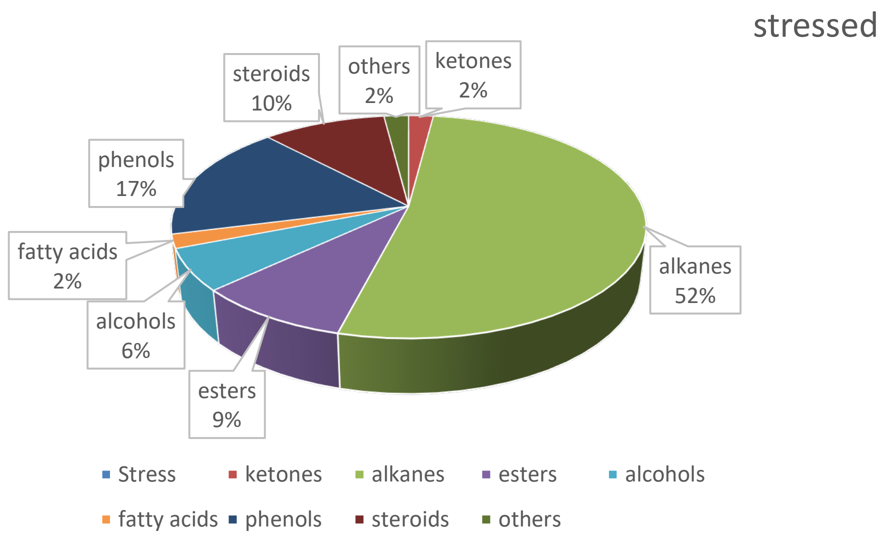
Figure 4.4.2I



Non-polar Metabolite Profiling –
To evaluate the biochemical impact of thermal stress, a comparative statistical analysis was performed between the control group (n=3) and the thermally stressed cohort (n=6). The non-polar metabolites identified through GC-MS were categorized into eleven distinct chemical classes to facilitate a comprehensive characterization of the coral's lipid and secondary metabolite profile. These classes include ketones, alkanes, esters, fatty acids, alcohols, steroids, phenols, hydrocarbons, aldehydes, and alkenes. Minor constituents and metabolites present in trace amounts were grouped into a collective "Others" category to streamline the comparative analysis. The quantitative shifts and distribution patterns observed between the control and stressed conditions are detailed in the following graphical representations.

Figure 4.4.2J
Non polar metabolites identified in control samples and stressed samples





Results of Non-Polar Metabolite GC-MS Analysis –
The untargeted GC-MS profiling of coral samples identified a complex array of non-polar metabolites, which were categorized into 11 distinct chemical classes. A total of 9 samples were evaluated, consisting of 3 control samples (maintained at 25°C) and 6 stressed samples (warmed to 30°C). Significant quantitative shifts were observed in the metabolic profiles between the two conditions:

- Alkanes: Represented the most dramatic increase, rising from 22% in control samples to 52% under thermal stress.

- Phenols: Also showed a marked increase, nearly quadrupling from 4% in controls to 17% in stressed corals.
- Esters: Experienced a significant "collapse," dropping from 22% in control conditions to just 9% after stress exposure.
- Fatty Acids: Decreased substantially from 11% to 2%.
- Steroids: Declined from 15% of the total profile in controls to 10% in stressed samples.
- Ketones: While present at 4% in control samples, ketones were entirely eliminated and "completely disappeared" in the stressed group.

The observed metabolic shifts indicate a profound biochemical reorganization in corals responding to thermal anomalies. The substantial increase in alkanes and phenols, alongside the drastic reduction in fatty acids and esters, suggests a potential breakdown of lipid-based structures or a shift toward the production of stress-related secondary metabolites. The decrease in steroid levels (from 15% to 10%) is particularly noteworthy. Steroids are recognized in marine literature as key indicators of physiological stress. For instance, Fel et al. (2019) demonstrated that specific steroids serve as sensitive biomarkers for chemical stress in corals, such as exposure to UV filters found in sunscreens. The reduction observed here under thermal stress suggests that temperature alone can disrupt steroidogenesis or accelerate the degradation of these bioactive compounds. The total disappearance of ketones further supports the hypothesis that high temperatures (30°C) inhibit certain metabolic pathways entirely.

Conclusions –

This research demonstrates that thermal stress significantly alters the non-polar metabolome of S. pistillata and P. damicornis. The increase in alkanes and phenols, combined with the loss of ketones and reduction in steroids, provides a "metabolic fingerprint" of heat stress. While these results are preliminary, they highlight the sensitivity of coral biochemical pathways to rising sea temperatures. Future studies utilizing internal standards will be essential to precisely quantify these changes and better protect these delicate ecosystems from the mounting threats of human-induced climate change.

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Chapter 5

Microfibers pollution in the UAE: from the Arabian Gulf to the desert.

Abstract
Atmospheric transport of microfibers is increasingly recognized as a potential pathway for contaminating remote and arid environments, yet knowledge about their occurrence in desert ecosystems remains limited. In this study, we conducted a preliminary survey spanning a gradient from the Arabian Gulf coast to inland deserts in the United Arab Emirates (UAE), sampling both beach and desert sand sediments at five representative locations: the beaches of Fujairah, Ajman and Umm Al Quwain, and the desert sites of Mleiha and Sharjah. Micro-FTIR analysis revealed the presence of various types of microfibers, with the majority being cellulosic/rayon. The highest microfiber concentration was observed in the desert of Mleiha, Sharjah (570 ± 35 kg^{-1} dry weight), whereas the lowest concentration was found on the beach of Fujairah (180 ± 20 kg^{-1} dry weight). This distinction of concentration between beach sediments along both the Arabian Gulf and the Gulf of Oman and desert sites strongly support the hypothesis of long-range aeolian transport driven by the prevailing northwesterly Shamal winds, rather than solely local deposition. Thus, although vehicle traffic and tourism may enhance microfiber contamination near accessible dunes, the consistent presence and higher abundance of fibers in remote desert sediments indicate that atmospheric transport is the primary driver of microfiber redistribution across the UAE landscape. Average diameter of the fibers were 14 micrometers (range 8-50 µm), and average length was 536 micrometers (range 50-2500 µm). Prevailing colours white/ transparent (90%) blue (9%) and few brown (1%). Statistical analysis did not highlight any significant difference in microfiber concentrations between beach and desert sites (alpha, 0.05 and P value 0.121); or in the microfiber count within each replicate (low Coefficient of variation). The findings suggest that wind-mediated transport plays a major role in the deposition of microfibers in desert environments.

Key words – Microplastic, fibers, Marine litter, FT-IR/FT-NIR, Arabian Gulf, UAE Desert

1. Introduction -
Microplastic (MP) contamination has emerged as one of the most critical and widespread environmental concerns of the contemporary era, fuelled by the rapid rise in plastic production since approximately 1950 (Geyer et al. 2017). Evidence suggests that MP contamination can be found even in remote environments such as mountains, deserts, oceans and virtually all ecosystems (Wang et al. 2021; Saliu et al. 2023; Abayomi et al. 2017; Baalkhuyur et al. 2020; Zhang et al. 2020; Alarif et al. 2023; Aslam et al. 2020; Elsergany et al. 2023; Allen et al. 2019). Multiple studies indicate that MP contamination in remote environments can be attributed to wind-driven and atmospheric transport (Wang et al. 2021; Chandrakanthan et al. 2023; Pu et al. 2024). In terms of morphology, microplastics display diverse shapes ranging from regular spherical or cylindrical pellets to fibres; physicochemical degradation enables the disintegration of larger particles into smaller irregular fragments (Rosal 2021). Recent evidence indicates that plastic microfibers represent the dominant fraction of microplastics in numerous environments and may constitute the largest contribution to microplastic contamination overall (Boucher and Friot 2017). Moreover, cellulosic/rayon fibres have been found to be prevalent in several environmental compartments including the sea surface, seafloor, rivers, lakes and mountains (Saliu et al. 2023; Abayomi et al. 2017; Baalkhuyur et al. 2020; Zhang et al. 2020; Alarif et al. 2023; Aslam et al. 2020; Elsergany et al. 2023; Wang et al. 2021; Allen et al. 2019). Among the different morphologies identified in microplastic research, fibres appear to be ubiquitous, with confirmed presence even in remote environments such as Antarctic ice (Jones-Williams et al. 2025). This has led to growing recognition that microfibers should be considered a distinct category of microplastic pollution, given their unique sources, transport pathways and environmental behaviour (Liu et al. 2019).
In particular, the atmospheric transport of microplastic fibres (MPFs) remains a poorly characterised, yet increasingly recognised, pathway for contamination of remote terrestrial ecosystems. While numerous studies have addressed microfiber pollution in marine and coastal environments, the fate of microfibers in arid regions such as deserts remains largely unknown. However, owing to their small size, low density and

aerodynamic properties, microfibers can remain airborne for extended periods and be transported over long distances from urban and coastal sources into remote areas (Wang et al. 2021; Chen et al. 2023). Evidence of such transport was documented during dust events in Arizona, where MPFs were found to be preferentially transported by wind compared to other plastic fragments, achieving high deposition rates in desert soils (Chandrakanthan et al. 2023). Studies report that fibres with aspect ratios of 20 and 50 exhibit 157 ± 26% and 272 ± 50% greater atmospheric transport distances, respectively, than spherical fibres of equivalent volume (Tatsii et al. 2023). Furthermore, Xiao et al. (2023) demonstrated experimentally that flat microfibers exhibit longer residual periods in the atmosphere and can travel considerable distances before deposition. The mean aspect ratio of the fibres found in the present study is approximately 39 (Table 5 in chapter 5), which is consistent with the dimensional characteristics considered suitable for long-range atmospheric transport.

The United Arab Emirates (UAE), with its extensive coastline, rapid urbanisation and arid desert interiors, represents a compelling natural laboratory in which to study the airborne dispersal of microfibers. Previous work in the Arabian Gulf has documented significant microplastic contamination in coastal sediments, marine organisms and aerosols (Baalkhuyur et al. 2020; Uddin et al. 2020), but these studies largely focused on fragments, films and industrial plastics rather than fibres. Evidence from Kuwait and along the UAE coastline suggests that airborne microplastics, particularly ultrafine fibres (< 0.5 µm), can be detected in aerosols and may be mobilised during dust storms or seasonal wind events (Pu et al. 2024; Al-Salem et al. 2020). The question of microfiber pollution in UAE environments is closely tied to prevailing winds and dust dynamics. Coastal cities such as Dubai are major sources of airborne debris, and the dominant northwesterly-to-southeasterly winds mean that microplastic fibres released in coastal urban areas can be carried inland by the Shamal. Dubai sits on the Arabian Gulf coast; when northwesterly winds blow, the downwind direction is towards the UAE's south-eastern deserts, effectively transporting microfibers from Dubai's atmosphere or shoreline tens of kilometres inland (Paparella and Burt 2023; Nazzal et al. 2019). However, there is virtually no information on microfibers in desert sediments of the UAE, and the potential gradient from coastal to inland environments has not been systematically investigated.

In this context, the present study aims to provide a preliminary assessment of microfibers along a gradient from coastal beaches to desert sands in the UAE. Five locations, ranging from urban coasts to remote deserts, were sampled to determine the prevalence, composition and concentration of microfibers. By comparing coastal and desert samples, the objective was to investigate whether microfibers accumulate differently in inland regions and to provide insights into the dominant transport mechanisms.

2. Materials and methods -

2.1. Sampling location -

The study was conducted during March 2025. Five distinct locations in the UAE were selected for sampling: three in beach areas and two in desert environments. The chosen locations were Umm Al Quwain, Sharjah, Ajman and Fujairah. In total, five samples were analysed: two from desert sites (sample IDs S3.D.R01 and S5.D.R01) and three from coastal sites (sample IDs S1.C.R01, S2.C.R01 and S4.C.R01). Details of sampling locations are presented in Table 2 in chapter 5. The geographical positions of all sites are illustrated in Figures 1a and 4b, with satellite views provided in Figure 4b in chapter 5.

Figure 1 **–** Photographs of sampling locations and map showing the position of all five sampling sites within the UAE. (a) Umm Al Quwain beach; (b) Fujairah beach; (c) Map of sampling locations within the UAE.

Table 2 - Details of sampling location, including site type, coordinates and sample IDs.

Location	**Sample ID**	**Latitude and Longitude**	**weight in gm per falcon**	**Habitat**
Umm Al Quwain	S1.C.R01	25°34'08.2"N 55°32'57.9"E	~75	Beach
Ajman	S2.C.R01	25°25'41.3"N 55°29'45.5"E	~75	Beach
Sharjah	S3.D.R01	25°10'56.3"N 55°43'08.3"E	~75	Desert
Fujairah	S4.C.R01	25°07'57.4"N 56°21'22.2"E	~75	Beach
Mleiha, Sharjah	S5.D.R01	25°09'18.2"N 55°52'11.4"E	~75	Desert

2.2 Sampling procedure -

Sediment samples were collected following a procedure previously published by Saliu et al. (2018). Briefly, approximately the top 1 cm surface layer was collected from a 50 cm × 50 cm grid using a small stainless steel shovel, and transferred into Falcon tubes. Each Falcon tube contained 45 ml of sediment, corresponding to an average weight of approximately 75 g (Table 2 in chapter 5). For each location, analyses were carried out in triplicate. Both field and laboratory operations were conducted under clean conditions; the use of plastic materials was minimised wherever possible; and all glassware and laboratory tools used for sample manipulation were pre-washed with distilled water and dried prior to analysis.

2.3 Micro-FTIR analysis -

Samples were analysed in the laboratory of the University of Milano-Bicocca using a stereomicroscope for particle and fibre selection and micro-FTIR for chemical characterisation. All potential plastic particles visible to the naked eye were collected from the sand using forceps and transported to the laboratory for further analysis. Submillimetre microplastic particles (< 1 mm) were analysed using a μ-FTIR instrument

Spotlight 200i (PerkinElmer), equipped with a liquid nitrogen-cooled mercury cadmium telluride (MCT) single detector. Spectra were collected in reflectance mode, with 32 co-added scans per analysis point, a scan range of 3,600–700 cm^{-1} and a resolution of 4 cm^{-1}. Micro-FTIR analysis was performed in a clean room (ISO standard 14644, ISO class 6) with 150–240 air changes per hour and 293 particles/m^3 $\geq$ 5 µm, with appropriate personal protective equipment. A background scan against air was performed prior to each sample measurement. Spectral analysis was conducted using the COMPARE™ algorithm, facilitating searches within a commercially available spectral library. Only spectral matches with a confidence level of 60% or higher were accepted. Rayon and cellulosic fibres were categorised together as a single group (rayon/cellulose), owing to the well-documented difficulties in discriminating between these two polymers using FTIR, particularly when aged or weathered (Lusher et al. 2014; Peeken et al. 2018; Suaria et al. 2020).

Figure 2 **–** Stereomicroscope images of sand sediment samples: (a) S1.C.R01 Umm Al Quwain; (b) S2.C.R01 Ajman; (c) S3.D.R01 Sharjah; (d) S4.C.R01 Fujairah; (e) S5.D.R01 Mleiha Sharjah desert.

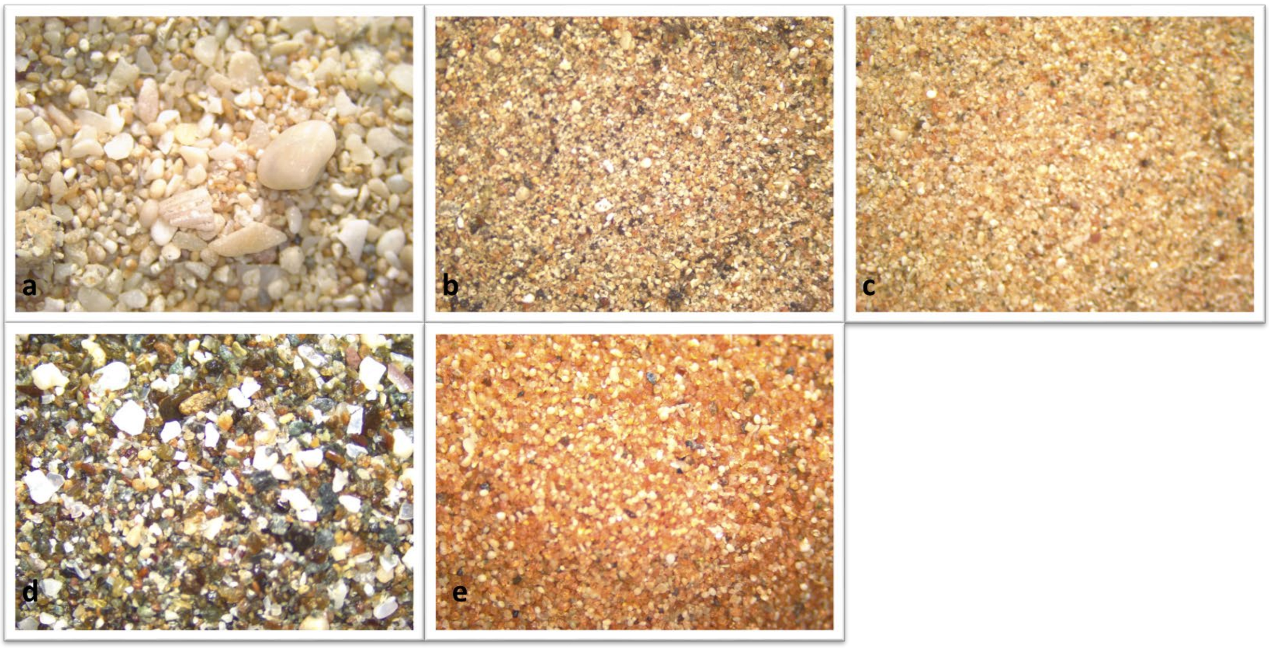


Figure 4ci – Micro-FTIR images of identified microfibers alongside their matching spectral library scans.

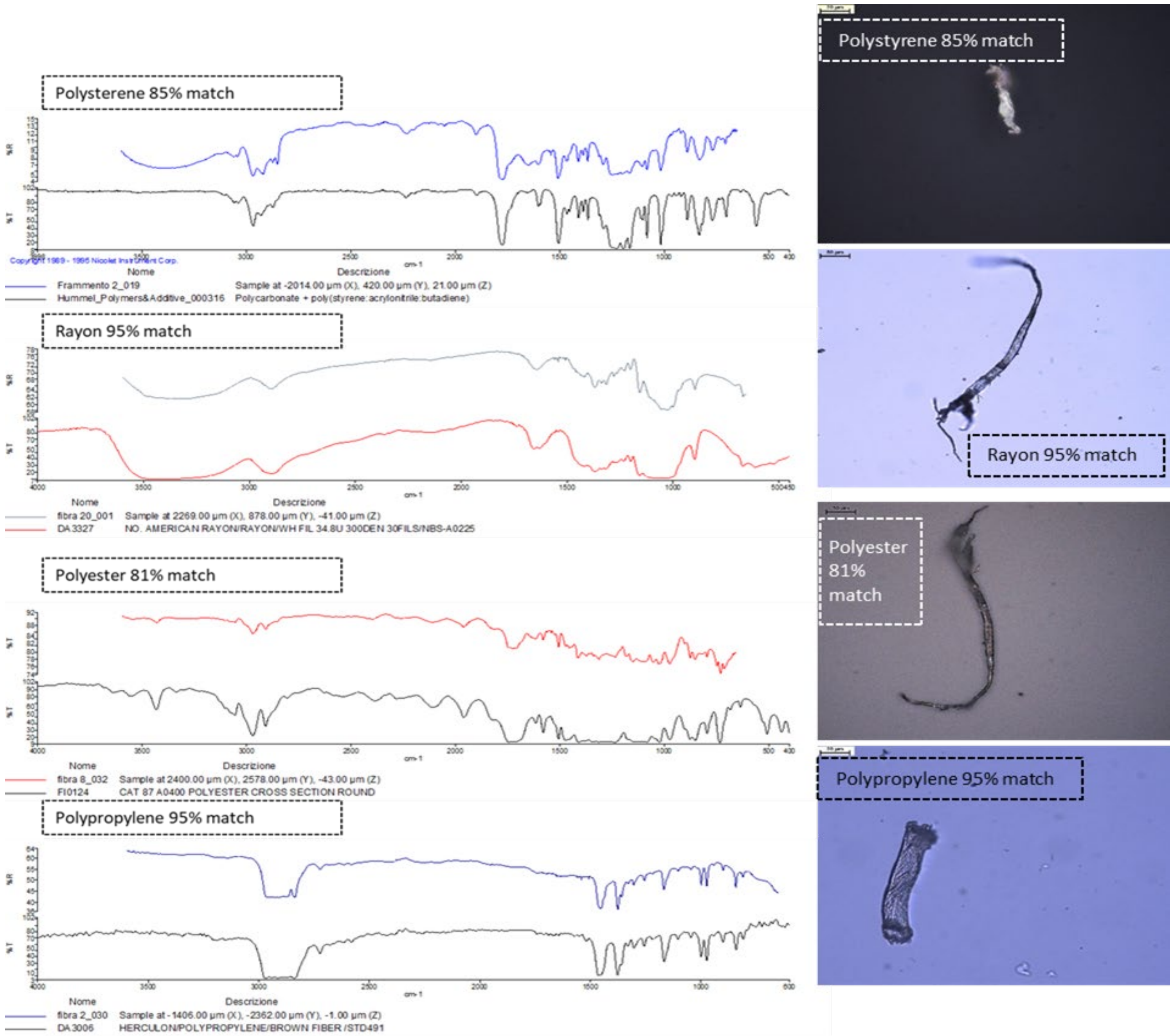


Figure 4cii – Enlarged microscope images of detected fibres. Scale bar: 20 µm corresponds to 500× magnification; 50 µm corresponds to 200× magnification.

2.4 Quality control -

To ensure quality control and avoid contamination several precautionary measures were adopted. Field and Lab operations were carried out in clean conditions; the use of plastic was avoided as much as possible. All glassware and laboratory tools used for sample manipulation were pre-washed with distilled water and dried before the analysis. Micro-FTIR analysis was performed in a certified clean room environment (ISO class 6) to minimise the risk of airborne fibre contamination during sample processing.

2.5 Statistical analysis -

All collected data were expressed as mean ± standard deviation (SD). Data normality was assessed using the Shapiro-Wilk test. Since the normality assumption was not met, the non-parametric Kruskal-Wallis test was applied to investigate differences in microplastic and microfiber concentrations between the samples. Principal Component Analysis (PCA) was performed to explore potential patterns and associations among environmental variables and microplastic concentrations. Given the non-normal distribution of most variables and the relatively small sample size, PCA was conducted on the Pearson correlation coefficients. Values were considered statistically significant at $p < 0.05$. All analyses were performed using XLSTAT (XLSTAT, Lumivero, Denver, USA).

3. Results and discussion -

3.1 MPFs identification and characterization -

From the five collected samples, 214 fibres were retrieved, identified and analysed, of which 195 yielded a positive match with the reference spectral library (Table 4 in chapter 5). Considering fibre concentrations, the highest value was recorded at sample site 5 (S5.D.R01: 570 ± 35 fibers kg^{-1} dry weight) and the lowest at sample site 4 (S4.C.R01: 180 ± 20 fibers kg^{-1} dry weight). The greatest absolute number of fibres was observed in sample S5.D.R01 (57 items), while the lowest was in S4.C.R01 (18 items). Among the library-matched fibres, 177 were rayon/cellulose, 8 polypropylene, 4 polyester, 4 nylon, 1 polystyrene and 1 polyurethane (Table 4 in chapter 5). The mean length of the detected fibres across all samples was 536 (SD ± 498) µm and the mean diameter was 14 (SD ± 6) µm (see Figures 5 and 6 for kernel density distribution plots in chapter 5, and Table 5 in chapter 5 for per-site dimensional data).
In beach samples (S1.C.R01, S2.C.R01 and S4.C.R01), the highest fibre concentration was recorded at S1.C.R01 (430 ± 15 fibers kg^{-1} dry weight) and the lowest at S4.C.R01 (180 ± 20 fibers kg^{-1} dry weight). The majority of fibres were rayon/cellulose and white/transparent in colour (Figure 7). Although polypropylene, nylon, polyester and polystyrene were also detected, their quantities were low (Table 4 in chapter 5). In desert samples (S3.D.R01 and S5.D.R01), fibre concentrations were higher: 550 ± 20 fibers kg^{-1} dry weight (S3.D.R01) and 570 ± 35 fibers kg^{-1} dry weight (S5.D.R01). As in the beach samples, the dominant fibres were rayon/cellulose and white/transparent in colour. Minor quantities of polypropylene, nylon and polyurethane were also detected (Table 4 in chapter 5). Statistical analysis indicated no significant difference in microfiber contamination levels between coastal and desert samples (Kruskal–Wallis test: $p = 0.121$, $\alpha = 0.05$).
The widespread occurrence of microfibers, and the dominance of cellulosic/rayon types, is consistent with findings from previous studies in arid UAE environments (Habib et al. 2022). It is important to note that prior studies in the UAE focused almost exclusively on synthetic fibres, excluding cellulosic/rayon types, which accounts for the considerably lower fibre abundance values reported previously. As researchers increasingly include cellulosic/rayon fibres in their analyses, the large prevalence of this category in environmental samples is becoming more widely recognised (Athey and Erdle 2022).

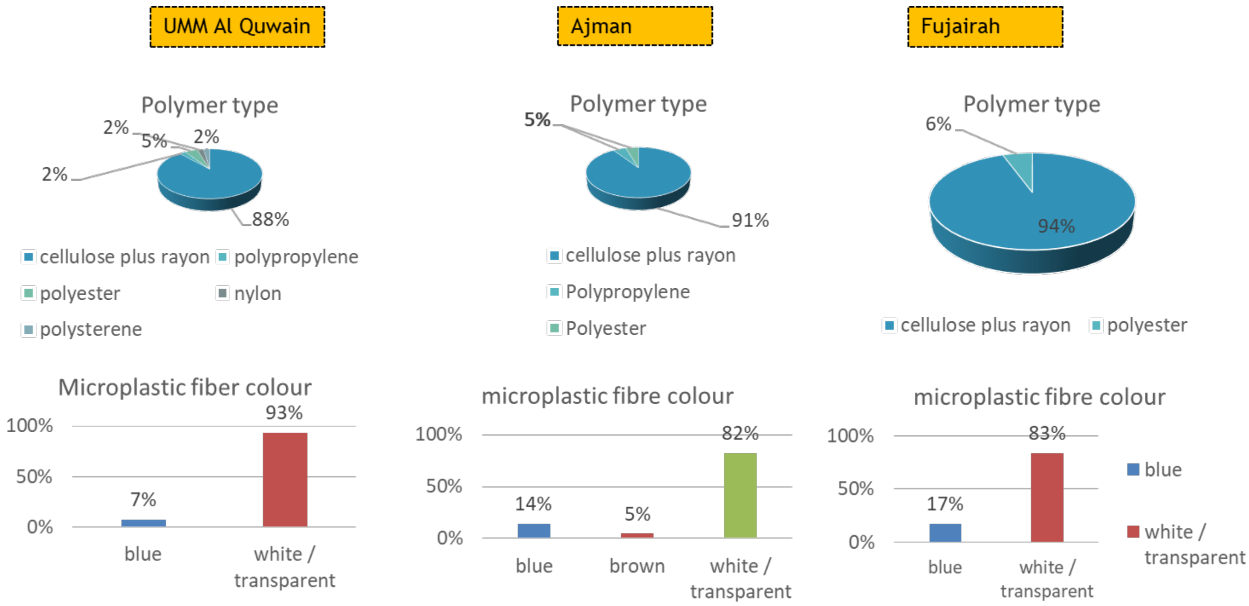
Table 3 –
Microfiber concentrations (fibers kg^{-1} dry weight) at each sampling site. Data expressed as mean ± SD.

Sampling location	**Sample ID**	**Total fibers (kg^{-1})**	**Synthetic fibers (kg^{-1})**	**Cellulosic/Rayon (kg^{-1})**
Umm Al Quwain	S1.C.R01	430±15	50±29	380±40
Ajman	S2.C.R01	220±21	20±12	200±21
Sharjah	S3.D.R01	550±20	50±29	500±35
Fujairah	S4.C.R01	180±20	10±6	170±25
Mleiha, Sharjah	S5.D.R01	570±35	50±29	520±44

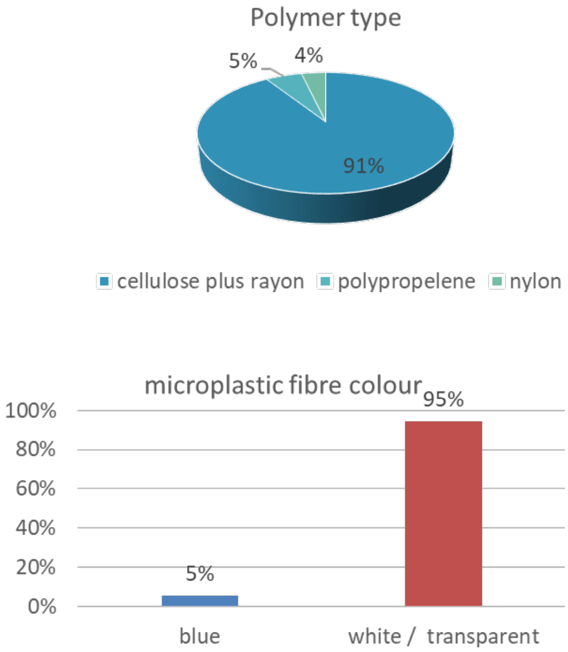
Table 4 –
Polymer composition of microfibers (expressed as percentage of total fibres detected).

Location	Sample ID	Non plastic fibers (rayon/ cellulose)	polypropylene	polyester	nylon	polystyrene	polyurethane
Umm Al Quwain	S1.C.R01	88%	2%	5%	2%	2%	-
Ajman	S2.C.R01	91%	5%	5%	-	-	-
Sharjah	S3.D.R01	91%	5%	-	4%	-	-
Fujairah	S4.C.R01	94%	-	6%	-	-	-
Mleiha, Sharjah	S5.D.R01	91%	5%	-	2%	-	2%

Figure 7 – Polymer type and colour composition of microfibers. Upper panel: beach samples (S1.C.R01, S2.C.R01, S4.C.R01). Lower panel: desert samples (S3.D.R01, S5.D.R01).



Sharjah



Mleiha Sharjah

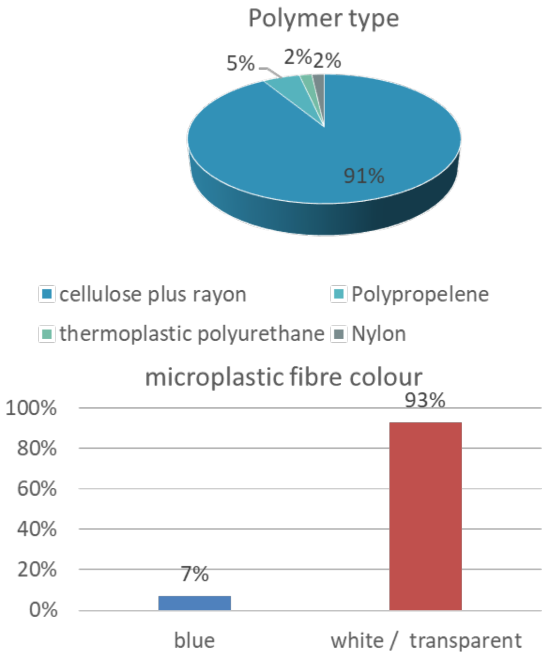


Table 5 – Average length and diameter of fibres (µm) per site, including aspect ratio (Length/Diameter).

Location	**Sample ID**	**Mean length (µm) ± SD**	**Mean diameter (µm) ± SD**	**Aspect Ratio (L/D)**
Umm Al Quwain	S1.C.R01	592 ±577	17±9	36
Ajman	S2.C.R01	676±620	17±10	40
Sharjah	S3.D.R01	535±434	13±3	40
Fujairah	S4.C.R01	427±551	11±2	40
Mleiha, Sharjah	S5.D.R01	477±350	12±4	39

Figure 5 – Kernel density estimate of fibre length distribution across all sampling sites. The density is normalised such that the integral over the domain equals unity.

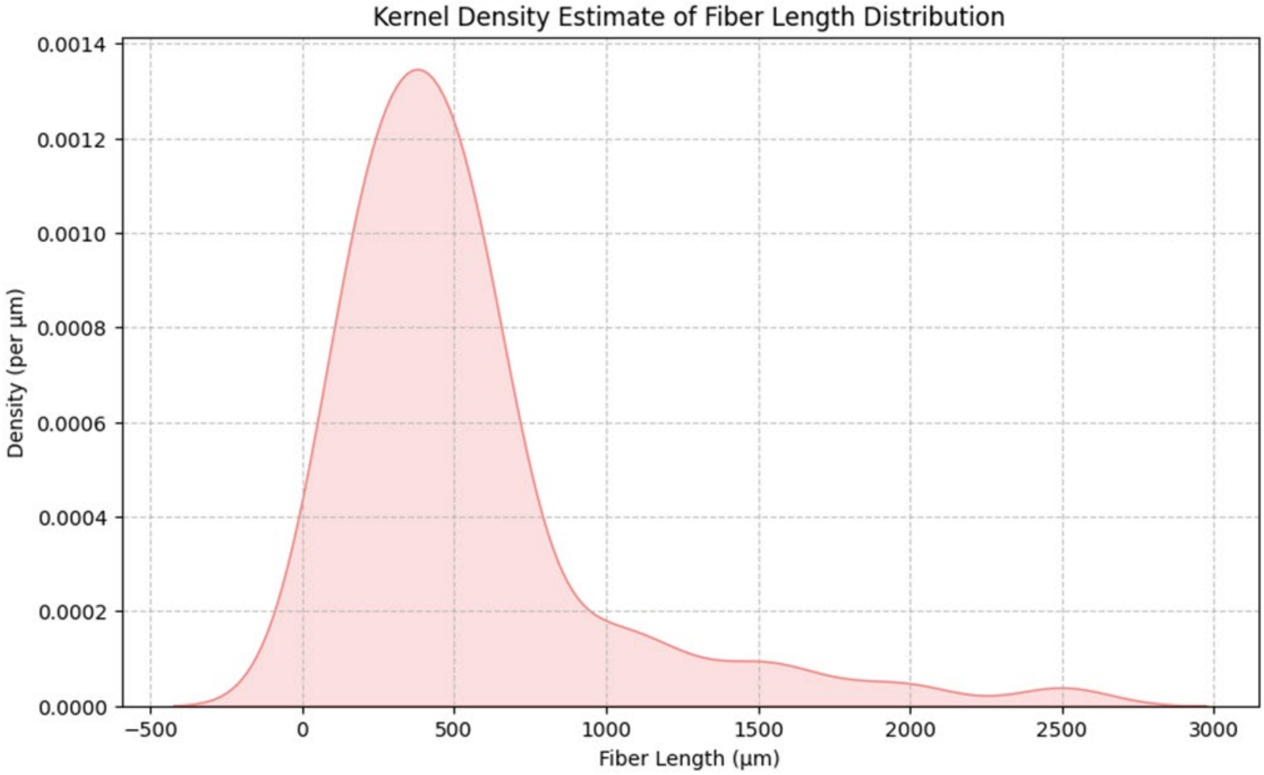
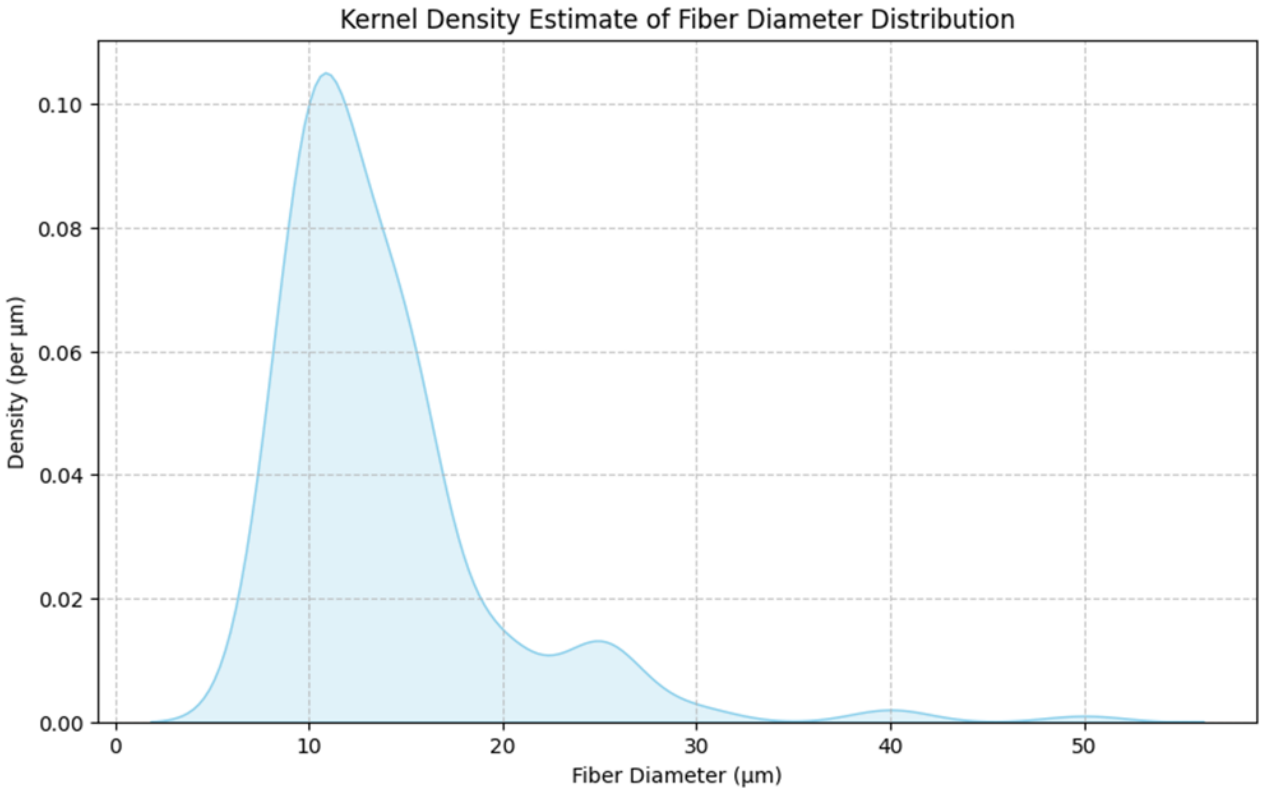


Figure 6 - Kernel density estimate of fibre diameter distribution across all sampling sites. The density is normalised such that the integral over the domain equals unity.



3.2 MPFs distribution -

The Kruskal–Wallis test applied to microfiber concentrations across all beach and desert samples showed no statistically significant difference between samples or replicates ($p = 0.121$, $\alpha = 0.05$; low coefficient of variation). Box plots comparing the distribution of values across beach and desert samples further confirmed this absence of significant difference (Figure 8). PCA was subsequently performed (Figures 9a and 9b), with the first two principal components (F1 and F2) together explaining 85.24% of the total variance. F1 is primarily associated with MPF abundance (17%), cellulosic/rayon fibre count (17%), non-cellulosic/rayon fibre count (16%), white/transparent fibre colour (17%) and distance from Mleiha (15%). F2 is primarily associated with mean fibre diameter (22%), mean fibre length (20%), distance from the Arabian Gulf coast (21%) and distance from the Indian Ocean coast (21%). Mleiha consistently emerges as a high-load inland site, and the dominance of MPF load rather than fibre size strongly indicates an Aeolian transport mechanism. The correlation matrix revealed a strong negative correlation between distance from the Arabian Gulf coast and mean fibre length and diameter ($r \approx -0.9$), implying that as distance from the sea coast increases, mean fibre length and diameter tend to decrease consistently — a pattern addressed in detail in Section 3.4.

Figure 8 – Box plots illustrating the distribution of microfiber concentrations across all sampling sites. No statistically significant difference was found between sites ($p = 0.121$, $\alpha = 0.05$). Sharjah and Mleiha exhibit the highest fibre concentrations.

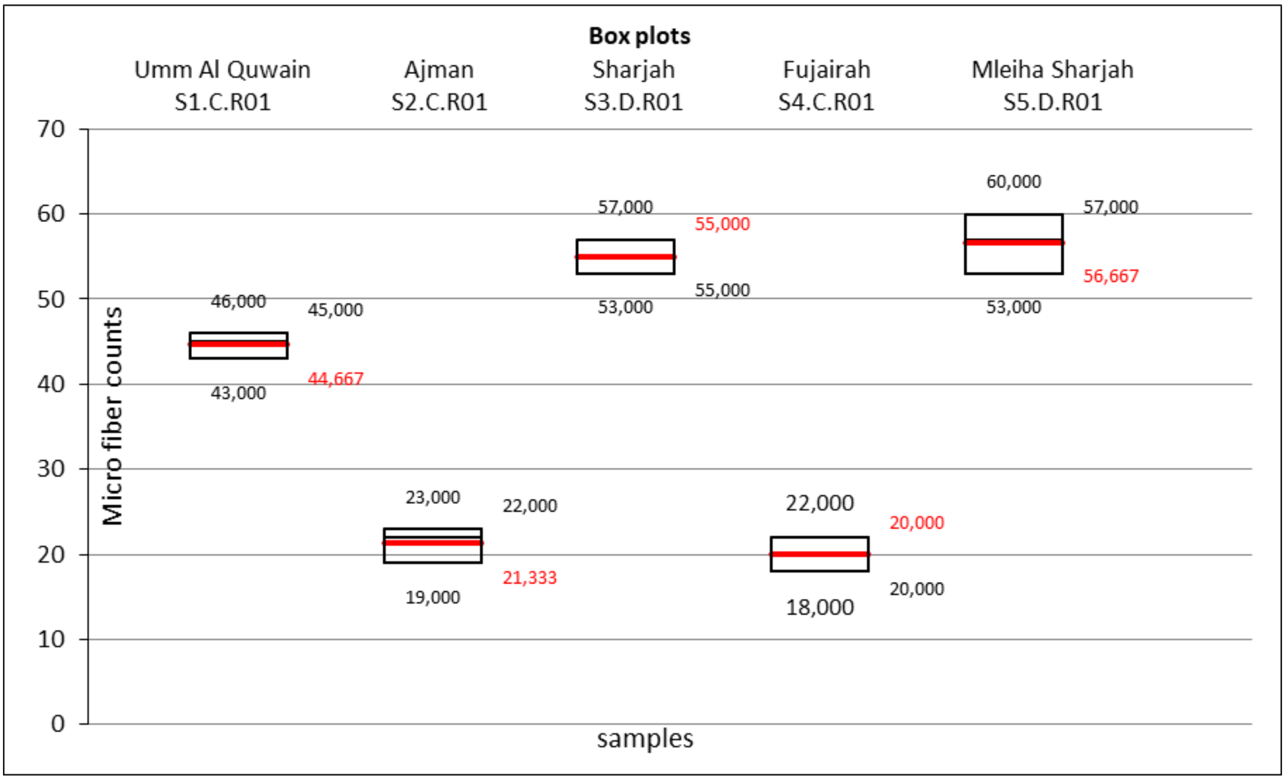


Figure 9a – Principal Component Analysis (PCA) biplot. The first two principal components (F1 and F2) together explain 85.24% of total variance. F1 is primarily associated with MPF abundance (17%), cellulosic / rayon fiber count (17%), non-cellulosic /rayon fiber count (16%), fiber colour of white / transparent (17%), distance from Mleiha (15%). F2 is primarily associated with Mean fiber diameter (22%), Mean fiber length (20%) distance from Arabian Gulf coast (21%) and distance from Indian Ocean coast (21%).

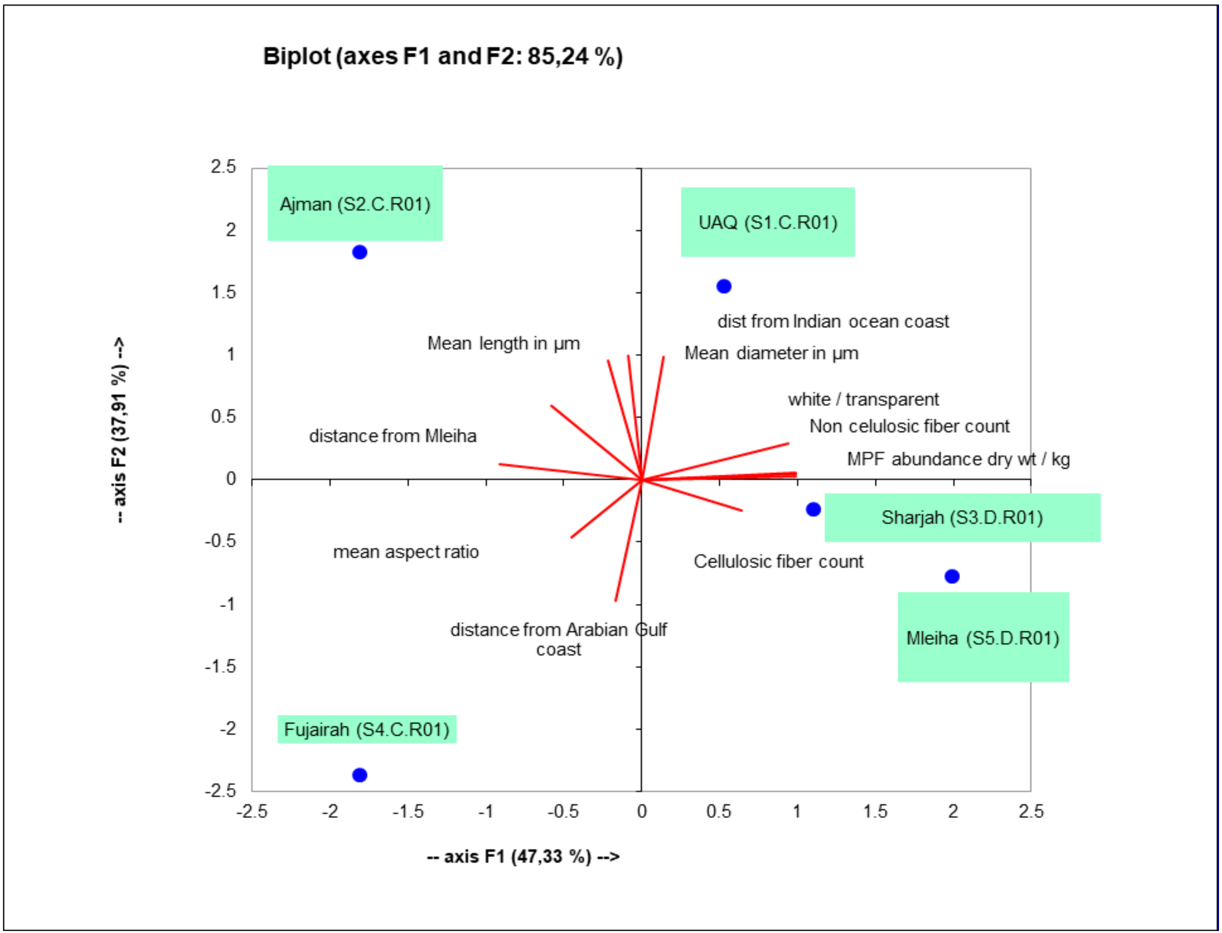


Figure 9b – Correlation matrix. Strong positive correlation between distance from the Indian Ocean coast and mean fiber length and diameter suggesting as the distance from the sea coast increases, the mean fiber length and diameter consistently tend to decrease. Similarly, there is very strong negative correlation between distance from Arabian Gulf coast and mean fiber length and diameter implying that as the distance from the sea coast increases, the mean fiber length and diameter consistently tend to increase.

Correlation matrix:

	Mean diameter in µm	Mean length in µm	mean aspect ratio	MPF abundance dry wt / kg	Cellulosic fiber count	Non celulosic fiber count	white / transparent	blue	brown	distance from Arabian Gulf coast	distance from Mleiha	distance from Indian ocean coast
Mean diameter in µm	1	**0.961**	-0.458	-0.031	-0.061	0.222	-0.038	-0.333	0.602	**-0.945**	0.227	**0.968**
Mean length in µm	**0.961**	1	-0.197	-0.134	-0.158	0.070	-0.144	-0.369	0.774	**-0.881**	0.258	**0.929**
mean aspect ratio	-0.458	-0.197	1	-0.380	-0.350	-0.614	-0.387	-0.083	0.351	0.548	0.099	-0.468
MPF abundance dry wt / kg	-0.031	-0.134	-0.380	1	**0.999**	**0.951**	**1.000**	0.553	-0.522	-0.195	**-0.909**	0.203
Cellulosic fiber count	-0.061	-0.158	-0.350	**0.999**	1	**0.939**	**0.999**	0.567	-0.526	-0.167	**-0.921**	0.176
Non celulosic fiber count	0.222	0.070	-0.614	**0.951**	**0.939**	1	**0.953**	0.401	-0.459	-0.416	-0.762	0.423
white / transparent	-0.038	-0.144	-0.387	**1.000**	**0.999**	**0.953**	1	0.534	-0.541	-0.183	**-0.899**	0.195
blue	-0.333	-0.369	-0.083	0.553	0.567	0.401	0.534	1	-0.250	0.010	-0.800	-0.178
brown	0.602	0.774	0.351	-0.522	-0.526	-0.459	-0.541	-0.250	1	-0.510	0.392	0.519
distance from Arabian Gulf coast	**-0.945**	**-0.881**	0.548	-0.195	-0.167	-0.416	-0.183	0.010	-0.510	1	0.055	**-0.971**
distance from Mleiha	0.227	0.258	0.099	**-0.909**	**-0.921**	-0.762	**-0.899**	-0.800	0.392	0.055	1	-0.015
distance from Indian ocean coast	**0.968**	**0.929**	-0.468	0.203	0.176	0.423	0.195	-0.178	0.519	**-0.971**	-0.015	1

In bold, significant values (except diagonal) at the level of significance alpha=0,050 (two-tailed test)

3.3 Large prevalence of cellulosic/rayon fibers within beach and desert sands -

MPF concentrations ranged from 180 ± 20 fibers kg^{-1} (S4.C.R01) to 570 ± 35 fibers kg^{-1} (S5.D.R01), with cellulosic/rayon fibres accounting for approximately 91% of all fibres detected (Table 4 in chapter 5). This finding aligns with previous studies in arid UAE environments, where cellulosic/rayon fibres constitute a significant portion of detected fibres in soils and atmospheric dust (Habib et al. 2022). Earlier studies in the UAE reported on synthetic fibres only, without accounting for cellulosic/rayon types, which explains the lower fibre abundance values reported previously. As researchers have begun to include this category, its large prevalence is becoming increasingly evident (Athey and Erdle 2022).
The presence of MP fibre contamination in remote environments has been reported in a limited number of studies. Wang et al. (2021) detected MPs ranging from 0.7 ± 1.5 to 11.7 ± 15.5 items/kg (mean 6.0 ± 15.4 items/kg) in a remote central Asian desert in China, and Taptiklis et al. (2025) report that contamination in such environments is dominated (99%) by fibrous MP shapes. Xiao et al. (2023) provide experimental data confirming that dry deposition of fibres with a diameter of 47 µm and density of 1,140 kg m^{-3} is feasible, consistent with the mean fibre diameter of 14 µm (range 8–50 µm) and the mean aspect ratio of ~39 recorded in the present study, both of which are characteristics suitable for long-distance atmospheric transport (Tatsii et al. 2023).

3.4 Source and transport of microfibers from the Arabian Gulf and in the UAE desert -

Microfiber contamination was investigated across a gradient from coastal to desert environments in the UAE, including sites along both the eastern (Gulf of Oman) and western (Arabian Gulf) coasts, as well as desert areas at increasing distance from the major urban centre of Dubai. Although one might expect a decline in contamination with distance from coastal cities, a surprising degree of homogeneity was observed, with desert samples showing comparable or even higher concentrations than coastal sediments. This pattern strongly supports the hypothesis of wind-driven transport from the north-western coast toward inland desert areas, consistent with the direction of the dominant Shamal winds (Paparella and Burt 2023). Approximately 60% of sand and dust reaching the UAE originates from the deserts of Iraq, Syria and Kuwait to the north-west, carried by Shamal winds (Nazzal et al. 2019). As the Arabian Gulf coast is an enclosed basin, onshore northwesterly winds trap and transport pollutants inland, creating MPF accumulation hotspots in the desert interior.

Figure 3a – Plot of monthly average wind speed and wind direction in Sharjah derived from ERA5 reanalysis data (Hersbach et al. 2020). The right y-axis indicates wind direction in degrees; values between 270° and 360° indicate a northwesterly origin.

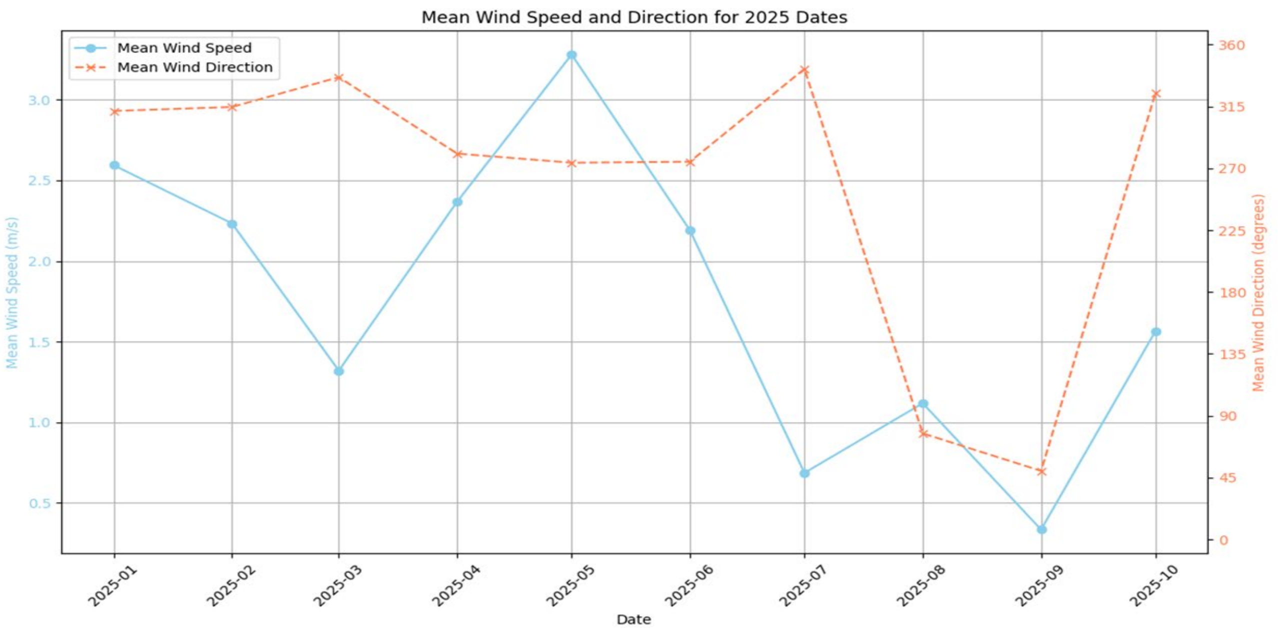


Figure 3b – Plot of monthly average wind speed at Sharjah International Airport, UAE, illustrating the prevalence of northwesterly Shamal wind flow. Data from METAR-TAF (2025).

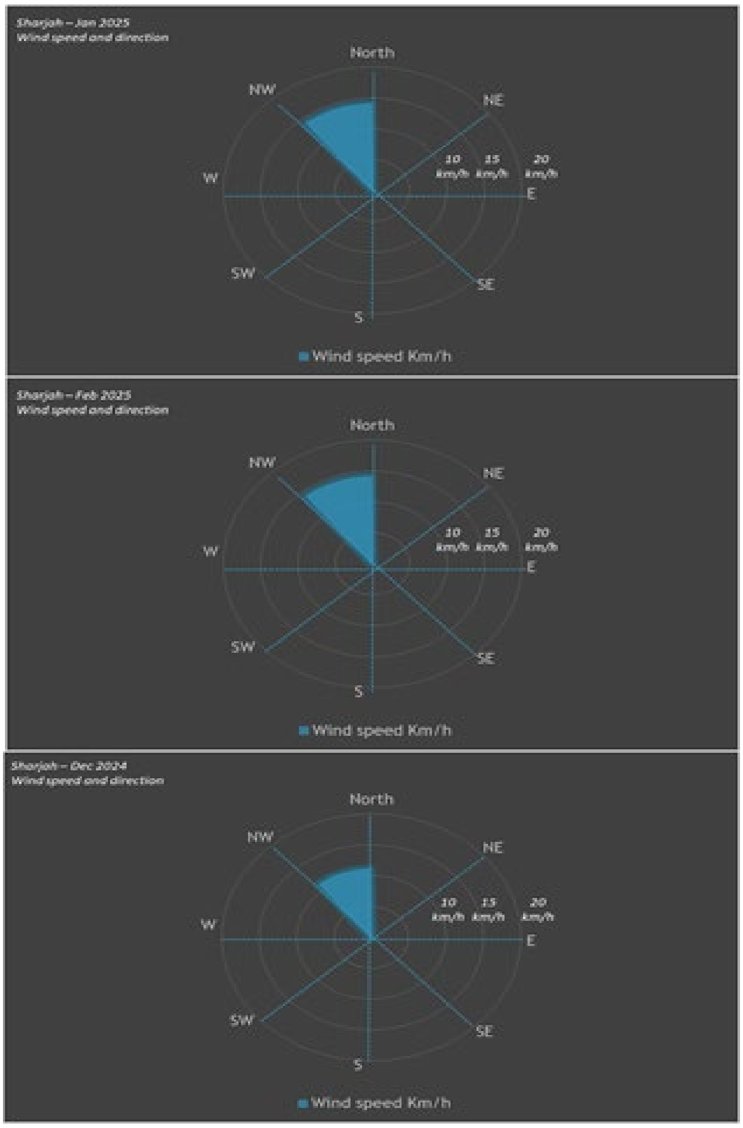
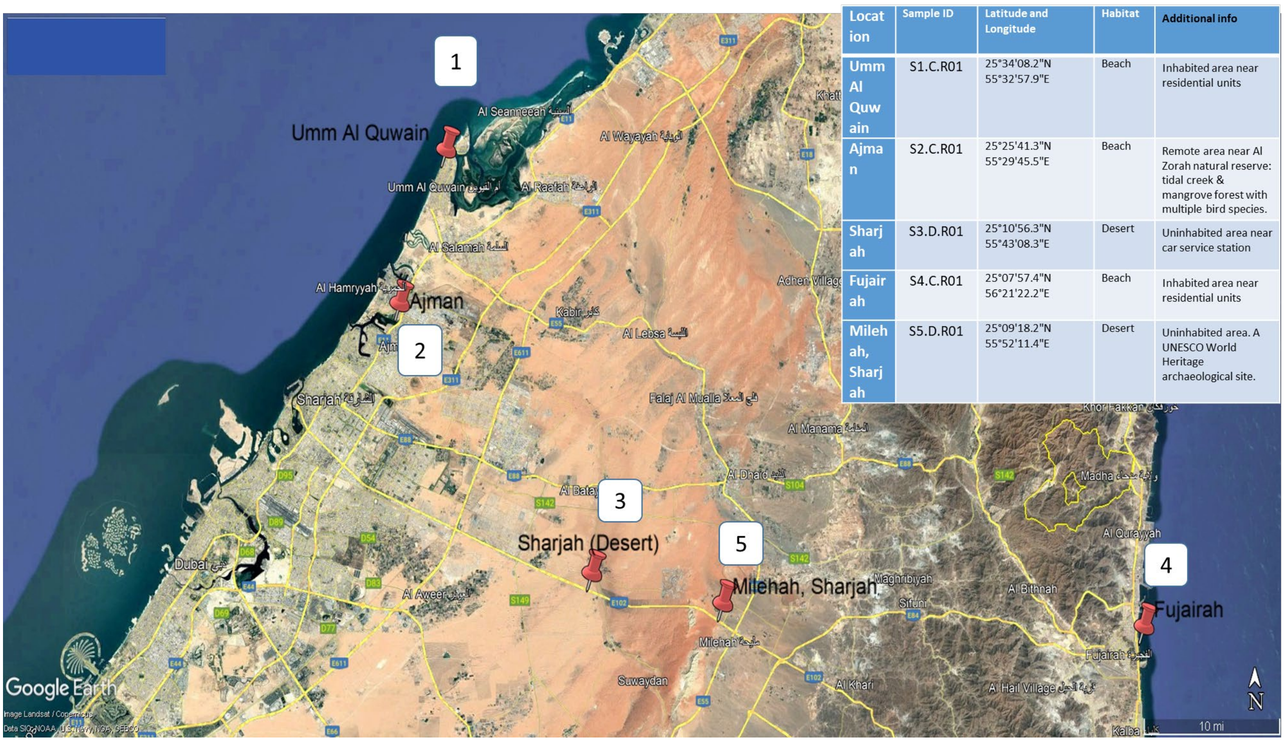
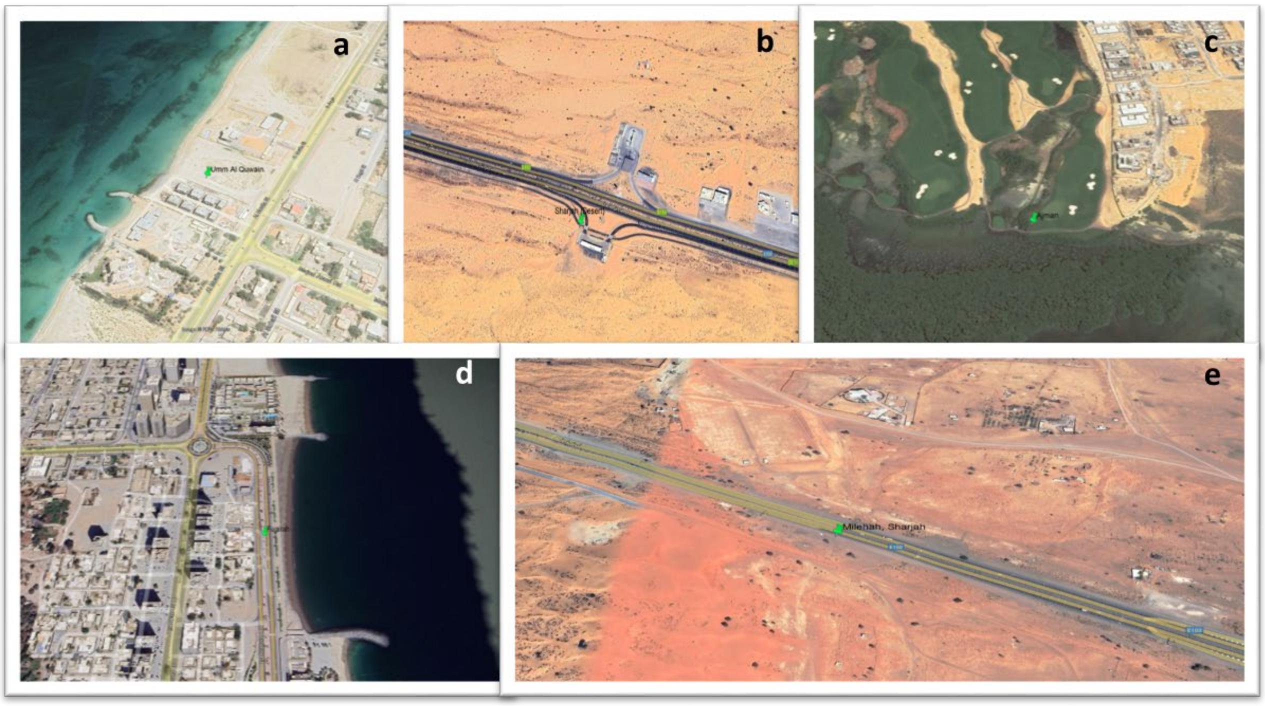


Figure 4 – Detailed map of all five sampling locations within the UAE, illustrating the positions of coastal and desert sites relative to major urban centres and prevailing wind direction.



Location	Sample ID	Latitude and Longitude	Habitat	Additional info
Umm Al Quwain	S1.C.R01	25°34'08.2"N 55°32'57.9"E	Beach	Inhabited area near residential units
Ajman	S2.C.R01	25°25'41.3"N 55°29'45.5"E	Beach	Remote area near Al Zorah natural reserve: tidal creek & mangrove forest with multiple bird species.
Sharjah	S3.D.R01	25°10'56.3"N 55°43'08.3"E	Desert	Uninhabited area near car service station
Fujairah	S4.C.R01	25°07'57.4"N 56°21'22.2"E	Beach	Inhabited area near residential units
Mileh ah, Sharjah	S5.D.R01	25°09'18.2"N 55°52'11.4"E	Desert	Uninhabited area. A UNESCO World Heritage archaeological site.

Figure 4b – Satellite views of sampling locations: (a) Umm Al Quwain; (b) Sharjah desert; (c) Ajman; (d) Fujairah; (e) Mleiha, Sharjah.



In contrast, the east coast faces the Indian Ocean, where monsoonal winds alternate seasonally. During winter, northeasterly monsoon winds blow from land to sea (Chaichitehrani et al. 2018), carrying locally generated dust and fibres offshore. In summer, the monsoon reverses, blowing onshore from the south-west, but the UAE's east coast is a narrow strip backed by the Hajar Mountains, which intercept much of this flow and prevent significant penetration of marine-derived material inland. The desert interior is therefore more strongly influenced by Arabian Gulf-side Shamal winds than by the Indian Ocean monsoon, consistent with the lower MPF concentrations observed at Fujairah (180 ± 20 fibers kg^{-1} dry weight), which is situated on the east coast and shielded by the mountain range.

Sedimentological characteristics reinforce this interpretation. Coastal sediments along the Arabian Gulf are predominantly carbonate-rich bioclastic sands derived from corals, mollusks and foraminifera, whereas desert dunes in the interior are composed largely of fine quartz sands with iron oxide hues. These two depositional environments are geologically distinct, and natural sediment reworking alone cannot account for the widespread occurrence of microfibers across both. The enrichment of fibres in desert samples relative to coastal sediments therefore supports the hypothesis of wind-borne contamination from coastal urban and industrial sources, overriding the slower timescales of sediment mixing and producing a more uniform contamination signal across contrasting environments.

The dimensional properties of microfibers are critical for understanding their Aeolian transport potential. Fibres with diameters below 20 µm and aspect ratios above 20 are particularly prone to remaining airborne for extended periods, behaving similarly to coarse particulate matter (PM10) (Evangelopoulos et al. 2024). Previous studies have documented predominantly shorter fibres (< 500 µm) with diameters of 10–30 µm in remote mountain and desert regions (Allen et al. 2019; Brahney et al. 2020), consistent with long-range transport. The relative homogeneity of microfiber concentrations between coastal and desert sites in the present study suggests that transport is dominated by thin, elongated fibres capable of suspension and downwind redistribution under prevailing Shamal winds.

The highest microfiber loads were recorded at Mleiha (570 ± 35 fibers/kg), an uninhabited desert site located downwind of Dubai and Sharjah and situated before a mountain ridge, suggesting that the mountain acts as a topographic barrier that enhances local deposition. Mleiha concentrations were comparable to those at Sharjah (550 ± 20 fibers/kg), a heavily urbanised coastal area, indicating the homogenisation of deposition in the downwind direction. The lowest values were at Fujairah (180 ± 20 fibers/kg), located on the Gulf of Oman coast beyond the mountain range, while Ajman (220 ± 21 fibers/kg) and Umm Al Quwain (430 ± 15 fibers/kg) displayed intermediate concentrations reflecting their respective levels of urbanisation and exposure to transported fibres.

In a broader perspective, the predominance of microfibers in desert sediments may provide a partial explanation for the "missing size classes" described in the global ocean by Cózar et al. (2014), where a fractal fragmentation model predicts a continuous size distribution but smaller fractions are underrepresented in surface waters. The essentially one-dimensional geometry of fibres may facilitate their preferential aerosolisation and subsequent deposition in downwind terrestrial environments such as arid deserts, potentially accounting for a fraction of the expected but unrecovered material in oceanic surveys. Taken together, these findings underscore that microfiber contamination in the UAE is not limited to urbanised coastal zones but extends across distinct sedimentary environments, including remote deserts, challenging the assumption that deserts act as low-contamination reference sites.

Conclusion -

The present study provides a preliminary baseline for understanding microfiber distribution across both desert and coastal environments in the UAE. Microfibers — predominantly cellulosic/rayon — were detected at all five sampling sites, with consistently higher concentrations in desert sediments (550–570 fibers/kg dry weight) than on coastal beaches (180–430 fibers/kg dry weight), despite the absence of statistically significant differences between the two groups. The spatial distribution and dimensional characteristics of the detected fibres are consistent with long-range Aeolian transport driven by the prevailing northwesterly Shamal winds from coastal urban areas toward the desert interior.

Sedimentological contrasts between carbonate-rich coastal sands and quartz-dominated desert dunes further support the interpretation that microfibers in desert environments are primarily of atmospheric origin rather than the product of local sediment reworking.
These findings highlight the need for more extensive future investigations, including atmospheric monitoring and source-tracing studies, to better characterise the ecological risks associated with microplastic fibre contamination in arid environments. Future sampling programmes should encompass a wider range of sites — including more remote desert locations, additional east coast stations and sites with contrasting levels of anthropogenic influence — in order to more robustly disentangle the relative contributions of local deposition and long-range atmospheric transport.

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Appendix -

Table 1 a - Abundance and distribution characteristics of microplastics in UAE and Arabian Gulf.

Article	First Author – Year of publication	DOI	Concentration	TYPE OF MICROPLASTIC – Polymer type Fiber / fragment	Environmental Matrix – Beach sediment, Subsurface water, waster column, seafloor sediment, Biota(fish, sponge)	Location	Instrument used - Pyrolysis-GC-MS | Pyrolysis Gas Chromatography, FTIR, Raman spectrometry
Microplastics in coastal environments of the Arabian Gulf	Abayomi, O. A 2017	https://doi.org/10.1016/j.marpolbul.2017.07.011	Sea surface abundance - varied between 4.38×10^{4} and 1.46×10^{6} particles·km^{-2} concentration of microplastics in intertidal sediments varied between 36 and 228 particles m^{-2}	mainly low-density polyethylene and polypropylene. Blue fibers, ranging between 1 and 5 mm	Sea water, Sediment, beach	Arabian Gulf	(FT-IR/FT-NIR) spectrometer, in conjunction with a database as reference and OMNIC FT-IR software

Microplastic Contaminants in the Sediment of the East Coast of Saudi Arabia	Hamza Jawad Al-Shaikh Ali 2022	DOI: 10.5772/intechopen.109019	Microplastic abundance ranged from 5.5 ± 1.55 to 21.2 ± 0.68 particle/kg (51.1 ± 14.71 to 152.8 ± 21.32 particle/m^2) in low tide region, and from 6.3 ± 4.05 to 16.5 ± 4.98 particle/kg (50.6 ± 31.21 to 204.5 ± 64.15 particle/m^2) in high tide region	average particle size of 1.55 ± 0.94 mm Dominant colours transparent (34%) and blue (30%) Dominant shape fibers(96%) Polyethylene terephthalates were the common polymer type of fibers, while polyethylene and high-density polyethylene were common in fragments and filaments	Sediment collected from beach	Arabian Gulf	Leica CME 1000× compound microscope. Bruker ATR-FTIR. spectrums were matched with referenced polymer spectra using the library in OPUS-spectroscopy software.

Evaluation of microplastics in beach sediments along the coast of Dubai, UAE	Aslam H 2020	doi.org/10.1016/j.marpolbul.2019.110739	number of microplastic is 59.71 items per kg of dry sediment (or 165 items·m^{-2})	Microplastics Blue and fibrous microplastics were dominant. Polyethylene strings and fibers were abundantly found	Beach sediment	Dubai, UAE	(FT-IR) analysis
MAN-MADE LITTER ON THE SHORES OF THE UNITED ARAB EMIRATES ON THE ARABIAN GULF AND THE GULF OF OMAN	Khordagui H K 1994	doi.org/10.1007/BF00482711	Not reported	Litter, manmade debris Shape, colour, polymer type not reported	Beach	UAE	NA
Mapping of heavy metal contamination associated with microplastics marine debris - A case study: Dubai, UAE	Attaelmanan AG 2023	doi.org/10.1016/j.scitotenv.2023.164370	Not reported. Analysis focuses on detection of heavy metal percentages	Microplastics Not reported. Analysis focuses on detection of heavy metal percentages	Beach sediment	Dubai UAE	X-ray fluorescence spectroscopy

Microplastic in Commercial Fish in the Mediterranean Sea, the Red Sea and the Arabian/Persian Gulf. Part 3. The Arabian/Persian Gulf	Habib R Z 2022	10.4236/jwarp.2022.146025	Qatari beach - observed MPs counts between 36 and 228 $MP{\cdot}m^{-2}$ or between 6 and 38 $MP{\cdot}kg^{-1}$ of dry sediment In UAE - average of 59.7 MP/kg of dry sediment equaling 165 MP/m^2 with a range of 337 ± 180 MP/m^2 (133.98 ± 67.48 MP/kg sediment) to 26 ± 5 MP/m^2 (8.43 ± 1.54 MP/kg sediment).	Microplastics	Review – beach sediment water	Arabian Gulf	NA review

Microplastic pollution in oyster bed ecosystems: An assessment of the northern shores of the United Arab Emirates	Hammadi M A 2022	doi.org/10.1016/j.envadv.2022.100214	mean abundance in the sediment samples was 191.7 ± 95.5 MP/Kg of dry weight	Microplastics Fibers(93%) and fragments(7%) were the most common shapes Polymers found – Low density polyethylene Lignin Polypropylene Polyethylene	MPs pollution in oyster bed, sediment	UAE	Microplastic analysis by attenuated total reflectance Fourier transform infrared spectroscopy (ATR-FTIR)

Microplastics in fishes of commercial and ecological importance from the Western Arabian Gulf	Baalkhuyur F M 2020	doi.org/10.1016/j.marpolbul.2020.110920	MPs reported inside fish species – Figures not comparable	Microplastics – On average, 5.71% of the fish dissected contained MPs, ranging from 5 to 15% of individual fish examined containing MPs among species (Siganus canaliculatus and Rastrelliger kanagurta, respectively). Ingested plastic consisted primarily of fishing threads (1.04 ± 0.06 mm), followed by	MPs found in gastrointestinal tract of fish	Western Arabian Gulf	FT-IR analysis

				fragments (1.16 ± 0.11 mm). Polyethylene (PE) and polypropylene (PP) were identified as the most abundant polymers ingested by the fishes.				

Risk Assessment Approach to Identify Possible Risks to the Marine Environment in the United Arab Emirates	Elsergany M 2023	doi.org/10.13189/eer.2023.110403	Not reported	Marine Pollution	Review	UAE	NA review
The occurrence of microplastic in marine ecosystems in the Middle East: A review	Alarif W M 2023	doi.org/10.1016/j.rsma.2023.103208	Arabian Gulf (up to 4.6×10^4 particles/km^2) The dominating MPs in the sediment on the shore of Dubai, UAE, are 0.33 mg/g or 953 mg/m^2 MPs with blue color and a fibrous form. Polyethylene and fiber (63.87%) are the most abundant MPs at the site	Microplastics	Review	Arabian Gulf and others	NA review
Trends of microplastic abundance in personal care products in the United Arab Emirates over the period of 3 years (2018–2020)	Habib R Z 2022	doi.org/10.1007/s11356-022-21773-y	Not reported	Microplastics Not reported	MPs microbead found in cosmetic product	UAE	FT-IR analysis

Marine litter and microplastic pollution in mangrove sediments in the Sea of Oman	Al Tarshi M 2024	doi.org/10.1016/j.marpolbul.2024.116132	microplastic levels in sediment ranging from 6 to 256 pieces/kg. Al-Sawadi's lagoon had the highest microplastic abundance (27.52 ± 5.32 pieces/kg), in contrast to Al Qurum's Marine Protected Area with the lowest (0.60 ± 1.12 pieces/kg).	Microplastic Primary plastic polymers identified were Polyethylene (PE) at 40 % and High-Density Polyethylene (HDPE) at 28 % Fibers are more abundant in the “mudflat” zone	Sediment	Arabian Gulf	The microplastic particles on the filter were visualized and examined using an OLYMPUS SZ61 stereomicroscope Identification of microplastic by attenuated total reflection-Fourier transform infrared spectroscopy (ATR-FTIR)
Linking effects of microplastics to ecological impacts in marine environments	N Khalid 2021	doi.org/10.1016/j.chemosphere.2020.128541	General article and reported figures are not from Arabian Gulf region.	Different Microplastic – e.g. PE PS PP PA PET LDP		NA Review	NA Review

Measuring achievements towards SDG 14, life below water, in the United Arab Emirates	Gulseven O 2020	doi.org/10.1016/j. marpol.2020.103972	Not reported	SDG 14 , water pollution	NA	NA	

Table 1 b - Shape and characteristics of MPF microplastic fibers with respect to wind transport

Article	**First Author – Year of publication**	**DOI**	**Concentration**	**dimension , Size range/ µm**	**Location**
Airborne microplastics in a suburban location in the desert southwest: Occurrence and identification challenges	Chandrakanthan et al 2023	https://doi.org/10.1016/j.atmosenv.2023.119617v	0.02 and 1.1 microplastics/m^3 (average concentration of 0.2 microplastics/m^3)	size range of (5–5000) µm.	Arizona

Long-range atmospheric transport of microplastics across the southern hemisphere	Chen et al 2023	https://doi.org/10.1038/s41467-023-43695-0		For compact MPs fragments with effective diameter, of ~120–160 µm and a density of ~1.4 g cm^{-3}	Southern Oceans
Consistent transport of terrestrial microplastics to the ocean through atmosphere. Environ.	Liu et al 2019	https://doi.org/10.1021/acs.est.9b03427	0–1.37m^3	16–2087	West Pacific Ocean
Large accumulation of micro-sized synthetic polymer particles in the sea surface microlayer	Song et al 2014	https://doi.org/10.1021/es501757s	55.93–174.97m^3	<500–5000	Surabaya, Indonesia

Characteristic of microplastics in the atmospheric fallout from Dongguan city, China: preliminary research and first evidence.	Cai et al 2017	DOI 10.1007/s11356-017-0116-x	175 to 313 particles/m^2/day in the atmospheric fallout. concentrations of microplastics ranged from 31 ± 8 to 43 ± 4 particles/m^2/day	majority of fibres in Dongguan to be 200–700 µm in length with fibres of ≥4,200 µm (longest fibre),	China
Perspectives on transport pathways of microplastics across the Middle East and North Africa (MENA) region.	Pu et al 2024	https://doi.org/10.1038/s41545-024-00410-w	In UAE - 191.7 ± 95.5 items/kg	In the UAE region – (1000-2000),	NA
Atmospheric transport and deposition of microplastics in a remote mountain catchment	Allen et al 2019	https://doi.org/10.1038/s41561-019-0335-5	Average daily MPs deposition of 365 m^{-2} d^{-1} (±69, particles ≥5 µm)	predominant fibre lengths of 100– 200 µm and 200–300 µm. fibres up to ~750 µm long	French Pyrenees

Long-distance atmospheric transport of microplastic fibres influenced by their shapes	Xiao et al 2023	doi.org/10.1038/s41561-023-01264-6	NA	most MPFs have a width within 5–30 µm (47 µm and 1,140 kg $m^{-3)}$	NA
Synthetic fibers in atmospheric fallout: A source of microplastics in the environment?	Dris R 2016	dx.doi.org/10.1016/j.marpolbul.2016.01.0	0.3-1.5m^3	50-1650	Paris

Atmospheric microplastic deposition in an urban environment and an evaluation of transport	Wright S.L 2019	doi.org/10.1016/j.envint.2019.105411	Atmospheric deposition in London. deposition rates ranging from 575 to 1008 microplastics/m^2/d. Among MPs Microfibers presence significantly high (92%)	The modal diameter of the observed fibres was 20–25 µm	London U.K

Airborne microplastics in indoor and outdoor environments of a coastal city in Eastern China.	Liao et al 2021	https://doi.org/10.1016/j.jhazmat.2021.126007	188.7 ± 84.8	5–1794	Wenzhou, Eastern China
Airborne fiber particles: types, size and concentration observed in Beijing	Li et al 2020	https://doi.org/10.1016/j.scitotenv.2019.135967	4500–7200	5-200	Beijing

New insights into the aging behavior of microplastics accelerated by advanced oxidation processes	Liu et al 2019	https://doi.org/10.1021/acs.est.9b00493	1.42 ± 1.42	23-9555	Shanghai
Microplastic abundance in atmospheric deposition within the Metropolitan area of Hamburg, Germany	Klein and Fischer 2019	https://doi.org/10.1016/j.scitotenv.2019.05.405	median abundance between 136.5 and 512.0 microplastic particles per m2/day	Regarding fibers the size classes were 5000 and 300 µm (was dominant 90 fibers), 34 fibers belonged to the size between 300–63 µm and 9 were identified in the smallest class (<63 µm).	Hamburg

Chapter 6

Conclusions -

With growing concerns emerging from anthropogenic microplastic contamination of our ecosystems arises the pressing need for a broad new initiative to formulate regulatory actions and strategic plans to reduce and mitigate the environmental hazards of microplastic pollution. Despite this broad narrative gaining traction, significant knowledge gap persists regarding distribution and behavior of microplastics and particularly microfibers leaving our understanding alarmingly insufficient. Impact of microplastic contamination on blue carbon ecosystems like mangroves, which are carbon sinks, deserves meticulous research, because our understanding of microplastic as reservoirs of contaminants and their pathways is limited. Coupled with that we need to take notice of the surging hazard of ubiquitous contamination of microfibers in our environment and the largely undocumented area of microfiber transport pathways. It is well established that microfiber contamination stems from byproducts of textile industry and presence of this menace is practically ranging from inhabited urban centers to the remotest locations. Increasingly studies confirm that these tiny fibers exhibit long range air transport aided by long residence time in the atmosphere, also we see evidence of remote sinks of microfiber which are shaped by local and regional geographic factors. This is a harbinger of an alarming situation from ecological, environmental and human health perspectives. Such kind of contamination remains largely invisible to the naked eye, and it can be coupled with a Trojan Horse effect (harmful consequences) as they can adsorb, concentrate and transport toxic chemicals and pathogens to widespread and diverse locations from source of origin. Multiple studies have confirmed with laboratory studies about the health hazards posed by microplastics and nano plastics entering the environment. One key thing to note is that these laboratory tests often produce results by administering commercially acquired particles that represent a singular type of polymer, typically characterized by uniform size and shape; however, in reality microplastics encountered in natural environments consist of a complex mixture of polymers that vary considerably in size, shape, and chemical composition. Furthermore, in nature such contaminants undergo a range of weathering processes that further alter their physical and chemical characteristics. Therefore, such complex variability in nature complicates the extrapolation of laboratory findings to real-world scenarios because the response elicited by different organisms to those heterogenous polymers might differ from laboratory results based on a single polymer.

This thesis makes a valuable contribution to this critical effort by investigating the mangrove sinks of Maldives along with urban and remote desert locations within UAE. We had the opportunity of collaborating with MaRHE center and the University of Dubai University in this regard. Best to our knowledge this is the first of a kind investigation exploring microfiber contamination in UAE desert and plastic contamination within the blue carbon mangrove ecosystems of Maldives.

Our investigation in the UAE unearthed a peculiar phenomenon of microfiber sink in an uninhabited desert region called Mleiha. Over there, there are several factors leading to unusually high concentration of microfibers. Wind driven long range transport of microfibers plays a key role by bringing in contaminants from far away regions. The weather pattern in UAE experiences strong inflow of north westerly winds from distant areas which can bring in micro fibers from far away. In UAE following the cold front in winter strong northwesterly winds commonly develop, known as the winter Shamal (Arabic for “north”). Winter Shamal events occur relatively infrequently, typically two to three times between November and March, but they can be intense, with wind speeds often exceeding 50 km/hour. These northwesterly winds may persist for two to five days, generating dust storms and causing sharp decreases in temperature. In spring cold weather can still trigger northwesterly Shamal events in early spring, while warming land surfaces enhance sea-breeze circulation. By May, rising temperatures and the development of a seasonal low-pressure system over the Rub’ al Khali (Empty Quarter) signal the onset of the summer circulation regime. During this time, prevailing wind directions gradually shift toward the northwesterly flow that dominates summer. Despite this transition, spring winds remain variable, with occasional dust storms and blowing sand, particularly in exposed desert areas. Summer conditions in the UAE are defined by the persistent northwesterly Shamal wind. Although the large-scale prevailing wind over the Arabian Gulf generally flows from northwest to southeast throughout the year, this pattern becomes especially dominant during summer. The summer Shamal originates over the Tigris–Euphrates basin in Iraq and moves southeastward across the Gulf. Its sustained nature has led to the nickname “120-day wind,” reflecting its near-continuous presence from late May through August. One major outcome of the UAE’s prevailing wind patterns is the long-range transport of sand and dust into and out of the country. Due to the arid nature of the region, winds can readily entrain fine sand and soil particles and transport them over hundreds or even thousands of kilometers. Research on dust storms has identified several key source regions that contribute airborne dust to the UAE. Approximately 60% of the dust present in the UAE’s atmosphere originates from Iraq and surrounding areas. The Tigris–Euphrates basin and the Syrian Desert are frequent sources of large dust storms, particularly under Shamal wind conditions. When strong northwesterly winds develop, dust from Iraq, Syria, and Kuwait is transported southeastward along the Arabian Gulf. These dust-laden winds often reach the UAE, especially impacting Abu Dhabi and Dubai after passing over the Saudi Arabian Gulf coast. For instance, spring dust storms originating in Iraq commonly travel southward with Shamal winds and can blanket the UAE in haze within one to two days. Satellite imagery and visibility records clearly show that northwesterly winds act as a transport corridor, moving dust from Iraq across the Gulf toward the UAE and onward to Oman or the Arabian Sea. Once airborne dust reaches the UAE, local geographic features influence where particles ultimately settle. Mountain ranges, such as the Hajar Mountains along the eastern boundary of the country, can block or channel winds, leading to preferential accumulation of dust in certain locations. Mleiha is also located near the Hajar mountain ranges and the incoming north westerly Shamal winds find a natural barrier there, making it a local hotspot for microfiber deposition. Broad desert basins and dune fields also act as sinks where wind-transported sand and dust can settle. In summary, dust and sand storms affecting the UAE are best understood as a regional phenomenon, driven by strong winds that transport material from Iraq, Iran, Saudi Arabia, and beyond into the Emirates, while some locally generated dust may also be carried onward to neighboring countries. This regional interconnectedness of dust transport is widely recognized as a shared environmental challenge. In order to explain the excess concentration of micro fibers in a remote desert like Mleiha, our hypothesis is centered on inflow of micro fiber emerging from long range atmospheric transport and dispersion and subsequent accumulation occurring due to Hajar Mountains acting as a barrier; which creates a hotspot for micro fiber sink. Our analysis is congruent with established knowledge of UAE wind dynamics and pollutant dispersion.

In the Maldives the blue carbon mangrove ecosystems are a dynamic sink of carbon, with varying carbon uptake rate from photosynthesis; depending on tides, nutrient availability, salinity, temperature, tree age and species. Mangrove ecosystems of Maldives are already feeling the pressure of climate change and a warming of ocean, in addition to it plastic contamination becomes a serious threat to the ecosystem and the biota. The issue of unmanaged waste disposal and burning waste in landfills aggravates plastic contamination which can leave a lasting impact from ecological perspective. In the face of such threats, this work contributes significantly to the expanding body of knowledge essential for informed policy making and can develop effective management strategies aimed at mitigating microplastic pollution hazards.

In conclusion, this thesis addresses the pressing need for further rigorous scientific studies in carbon sink hotspots like Maldive and microfiber sink regions located in the remote deserts of UAE. To the best of our knowledge these remote geographical regions remain largely unexplored, making our investigation first of a kind, and the environmental hazards stemming from micro fiber contaminants warrant immediate attention.

Appendix

International Conferences and publications

1. Mazumdar A., Losi N., Ferrero L. (2023) Sea spray hygroscopicity in synergy with atmospheric aerosol. In *Book of Abstracts: ICYMARE 2023 Oldenburg*. ICYMARE Organizing Committee. September 18 – 22, Oldenburg, Germany

2. Mazumdar A., Saliu F., Galli P. (2025) Microplastics as emerging contaminants in mangrove ecosystems of Maldives. . In Book of Abstracts: *ICYMARE 2025 Bremerhaven. ICYMARE Organizing Committee*. September 8 – 12, Bremerhaven, Germany

3. Cerri, F., Louis, Y. D., Fallati, L., Siena, F., Mazumdar, A., Nicolai, R., ... & Galli, P. (2024). Mangroves of the Maldives: A review of their distribution, diversity, ecological importance and biodiversity of associated flora and fauna. Aquatic Sciences, 86(2), 44.

4. Mazumdar A., Riseri D., Zitouni M.S., Saliu F., Galli P., Microfibers pollution in the UAE: from the Arabian Gulf to the desert? (Manuscript stage)

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