

Spatio-temporal density patterns of corallivorous invertebrates around Rodrigues Island, Western Indian Ocean

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Abstract. Priest JS, Ramah S, Ramkalam M, Korimbocus AM, Bhagooli R, Kaullysing D. 2025. Spatio-temporal density patterns of corallivorous invertebrates around Rodrigues Island, Western Indian Ocean. *Indo Pac J Ocean Life* 9: 92-99. Coral reef ecosystems globally are subject to multiple stressors making them targets for corallivore predation by organisms such as the crown-of-thorns starfish (COTS), *Acanthaster* (Gervais 1841) and gastropods, mainly *Drupella* (Thiele 1925). Around Rodrigues Island, little is known about COTS and gastropod corallivory. Surveys to assess COTS and *Drupella cornus* (Röding 1798) density were conducted using the belt transect method and random quadrat sampling, respectively, around Rodrigues Island from 2017 to 2020 at least once at six lagoon sites (1-2 m). In April 2024, additional surveys were carried out at nine lagoon sites (1-2 m) and seven dive sites (9-28 m), including previously surveyed locations. During this study, no COTS were observed along the surveyed transects, however, suspected COTS feeding scars were spotted on some coral colonies at two lagoonal sites. Significant spatio-temporal variations in *D. cornus* density were observed, with considerable increases at Rivière Banane from 0.07 ± 0.07 ind/m² in 2018 to 0.93 ± 0.48 ind/m² in 2024 and Var Brûlée from 0.40 ± 0.16 ind/m² in 2017 to 2.07 ± 0.89 ind/m² in 2024. Two-way ANOVA test revealed significant interaction between site and year ($p < 0.01$), suggesting that both spatial (site) and temporal (year) factors interact in influencing *D. cornus* densities. These findings suggest potential reef stress, possibly caused by fishing pressure, predator loss, and/or rising sea temperatures, and highlight the necessity for corallivore monitoring and intervention to protect the health and resilience of the coral reefs of Rodrigues. The study revealed that *D. cornus*, rather than COTS, poses a potential threat to Rodrigues reefs, particularly at Var Brûlée and Rivière Banane. Further research is warranted to gather additional data on corallivore distribution and their impact on the coral reefs of Rodrigues.

Keywords: *Acanthaster*, corallivory, crown-of-thorns starfish, *Drupella cornus*, Rodrigues

Abbreviations: COTS: Crown-of-thorns starfish, WIO: Western Indian Ocean

INTRODUCTION

Corals are diverse and important organisms, particularly scleractinian corals as they are reef- or habitat-building corals essential for coral ecosystem productivity, promoting nutrient recycling, primary production, and reef development (Pratchett et al. 2008). However, these vital ecosystems are under multiple anthropogenic and natural, regional and local-scale threats such as deleterious fishing practices and overfishing, coral diseases, climate change, and poor water quality stemming from land-based pollution, all of which are exerting extreme pressure on these fragile ecosystems and influencing their functioning, making them susceptible to corallivore predation (Souter et al. 2021). Corallivores such as the crown-of-thorns starfish (*Acanthaster* Gervais, 1841 - COTS) and gastropods such as *Drupella* (Thiele, 1925) and *Coralliophila* (H. Adams & A. Adams, 1853) are contributors to global coral reef degradation by preying on stressed corals (Tsang and Ang 2015; Kaullysing et al. 2016; Zhang et al. 2024). The indicative sign of

corallivores feeding on corals in an area is the white scar left behind on the coral, symptomatic of the loss of living coral tissue (Kaullysing et al. 2017; Gautrand et al. 2023). Corallivore outbreaks of species such as *Acanthaster* spp. and *Drupella* spp. have been reported on coral reefs globally.

COTS predate primarily on scleractinian corals such as *Acropora* spp. (Oken 1815) but can diversify their diets to soft corals and molluscs when coral prey is limited (Pratchett et al. 2017). COTS can cause massive depletion of living hard coral cover of up to 90% as they can consume up to 150 to 250 cm² of live coral tissue daily (Souter et al. 2021). COTS outbreaks have been reported globally throughout the Indo-Pacific Ocean. The Island of Moorea had more than 96% of its corals killed due to high COTS densities from 2005 to 2010 (Kayal et al. 2012), and COTS have contributed to 50% reduction in live hard coral from 1985 until 2012 in the Great Barrier Reef (Westcott et al. 2020). Specifically, in the Southern Indian Ocean, COTS outbreaks were recorded in South Africa's Two-mile Reef during the 1990s, lasting almost 10 years and peaking between

1994 and 1996 (Celliers and Schleyer 2006). Outbreaks were also reported in the Seychelles in 1997 and 2014, and in Mauritius at certain sites in 2014 (Obura et al. 2017; Uthicke et al. 2023). Although published scientific data on COTS distribution around the Republic of Mauritius is currently scarce, Ramah et al. (2021) reported observing a single *Acanthaster* sp. individual at Saya de Malha Bank. There are five known COTS species, and they are relatively geographically defined throughout the Indian and Pacific Oceans. The COTS commonly found in the Southwest Indian Ocean is *Acanthaster mauritiensis* de Loriol, 1885 species (Uthicke et al. 2023).

The less conspicuous corallivorous *Drupella* spp., distributed in the shallow waters of the tropical and subtropical Indo-Pacific region, feed predominantly on branching *Acropora* spp., which comprises up to 80% of their diet (Raymundo et al. 2016; Bessey et al. 2018). Outbreak densities of *Drupella* spp. range from 1.4 to 6.4 ind/m² (Cumming 2009). *Drupella* spp. outbreaks have been recorded throughout the Pacific and Indian Oceans, whereby four massive population outbreaks of *Drupella* in the South Pacific Ocean were recorded in 1983, 1989, 1993, and 2014 to 2016 (Zhang et al. 2024). While several scientific studies on corallivory by gastropods (Kaullysing et al. 2016, 2017, 2019, 2020) and fish (Tiddy et al. 2021, 2023; Ricot et al. 2023) are available for Mauritius, the presence and impact of corallivores on the coral reefs of Rodrigues Island is largely unknown as the published information on corallivores in Rodrigues is limited. Rodrigues is considered a biological hotspot, and despite the ecological importance of the reefs of Rodrigues, very few studies on coral predation have been conducted, presenting a crucial data gap concerning corallivory in Rodrigues. Thus, this study aimed at observing the presence of corallivores and assessing the density of corallivorous invertebrates (*Acanthaster* spp. and *Drupella cornus*) on coral reefs around Rodrigues Island to understand corallivore dynamics and to provide essential baseline data needed for reef conservation efforts.

MATERIALS AND METHODS

Study area

Rodrigues, a remote island in the Western Indian Ocean, is situated about 560 km east of Mauritius (Figure 1). Rodrigues is located at the eastern end of an east-west trending ridge that intersects the Mascarene Plateau, and it does not share the same geological origin as the other Mascarene Islands. As an autonomous outer island of the Republic of Mauritius, Rodrigues spans roughly 108 km² with a hilly terrain, beaches and coral reefs. It has a vast lagoon area covering 300 km² bordered by a 200 km² fringing reef. The island's relatively uncharted marine waters appear to host a rich biodiversity (Bhagooli and Kaullysing 2019).

Field surveys

Surveys were carried out by snorkelling around Rodrigues Island at lagoon sites (1-2 m), namely, Rivière Banane, Pointe Coton, St François, Passe Semone, Var Brûlée, and Couzoupa_S (shallow waters) at least once in April from 2017 to 2020 to assess crown-of-thorns starfish (COTS) and corallivorous gastropods density. Additionally, in April 2024, snorkelling or SCUBA diving surveys were conducted once at each of 16 lagoonal (shallow) and dive sites around Rodrigues (Figure 1) to assess the distribution and density of the corallivores, including the sites surveyed at least once during the period 2017-2020. The lagoon sites (1-2 m) surveyed in 2024 were Rivière Banane, Pointe Coton, Anse Ali, St François, Passe Semone, Var Brûlée, Hermitage Island, Paté Reynieux, and Couzoupa_S, and the dive sites (9-28 m) were Antonio's Finger, Aquarium, Plateau Bénitier, Ti Trou, Couzoupa_D, Tombant Coco, and Ti Pate. It is noteworthy that restricted temporal coverage and potential under-detection of cryptic or nocturnal individuals may lead to underestimation of true population densities and spatial coverage. Site selection was largely determined by accessibility and historical coral cover.

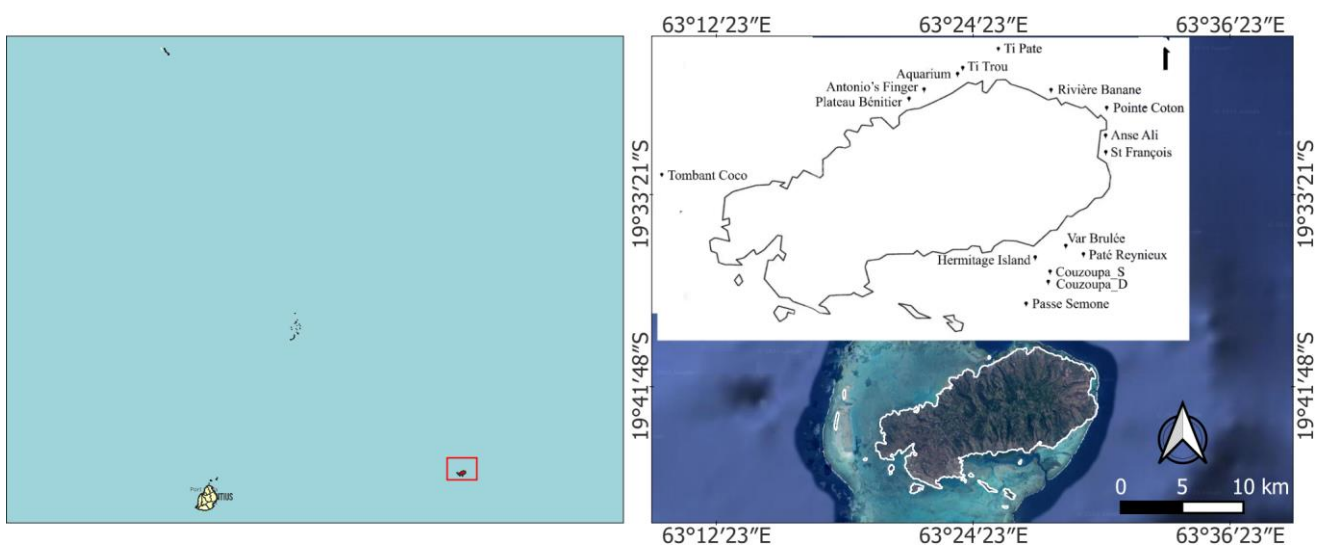


Figure 1. Study sites around Rodrigues Island, Mauritius

At each survey site, the belt transect method was used to assess COTS density (Samanthi 2021). Three belt transects each of 30×10 m (300 m^2), at 10 m intervals, were randomly laid to conduct a thorough search for COTS by direct observation, totalling a survey area of 900 m^2 at each site. The same belt transects were then used to conduct the line-intercept transect method to determine hard coral cover at each site. For corallivorous gastropod surveys, random quadrat method was used wherein 15 quadrats of 1 m^2 each were laid on coral colonies. *Drupella cornus* individuals were identified *in-situ* based on their shell morphology - distinct elongated conical shells with prominent axial ribs (Kaullysing et al. 2016), following morphological descriptions provided in standard identification guides for Indo-Pacific gastropods.

Invertebrate corallivore density measurements

The density of invertebrate corallivores was determined by counting the number of individuals present on the corals within each quadrat/belt transect and was expressed as the mean number of individuals/ m^2 (ind/ m^2). The species of coral upon which the invertebrates was also noted. The corallivores and the feeding scars were photographed using a digital camera (Olympus Tough TG-6 and TG-7).

Data analysis

Spatial maps were produced using QGIS software and all statistical analyses were performed in IBM SPSS Statistics 21. The Shapiro-Wilk test was used to assess the normality of the data. Arcsin(sqrt) transformation was applied to non-normal data before conducting further statistical analysis. A two-way ANOVA test ($\alpha = 0.05$) was performed to determine significant difference in the mean number of *D. cornus* across all sites and years of study. The two-way ANOVA test was performed on sites with repeated surveys, sites surveyed only once were excluded from the analysis.

RESULTS AND DISCUSSION

Crown-of-thorns starfish assessment

COTS were not observed during the surveys along the belt transects in any year at the study sites. The lack of sightings of COTS was uniform across all sites with varying depths, coral species and structural complexity, suggesting that such an observation was not restricted to a particular habitat type. Generally, outbreaks of COTS have been linked to significant coral degradation in various regions (Deaker and Byrne 2022; Foo et al. 2024). The lack of observed COTS in the present study along the surveyed transects suggests that local environmental conditions, effective natural predation, or recent management interventions might be inhibiting the proliferation of this coral predator. This tends to indicate a potentially stable status in terms of COTS predation for Rodrigues reef ecosystems. However, the lack of observation of COTS does not necessarily indicate their complete absence from the entire reefs around Rodrigues. It may also suggest that they possibly occur at very low densities. COTS in low densities are beneficial to coral reef ecosystems as they consume the faster-growing corals, such as *Acropora* spp., thus allowing the slower-growing coral species to proliferate, which improves the coral species diversity and richness of that coral reef and promotes greater biological diversity and ecological functioning (Foo et al. 2024).

Although no COTS were observed during the surveys, suspected feeding activity by COTS as large and distinct white patches was noted on some coral colonies at Rivière Banane (Figure 2.A) and Var Brûlée in April 2024 (Figure 2.B). COTS feeding scars were observed as white patches of recently dead coral, and were distinguished by their large, distinct, and striking white color (Figures 2.A and 2.B). While such scarring can be attributed to COTS feeding, it may also result from White Syndrome (Roff et al. 2011) or coral bleaching. In contrast, scars left by *Drupella* spp. tend to be less distinct (Figure 2.C), often exhibiting blurred edges due to the subsequent colonization of turf algae, except in the most recent feeding areas (Miller et al. 2018). The scars are generally more focused at the base of the coral.



Figure 2. Suspected COTS feeding scar on *Acropora* (Oken 1815) at A. Rivière Banane and B. Var Brûlée in 2024 in Rodrigues, C. *Drupella* (Thiele, 1925) feeding scars indicated by yellow arrows at St François in 2024

In Rodrigues, there are four no-take Marine Reserves located in the north of the island (Rivière Banane, Anse Aux Anglais, Grand Bassin, and Passe Demie), and a large Marine Protected Area, the South East Marine Protected Area (SEMPA), located in the Southeast region of the Island (Pasnin et al. 2016). The presence of these Marine Protected Areas (MPAs) could have contributed to the possible absence or low COTS densities as fisheries management has been seen to influence COTS densities. MPAs, including no-take reserves, can protect the predators of COTS, such as the giant *Triton* (Montfort 1810), triggerfish, and butterflyfish. A study by Kroon et al. (2021) indicated that no-take reserves affect COTS densities as these areas have high COTS predator densities which in turn regulates COTS densities and reduces their outbreak frequency. There could be a multitude of factors influencing the density of COTS around Rodrigues, such as the ‘predator removal hypothesis’ and the ‘terrestrial runoff hypothesis’ (Kroon et al. 2021), therefore, more research needs to be done to determine the factors regulating the densities of COTS around Rodrigues.

***Drupella cornus* (Röding, 1798) density**

Even though no COTS individuals were found at the surveyed sites, another less-noticeable coral predator, *D. cornus*, was observed impacting six shallow reef sites, namely, Rivière Banane, Pointe Coton, St François, Passe Semone, Var Brûlée, and Couzoupa_S. In the Indo-Pacific, the exceptionally high densities of these snails seem to have caused widespread and severe coral mortality, a level of impact previously attributed only to the crown-of-thorns starfish, *Acanthaster planci* (Linnaeus, 1758) (Turner 1994). Considerable spatio-temporal variations in the mean density of *D. cornus* were revealed (Figure 3). *Drupella cornus* has remarkable camouflage ability. Its shell, which is often

covered with red coralline algae, allows it to blend with the substrate, particularly under coral colonies, in between coral rubbles, and under algal-covered surfaces. This enables *D. cornus*, as well as other *Drupella* species, to escape predation during feeding activities.

In the present study, *D. cornus* was exclusively observed at shallow depths, with no individuals found in deeper reef zones at the dive sites. This pattern aligns with previous observations, that is, *Drupella* spp. show a strong habitat preference for sheltered reef slopes (Zhang et al. 2024). The tendency for aggregation at shallower depths (Dehnert et al. 2022) may be influenced by factors such as the availability of preferred coral hosts such as *Acropora* (Oken 1815), increased coral damage from wave action, or higher temperatures that may facilitate feeding and reproduction. The absence of *Drupella* at relatively greater depths could also reflect habitat specificity or predator pressures that vary with depth (Osuka et al. 2022).

One of the main observations from this study was the increase in *D. cornus* density at certain sites. At Rivière Banane, the density increased from 0.07 ± 0.07 ind/m² in 2018 to 0.93 ± 0.48 ind/m² in 2024, a more than 13-fold increase. Similarly, at Var Brûlée, *D. cornus* densities increased from 0.40 ± 0.16 ind/m² in 2017 to 2.07 ± 0.89 ind/m² in 2024 (> 5-fold increase), despite a temporary decrease in 2019 (Figure 3). This marked rise may reflect outbreak dynamics similar to those observed at Ningaloo Marine Park (Ayling and Ayling 1987), where *Drupella* feeding on *Acropora* spp. contributed to over 30% coral loss within two years. Notably, *D. cornus* was absent from surveyed quadrats at Passe Semone in 2018, but by 2024, densities reached 0.33 ± 0.27 ind/m², indicating that previously uninhabited or sparsely populated areas are now experiencing *D. cornus* population growth.

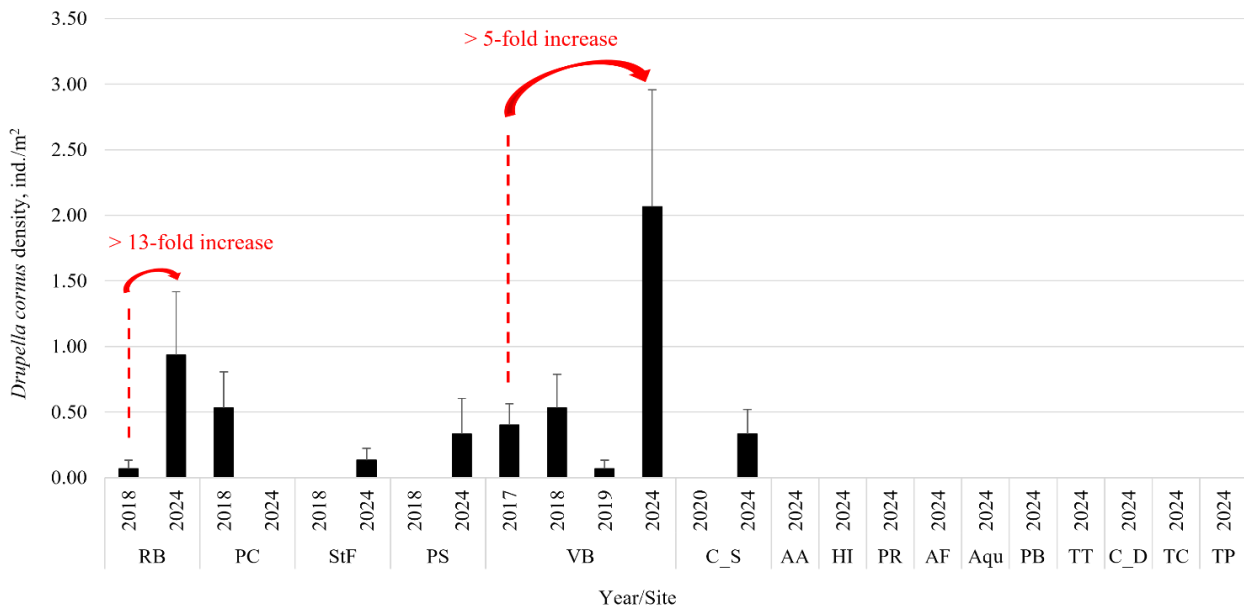


Figure 3. *Drupella cornus* (Röding 1798) density variations from 2017 to 2024 at study sites around Rodrigues. RB: Rivière Banane, PC: Pointe Coton, StF: St François, PS: Passe Semone, VB: Var Brûlée, C_S: Couzoupa_S (shallow waters), AA: Anse Ali, HI: Hermitage Island, PR: Paté Reynieux, AF: Antonio’s Finger, Aqu: Aquarium, PB: Plateau Bénitier, TT: Ti Trou, C_D: Couzoupa_D (dive site), TC: Tombant Coco, and TP: Ti Pate

Turner (1994) suggested that both anthropogenic factors, such as increased terrestrial runoff, overfishing, and reef damage, and natural factors like variable larval recruitment may drive *Drupella* outbreaks. These factors may also contribute to spatial and temporal variations in *Drupella* densities, as changing environmental conditions and reduced predator populations can influence recruitment success and survival. Similarly, Haslam et al. (2025) revealed high genetic connectivity among *D. cornus* populations along the Western Australian coast, indicating they are not genetically distinct. This suggests that larval dispersal plays a significant role in the spread of *Drupella* across regions, potentially leading to outbreaks in new areas. Such connectivity may facilitate rapid colonization of reefs following disturbances, underscoring the importance of understanding genetic flow when managing *Drupella* densities and mitigating their impact on coral reef ecosystems.

Drupella densities recorded align closely with what are considered typical background levels on the Great Barrier Reef, where densities generally remain low, often below 2 ind/m² (Cumming 2009). While these figures are relatively low compared to the severe outbreaks documented elsewhere, they are noteworthy given the potential for *Drupella* to contribute to coral stress, especially on already compromised reefs. In contrast, other regions have experienced far more devastating outbreaks. An estimated population of approximately 500 million individuals was reported in the Ningaloo Marine Park of Australia in 1983 (Ayling and Ayling 1987). A rapid rise in *D. cornus* numbers led to a 75% loss of live coral cover in the back-reef areas of the Ningaloo Reef in the 1980s and 1990s (Turner 1994). In the southwest Indian Ocean, Kaullysing et al. (2016, 2017) reported *Drupella* sp. outbreaks around the island of Mauritius from 2010 to 2014, and in 2016 an outbreak density of 158.14 ± 13.85 ind/m² was reported.

The ecological implications of these trends at these sites in Rodrigues could potentially be pointing to environmental stressors affecting coral reefs, such as climate change and coral bleaching or the decline of natural predators, which may have caused *D. cornus* populations to increase (Zhang et al. 2024). The increasing trend may have been driven by reduced corallivore predator pressure, predator removal and/or increased nutrient loading from coastal or agricultural runoff, promoting algal growth and supporting *Drupella* proliferation (Kaullysing et al. 2016; Zhang et al. 2024). As documented by Kaullysing et al. (2016), increased nutrient flow from riverine and coastal runoff promotes macroalgal growth when conditions are favorable leading to the smothering and stress of coral colonies. This makes the corals more susceptible to increased predation by *D. cornus* (Zhang et al. 2024). Excessive input of nutrients in the coastal region may also lead to coral bleaching, further stressing the corals and allowing *D. cornus* densities to increase.

In contrast, some reef sites experienced a decline in *D. cornus* density, such as Pointe Coton, where the population dropped from a relatively high density of 0.53 ± 0.27 ind/m² in 2018 to nil observations in 2024, possibly indicating localized environmental recovery, changes in

coral health, reduced stress by anthropogenic activities, or natural predation effects. During the surveys at Pointe Coton, live hard coral cover was estimated at 15% in 2018 and 5% in 2024 (personal observations). The abundance of corallivorous gastropods may be better predicted by the abundance of their predators rather than the availability of their coral prey due to the important trophic interactions predators have on controlling the population of gastropods (Shaver et al. 2020). Natural predators of *D. cornus*, such as durophagous fish species and other marine invertebrates, may have increased in abundance (Roff et al. 2019). Nonetheless, Saponari et al. (2021) observed that the distribution of *Drupella* spp. varied significantly at the reef scale, with the highest densities observed in areas with greater coral cover. The loss of coral cover led to the decrease in *Drupella* spp. population, lowering their impact. Lei et al. (2022) also observed higher densities of *Drupella* spp. in areas with higher coral cover. They also reported considerably lower densities of *Drupella* spp. in areas where their preferred diet was limited. Thus, decrease in live hard coral cover or habitat loss may be considered as one of the major drivers of *D. cornus* population decline. Coral bleaching events have increased in frequency and intensity due to rising sea surface temperatures (Reimer et al. 2024). This can lead to large-scale coral mortality (Hughes et al. 2018). As *D. cornus* preferentially preys on stressed corals (Kaullysing et al. 2016), their initial high density may have been linked to a preceding coral bleaching event at Pointe Coton that provided an ample food source of stressed corals. However, a subsequent decline in coral cover due to continued environmental stress may have resulted in the drop of *D. cornus* density (Lei et al. 2022).

The mean number of *D. cornus* differed significantly across sites ($p < 0.001$) and years ($p < 0.001$). The significant interaction between site and year ($p < 0.01$) suggested that both spatial (site) and temporal (year) factors interact in influencing *D. cornus* densities.

The increase in observed population density of *D. cornus* in 2024 around Rodrigues requires further in-depth ecological investigations to assess whether the predation level is indicative of broader reef degradation. The findings of this study point towards patchy distributions of *D. cornus* instead of large aggregations as reported elsewhere globally. It is noteworthy that the density of gastropods around Rodrigues is significantly lower than the densities observed around Mauritius in 2016 (158.14 ± 13.85 ind/m²) (Kaullysing et al. 2017). This lower density may be due to reduced anthropogenic influences and heavy tourism such as high diving or snorkeling activities on Rodrigues' coral reefs and the greater distance of these reefs from the shore, factors contributing to minimizing stress on the corals (Scott et al. 2017). Altogether, this study emphasizes on the importance of continued monitoring of *D. cornus* populations in Rodrigues, as their dynamics could serve as an indicator of reef health and ecosystem changes. The observed site-specific population increase of *D. cornus* emphasizes the need for strategic and adaptive management strategies at specific sites, for example at Var Brûlée.

Prey preference of *Drupella cornus*

Six shallow sites around Rodrigues Island were found to be impacted by corallivorous gastropod *D. cornus* during the study periods (Figures 4.A-4.F), namely, Rivière Banane, Pointe Coton, St François, Passe Semone, Var Brûlée, and Couzoupa_S. The preferred coral hosts varied with predation observed on numerous hard coral species across the affected sites. In general, *D. cornus* showed a

preference for tabular and corymbose *Acropora* spp. This may be largely attributed to the availability of the feeding area, ease of access to coral tissue from the base of the coral, as well as shelter (Kaullysing et al. 2016). *Acropora humilis* (Dana 1846) with branching and closely packed structures was also among the targeted species. These findings corroborate the widely documented association of *Drupella* snails with Acroporids.

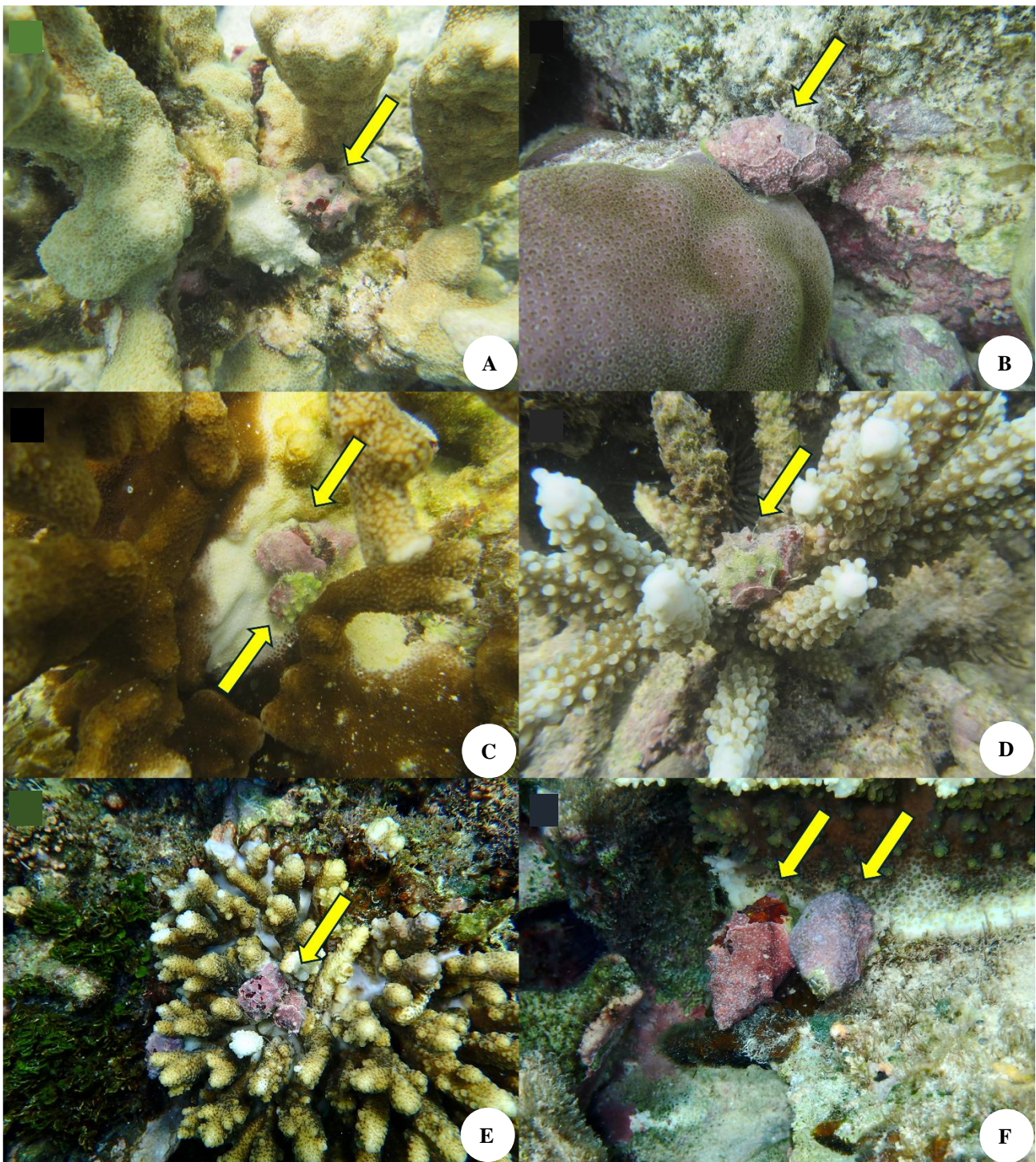


Figure 4. *Drupella* spp. individuals, indicated by yellow arrows, feeding on A. *Montipora* sp. at Var Brûlée in 2017, B. massive *Porites* (Link 1807) at Pointe Coton in 2018, C. *Montipora* (Blainville 1830) at Var Brûlée in 2018, D. *Acropora* sp. at Var Brûlée in 2018, E. corymbose *Acropora* sp. at Var Brûlée in 2024, and F. tabular *Acropora* sp. at Rivière Banane in 2024

Specifically, at Rivière Banane, *D. cornus* was mostly found on tabular *Acropora* as well as in crevices of dead corals, which indicate that these gastropods find shelter in microhabitats for protection from predators (Cumming 2009) or for laying their eggs (Koido et al. 2017; Sam et al. 2017; Kaullysing et al. 2020). At Pointe Coton, *D. cornus* was spotted on corymbose *Acropora* sp., *Pocillopora eydouxi* and on massive *Porites*. Occurrence of *D. cornus* on massive *Porites* Link, 1807 indicates that these gastropods may sometimes occupy less-preferred coral hosts. At St François and Passe Semone, *D. cornus* was seen on tabular *Acropora*, while at Var Brûlée, it was found on *Montipora* sp. and tabular, branching and corymbose *Acropora*. *Pocillopora eydouxi*, massive *Porites* sp., and *Montipora* sp. exhibited lower levels of predation compared to the *Acropora* species. The presence of *D. cornus* on *Montipora* sp. suggests a behavior of opportunistic feeding, likely influenced by coral availability (Cumming 2009). At Couzoupa_S, *D. cornus* was associated with branching *Acropora*.

The prey preference of *Drupella* spp. exhibits dietary plasticity according to the abundance and availability of their favorite coral prey, as well as the “encounter probability” between them (Lei et al. 2022). The observations from the surveys carried out around Rodrigues are in line with prey preference observations through prey choice experiments by Al-Horani et al. (2011), and field observations by Kaullysing et al. (2016, 2017) and Lei et al. (2022), among others, where Acroporidae were found to be the preferred diet of *Drupella* spp. Acroporidae has high protein and energy content compared to other coral families (Keesing 1990), thus offering both nutritional benefits and shelter to *D. cornus*.

In conclusion, the lack of observed crown-of-thorns starfish across all surveyed sites in Rodrigues suggested relatively stable reef conditions. However, the presence of suspected COTS feeding scars highlights the need for continued monitoring to confirm whether COTS are present at low densities. In contrast, *D. cornus* was observed at multiple sites, with significant spatio-temporal variations in density, potentially linked to environmental stressors, coral degradation, or predator declines. Therefore, the study revealed that *D. cornus*, rather than COTS, posed a potential threat to Rodrigues reefs, particularly at Var Brûlée and Rivière Banane which can be considered as potential priority reef zones for intervention. The species exhibited a preference for *Acropora* spp. aligning with previous studies on *Drupella* feeding behavior. The preference of *D. cornus* for *Acropora* as prey highlights the potential susceptibility of these corals to *D. cornus* outbreaks, especially in reefs already affected by environmental stressors. Understanding these prey preferences is crucial for monitoring reef health and managing coral predator outbreaks to mitigate their ecological impact. Annual monitoring of *D. cornus* densities is recommended, particularly in shallow reef habitats, and the manual removal of gastropods can be considered once densities exceed 2 ind/m². Culling or removing corallivores has been shown to enhance short-term coral recovery, especially when thermal stress is low to moderate, corals have lower heat sensitivity, and corallivore recruitment rates

are high (Rogers and Plagányi 2022). Continued monitoring is warranted to ensure early detection in the event that environmental conditions change, but the current results of this study offer an encouraging snapshot of reef stability in Rodrigues in terms of corallivory by COTS and gastropods as none of the values, in terms of spatial distribution, indicate alarming densities so far. Particularly during coral bleaching events, increased monitoring and surveys of corallivores should be carried out as stressed corals are the most susceptible to predation, leading to reduced coral reef health and resilience.

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