State of Nature Report for the Caribbean Netherlands 2024

A second 6-year assessment of the Conservation State, threats and management implications for habitat and species in the Caribbean Netherlands

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Wageningen Marine Research report: C001/25

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Photo cover: An extreme example from St. Eustatius of how uncontrolled livestock husbandry can overgraze vulnerable slopes to the point at which even infrastructure at the top of the cliff comes in danger from erosion.

Photo: J. Hazenbosch

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Samenvatting

Caribisch Nederland (Bonaire, Saba en St. Eustatius) maakt deel uit van de Caribische "biodiversiteits-hotspot", met zeer hoge biodiversiteit, maar die tegelijk onder grote menselijke druk staat. Het herbergt ongeveer 130 endemische soorten en 143 internationaal bedreigde soorten met beleidsrelevantie. Het ministerie van Landbouw, Visserij, Voedselzekerheid en Natuur draagt de eindverantwoordelijkheid voor de uitvoering en handhaving van internationale natuurbehoudsverdragen voor deze eilanden. Dit brengt verplichtingen met zich mee en leidt tot beleidsvragen. Daarom wordt er elke vijf jaar een Natuurbeleidsplan opgesteld. Echter, sinds 2020, werd dat gecombineerd met aspecten van het milieubeleid en gepresenteerd als een integraal tienjarig Natuur en Milieubeleidsplan (NMBP). Voor de evaluatie van het natuurbeleid en het opstellen van nieuwe natuurbeleidsplannen is rapportage over de staat van de natuur (SvN) essentieel. Hiervoor werd de "Staat van Instandhouding" (SvI) bepaald, conform de aanpak zoals voorgeschreven door de Habitatrichtlijn (HR) van de Europese Unie.

Het eerste SvN-rapport betrof de periode tot 2017. Er worden in dit nieuwe rapport over verschillende soorten/soortgroepen die toen aan bod kwamen opnieuw gerapporteerd. Het betreft vooral soorten waarvoor in de tussenliggende jaren voldoende nieuw onderzoek is verricht om een herbeoordeling te kunnen te maken. Daarnaast worden hier ook negen nieuwe kwetsbare soorten/soortgroepen voor het eerst beoordeeld.

In de vorige beoordeling (tot 2017) werden 45% van habitats en 50% van soorten/soortgroepen van Caribisch Nederland beoordeeld als zijnde in matig ongunstige tot zeer ongunstige SvI. Wat we nu (tot 2024) zien is een verslechtering naar respectievelijk 61% en 71% in matig ongunstige tot zeer ongunstige SvI. Dit is mogelijk te wijten aan de opname van veel meer gevoelige soorten/soortgroepen voor het eerst deze keer. Voor beide beoordelingen waren de habitats in een iets minder slechte SvI dan soorten/soortgroepen. Dit verschil schrijven we vooral toe aan het grote aantal zeldzame soorten die op deze eilanden overleven in kritiek lage populatiegroottes. Lage populatiegroottes zijn inherent aan kleine eilanden waar er ook sprake is van een lage beschikbaarheid van leefruimte.

Op basis van deze nieuwste beschikbare analyses moet de SvI van de natuur in Caribisch Nederland algeheel als matig ongunstig tot zeer ongunstig worden beoordeeld. Hierbij moet worden opgemerkt dat vanwege het (algemene) gebrek aan gegevens na 2020, onze beoordeling van SvI de recentere invloed van het in 2020 geïmplementeerde NMBP nog niet voldoende kan meten.

Er zijn belangrijke verschillen maar ook overeenkomsten te melden tussen Caribisch Nederland en de EU. Zo is in de Europese Unie, ondanks een vergelijkbare achteruitgang met eerder, de SvI toch voor een aanzienlijk deel van de habitats (15%) en soorten (27%) zelfs als gunstig beoordeeld, hetgeen niet het geval is in Caribisch Nederland. Deels is dit ogenschijnlijk veel positiever resultaat mogelijk te verklaren doordat in Europa de monitoring zich vaak richt op algemene en wijdverspreide (en dus minder kwetsbare) soorten. In Caribisch Nederland ligt de focus juist op kwetsbare soorten en wordt een rechtstreekse vergelijking tussen cijfers daarom lastig. In tegenstelling tot Caribisch Nederland, is ook in de EU het aandeel habitats in een ongunstige SvI veel hoger dan voor soorten/soortgroepen. Dit verschil schrijven we vooral toe aan de hogere druk op natuur en leefgebied in de EU en de nog relatief lage invloed van verstedelijking, landbouw en industrie op het landgebruik in Caribisch Nederland. Pas recent heeft de druk van verstedelijking serieuze vormen aangenomen op Bonaire en, in mindere mate, op St. Eustatius.

Wanneer we kijken naar bedreigende factoren, identificeren we nu opnieuw dezelfde drie belangrijkste bedreigingen die brede gevolgen hebben voor de natuur van Caribisch Nederland, op zowel korte als midden-lange termijn. Dit zijn "loslopend vee", "invasieve soorten" en "klimaatverandering". Gelukkig richt de implementatie agenda van de huidige NMBP zich al op verschillende van de ernstigste

bedreigingen. Overbevissing en de eutrofiëring van kustwateren zijn vergelijkbare kwesties met grote gevolgen voor de natuur, maar worden niet afzonderlijk behandeld in dit rapport maar komen wel uitgebreid aan de orde in de hoofdstukken over het koraalrif en de visstand.

In Caribisch Nederland is klimaatverandering (net als in de EU) duidelijk een in belang toenemende factor, terwijl factoren zoals landbouw, landverlating en verstedelijking (nog) geen sleutelrol spelen omdat die op veel kleinere schaal zijn in relatie tot de aanwezige natuur. Behalve klimaatverandering (in de vorm van toenemende hitte, orkanen, zeeniveaustijging en veranderde neerslagpatronen) zijn de belangrijkste bedreigingen voor terrestrische habitats en soorten op dit moment vooral loslopend vee en invasieve soorten. Voor specifieke habitats spelen extra factoren een rol (zoals de eutrofiëring van kustwateren en ziektes, evenals overmatige bevissingsdruk in de kustgebieden van het koraalrifhabitat). Voor de in kolonies nestende broedvogels (zoals sterns) kan recreatieve verstoring door de mens worden toegevoegd als een snel groeiende bedreiging. De huidige toename van de bevolking en de geconcentreerde verstedelijking op Bonaire gaat gepaard gaan met een gebrek aan handhaving, onvoldoende regelgeving (voor recreatieve dichtheden en gedrag) en te weinig milieubeschermende maatregelen (zoals rioolwaterzuivering, ontwikkelingsplanning en richtlijnen voor landgebruik). Voor de mangroven van Lacbaai op Bonaire is de belangrijkste bedreiging de dichtslibbing met sediment waardoor het aquatische oppervlak van de baai vermindert (en daarmee ook mangrove-, zeegras- en vishabitat). Tegelijk vormt onbeperkt en overmatig recreatief gebruik van de baai een bedreiging voor de waterkwaliteit en resulteert het in de vertrapping van ondiepe zeegrasbedden. Tot slot, toont nieuw onderzoek naar verontreiniging rond de vuilstortplaats bij Lagun, dat chemische verontreiniging een opkomende milieubedreiging wordt voor de mariene habitatkwaliteit rond Bonaire.

Voor de in de analyse meegenomen mariene soorten/soortgroepen, zijn overbevissing en habitatdegradatie (de afname van het koraalrif) de belangrijkste factoren die de bijbehorende SvI beïnvloeden. De koraalriffen staan zwaar onder druk door watervervuiling (sediment en nutriënten afkomstig van het land), orkaanschade en hoge water temperaturen door klimaatverandering, allemaal factoren die ze ook extra kwetsbaar maken voor het toenemend aantal besmettelijke koraalziekten. Voor terrestrische soorten en soortgroepen zijn de drie belangrijkste schadelijke invloeden: a) de overbegrazing, voornamelijk door loslopend vee (die leidt tot verwoestijning, erosie, verlies van plantensoorten en grotere kwetsbaarheid voor klimaatverandering); b) predatie door invasieve roofdieren (waarvan de verwilderde huiskat de belangrijkste is); en; c) genetische verdringing door geïntroduceerde invasieve leguanen. Deze drie invloeden zijn rechtstreeks toe te schrijven aan het overkoepelend probleem van invasieve soorten (loslopende soorten vee zijn invasieve soorten). Sinds de laatste inventarisatie (2011 en 2012) zijn maar liefst 710 nieuwe meldingen van niet-inheemse soorten in de Nederlandse eilanden (waar Caribisch Nederland deel van uitmaakt). Meer dan de helft betreffen planten. Het huidige gemiddelde aantal soorten die zich nieuw vestigen is hoger dan 54 soorten per jaar. Invasieve soorten vormen een extreme bedreiging voor de biodiversiteit, maar tot nu toe zijn er slechts een paar van deze soorten aangepakt in pilotstudies en korte-termijn opportunistische projecten. Er is tot nu toe geen fytosanitaire wetgeving om dit extreem probleem beheersbaar te maken.

De invloeden van menselijke activiteiten in Caribisch Nederland zijn zo groot en alom tegenwoordig, dat actieve interventie belangrijker is dan ooit en essentieel voor het omkeren van negatieve trends en feedbackloops (bijvoorbeeld tussen het verlies van plantensoorten en klimaatkwetsbaarheid, zoals veroorzaakt door overbegrazing). Bij het stellen van prioriteiten voor interventies is het derhalve belangrijk om te concentreren op maatregelen die meerdere voordelen hebben (in plaats van acties die gericht zijn op enkelvoudige oplossingen) en op maatregelen waarbij het liefst gebruik gemaakt wordt van de eigen veerkracht van de natuur (zogeheten “Nature-based Solutions”). Uitzonderingen op deze regel kunnen gerichte acties zijn die nodig zijn om bepaalde iconische en endemische soorten of populaties te beschermen. Het rapport wordt afgesloten met een korte lijst van prioriteiten voor beschermingsmaatregelen, monitoring en toegepast wetenschappelijk onderzoek.

Abstract

The Caribbean Netherlands (Bonaire, Saba, St. Eustatius) is part of the Caribbean "biodiversity hotspot," which has very high biodiversity and is under significant human pressure. It hosts about 130 endemic species and 143 internationally threatened species of policy relevance. The Netherlands Ministry of Agriculture, Fisheries, Food Security and Nature has the final responsibility for the implementation and enforcement of international nature conservation treaties for these islands. This comes with policy and management obligations and raises various policy questions. To address these, a Nature Policy Plan is drawn up every five years. However, as of 2020, this was combined with aspects of an environmental plan and presented as an integral ten-year Nature and Environment Policy Plan (NEPP). For the evaluation of the results of nature policy and the drafting of new nature policy plans, reporting on the State of Nature (SoN) is essential. For this, we assessed the “Conservation State” (CS) of habitats and species according to the methods prescribed by European Union’s Habitats Directive (HD).

The first SoN reporting round addressed the period up to 2017. Several species or species groups and/or habitats addressed then, are reported on here again. Those are particularly the habitats and species for which sufficient new research was conducted in the intervening years to makes a re-assessment relevant. In addition, nine new vulnerable species and species groups are assessed here for the first time.

Based on these newest results, we conclude that, without exception, the SoN in the Caribbean Netherlands must be assessed as unfavourable to very unfavourable. However, it is important to note that due to the general lack of data after 2020, our assessment cannot fully measure the more recent effects of the NEPP as implemented in 2020.

In the previous assessment period, 45% of assessed habitats and 50% of assessed species and species groups of the Caribbean Netherlands were considered to be in a unfavourable-bad CS. What we see now is a worsening trend in which these percentages have now increased to respectively, 61% and 71%. This could in part be due to the inclusion for the first time of many more sensitive species/species groups. In both assessments, habitats were in a less unfavorable CS than species/species groups. We primarily attribute this difference to the large number of rare species that survive on these islands in critically low population sizes. Low population sizes are inherent to small islands, which are characterised by low availability of living space.

There are significant differences but also similarities to report between the Caribbean Netherlands and the EU. For instance, in the EU, despite a similar decline as compared to earlier assessments, the CS is still assessed as favorable for a significant portion of habitats (15%) and species (27%), which is not the case in the Caribbean Netherlands. This seemingly more positive result can partly be explained by the fact that in the EU, monitoring often focuses on common and widespread species. In the Caribbean Netherlands, the focus is specifically on vulnerable species, making a direct comparison between the figures difficult. Additionally, but unlike in the Caribbean Netherlands, the EU has a much higher proportion of habitats in an unfavorable CS compared to species/species groups. We primarily attribute this poorer CS of habitats as opposed to species to the higher pressures on wilderness in the EU and the still relatively low impact of urbanization, agriculture, and industry in the Caribbean Netherlands. Only recently has the pressure from urbanization taken a serious form on Bonaire and, to a lesser extent, on St. Eustatius.

As for threat factors, we identify the same three main threats with broad implications for the nature of the Caribbean Netherlands as in the earlier assessment. These are "free-roaming livestock", "invasive species" and "climate change." Overfishing and the eutrophication of coastal waters also have major consequences for nature, but are not addressed separately in this report. However, they are extensively

discussed in the chapters on coral reefs and fish stocks. Fortunately, the implementation agenda for the current NEPP already addresses several of the most serious threats.

In the Caribbean Netherlands, climate change is (as in the EU) clearly increasingly important, while factors such as agriculture, land abandonment and urbanization (as of yet) do not play a key role because they are on a smaller scale in relation to the available natural habitat. Aside from climate change (which includes increasing temperatures, hurricane impacts, rising sea level and altered rainfall patterns), key threats to terrestrial habitats and species at present are especially roaming livestock and invasive species. For specific habitats, additional factors come into play (such as the eutrophication of coastal waters and diseases, as well as excess fishing pressure in near-shore areas of the coral reef habitat). For colonial nesting birds (like terns) human recreational disturbance can be added as a growing problem. Current human population size increases and clustered urbanization on Bonaire wouldn't even be so problematic if it were not for the lack of sufficient restrictions (to recreational densities and behaviour), environmental safeguards (like sewage treatment, development planning and guidelines for land clearance) and enforcement. For the mangroves of Lac Bay in Bonaire, accumulated sediments which reduce the aquatic habitat surface of the bay and thereby destroy mangrove and seagrass habitats, can be identified as the principal threat, whereas unrestrained and excessive recreation in the bay is a threat to water quality and sea grass beds due to trampling. Finally, new research on contaminants leaching from the landfill at Lagun, suggests that chemical contaminants are an emerging environmental threat to marine habitat quality, certainly around Bonaire.

For the marine species and species groups studied, overfishing and habitat degradation (coral reef decline) are principal factors impacting their CS. For terrestrial species and species groups the three main deleterious factors identified are overgrazing, principally by uncontrolled roaming livestock (which cause aridification, erosion, plant species loss and greater vulnerability to climate change), predation by invasive predators (foremost of which is the feral cat) and genetic swamping due to introduced invasive iguanas. Hence, all three of these impacts are directly ascribable to the overarching problem of invasive species.

Since the last inventory (2011 and 2012) no less than 710 new island records have been compiled of non-native species entering the wild on one or more Dutch Caribbean islands. Over half of these are exotic plants. The current rate of increase exceeds 54 species per year entering the wild. Invasive species amount to an enormous risk to biodiversity, but only a few have so far been addressed in pilot studies and short-term opportunistic projects. There is as yet no phytosanitary legislation to help stem this extreme threat.

The impact of man's activities in the Caribbean Netherlands has become so large and pervasive, that active intervention is more important than ever and essential to reversing negative trends and feedback loops (for instance between plant species loss and climate vulnerability, as caused by overgrazing). In suggesting and setting priorities for conservation interventions it is important to focus on actions that have multiple cascading benefits instead of actions directed to single solutions. Preferably the interventions should also focus on Nature-based Solutions to help make use of nature's own resilience. Exceptions to this rule might be highly specific actions needed to safeguard certain iconic endemic species or populations. Our assessment is concluded with a short list of priorities to keep in mind for conservation action, monitoring and scientific research.

List of Abbreviations

AICOM - Áreas de Importancia para la Conservación de Murciélagos
AIS – Automatic Identification System
BD – Birds Directive
BO – Beleidsondersteunend Onderzoek
BWM - Ballast Water Management Convention
CARICOM – Caribbean community
CBD – Convention on Biological Diversity
CBDB – Caribbean Biodiversity Data Base project
CBS – Centraal Bureau voor de Statistiek
CITES – Convention on International Trade in Endangered Species
CL - Carapace Length
CMS – Convention on Migratory Species
CN – Caribbean Netherlands
CPUE – Catch per unit effort
CS – Conservation State
DCNA – Dutch Caribbean Nature Alliance
EEZ – Exclusive Economic Zone
EU – European Union
EZ – Economische Zaken (Economic Affairs)
FP – Fibropapillomatosis
FRR – Favourable Reference Range
FRV - Favourable Reference Value
GDP – Gross Domestic Product
HD – Habitats Directive
HEN (Curaçao Ministry of Health, Environment and Nature)
IAC – Inter-American Sea Turtle Convention
IAS – Invasive Alien Species
IBA – Important Bird Area
ILOS – International Law of the Sea
IMARES - Institute for Marine Resources & Ecosystem Studies
IMO – International Maritime Organization
IPCC – International Panel on Climate Change
IPPC - International Plant Protection Convention
IUCN - International Union for Conservation of Nature
KNMI - (Royal Netherlands Meteorological Institute)
LVVN – Ministry of Agriculture Fisheries, Food Security and Nature (Landbouw, Visserij, Voedselzekerheid en Natuur)
MVP – Minimum Viable Population
MWTL – Monitoring Waterstaatkundige Toestand des Lands
NEPP – Nature and Environmental Policy Plan, Caribbean Netherlands
NGO – Non-Gouvernemental Organization
NbS – Nature-based Solutions
NSF - National Science Foundation (USA)
NWO – Dutch Research Council (Nederlandse Organisatie voor Wetenschappelijk Onderzoek)
OLB – Island Gouvernment (Openbaar Lichaam) of Bonaire
OLE – Island Gouvernment (Openbaar Lichaam) of St. Eustatius
OLS – Island Gouvernment (Openbaar Lichaam) of Saba
PES – Public Entity Saba
PRECIS - Providing Regional Climates for Impact Studies

RCP - Representative Concentration Pathways
RELCOM - The Latin American and Caribbean Network for Bat Conservation
RHI – Reef Health Index
SCF – Saba Conservation Foundation
SENA - Stichting Encyclopedie van de Nederlandse Antillen
SICOM - Sitios de Importancia para la Conservación de Murciélagos
SoN – State of Nature
SPAW – (Protocol Concerning) Specially Protected Areas and Wildlife
STCB – Sea Turtle Conservation Bonaire
STENAPA – St Eustatius National Parks Foundation
STINAPA – Stichting Nationale Parken Bonaire
SVL - Snout-Vent Length
TEEB – The Economics of Ecosystems and Biodiversity
UNCLOS - The United Nations Convention on the Law of the Sea
USFWS - United States Fish and Wildlife Service
WMR – Wageningen Marine Research
WOT – Wettelijke Onderzoekstaken
WOTRO – Wetenschappelijk Onderzoek in de Tropen

Introduction to the 2024 State of Nature Report for the Caribbean Netherlands

Frameworks and Context

In the European Union (EU), countries are required by the EU Habitats Directive (HD) (Art. 17) to report on the "State of Nature" (SoN) of nature every six years. For the Birds Directive (BD) only the sizes and trends of bird populations are reported. Since 10-10-2010, the Caribbean islands of Bonaire, St. Eustatius, and Saba (together known as the Caribbean Netherlands) have been part of the Netherlands as public entities. The current Netherlands Minister of Agriculture, Fisheries, Food Security and Nature is thus directly responsible for the implementation and execution of international treaties for these islands: the Convention on Biological Diversity (CBD) which require National Biodiversity Strategies and Action Plans (NBSAPs), the Convention on International Trade in Endangered Species (CITES), the Bonn Convention, the Ramsar Convention, the Cartagena Convention/Specially Protected Areas and Wildlife (SPAW) protocol, the Inter-American Sea Turtle Convention, and others such as the United Nations Convention on the Law of the Sea (UNCLOS). The Ministry of Agriculture, Fisheries, Food Security and Nature has frequently changed names. From 2003 to 2010 it was known as the Ministry of Agriculture, Nature and Food Quality, from 2010 to 2012 together with Economic Affairs it was known as the Ministry of Economic Affairs and Innovation, from 2012 to 2017 it was part of the Ministry of Economic Affairs, since the end of 2017 it was separately known as the Ministry of Agriculture, Nature, and Food Quality, and since 2023 as the Ministry of Agriculture, Fisheries, Food Security and Nature. The responsibilities of the ministry entail obligations and lead to various policy questions. To address these, a nature policy plan is developed every five years. However, as of 2020, this is combined with aspects of an environmental plan and presented as an integral ten-year Nature and Environment Policy Plan (NEPP).

Nature policy is not only about ensuring vital nature with rich biodiversity but also about protecting and sustainably using our natural capital (Min. EZ, 2013; Min. LNV et al., 2020). With this is meant the stock of natural ecosystems that provides a flow of valuable products and services to humans. There is increasing knowledge about the economic value of such ecosystem services, such as natural coastal protection, water purification, pollination, pest control, and space for tourism and recreational use (de Knegt, 2014). However, the European Netherlands economy is less dependent on these ecosystem services than are the Caribbean Netherlands, where nature-based tourism is of utmost importance and fisheries is of (relatively) greater importance to the local economy than in the European Netherlands. TEEB research (The Economics of Ecosystems and Biodiversity) on the economic value of nature in Bonaire, St. Eustatius, and Saba has shown an annual Total Economic Value of nature (TEV) amounting to USD 105 million, 25.2 million, and 28.4 million, for these islands respectively (Cado van der Lely et al., 2013, 2014a, 2014b). The share of nature-based tourism was found to be 48% for Bonaire (Schep et al., 2012) and 12% and 27% for St. Eustatius and Saba, respectively (van de Kerkhof et al., 2014a, 2014b). These findings highlight the special significance of nature to these islands (van Beek et al., 2015). Based on these figures, it was shown that in 2013 the economic value of ecosystem services for Bonaire, Saba, and St. Eustatius represented 31%, 63%, and 24% of the gross domestic product (GDP), respectively (CBS, 2014). Hence, the Netherlands Caribbean island communities intimately depend on nature.

The Caribbean Netherlands is part of the "The Caribbean Islands" biodiversity hotspot (Myers et al., 2000; Mittermeier et al., 1999). A biodiversity hotspot is a biogeographical region with very high biodiversity, often with many endemic species (limited to a very small distributional area; also often referred to as "range-restricted") (Debrot, 2006; Bos et al., 2018), but also many threatened species (see Appendix 1). The Caribbean Netherlands does not fall under the so-called "European Directives", so there are no formal obligations arising from them. However, treaties such as CITES, Ramsar, Convention on Migratory Species (CMS), Cartagena (SPAW-protocol), and the Convention on Biodiversity (CBD) do apply. These in turn, lead to formal international obligations such as reporting on the status and threats to habitats and species in the Dutch Caribbean islands (Jongman et al., 2009; Verweij et al., 2015).

Within the European Union legal context, reporting on the State of Nature is essential for evaluating implemented nature policies and for drafting new nature policy plans. For consistency and comparable policy planning purposes, and following the European Union example, the Netherlands decided to apply a similar process in the Caribbean Netherlands even though the process there does not carry the same legal ramifications. The first SoN reporting was done for the Caribbean Netherlands up to 2017 (Debrot et al., 2018). However, SoN reporting, are also needed every five years in the Caribbean Netherlands for the CBD Convention and the SPAW protocol of the Cartagena Convention. To meet these various needs and obligations efficiently, suitable indicators must be chosen as well as an analysis and reporting framework that simultaneously addresses all the different needs. For the Caribbean Netherlands SoN reporting, a methodology was chosen that closely aligns with the methodology used for determining the Conservation State (CS) as used in the European Habitats Directive (HD). In this assignment, the SoN in the Caribbean Netherlands is reported on for the second time. Such reporting is to be done every five years; to parallel the evaluation of the Caribbean Netherlands Nature and Environmental Policy Plan (NEPP) and the development of new policy plans every five years as required by Dutch law (Wgnb BES) (i.e., a shorter reporting cycle requested than the EU 6-year reporting cycle).

The concept of CS is explained by the European Commission in the document DocHab-04-03/03 rev.3. According to the definition, the national CS for a habitat type or species (by assessing distribution area, population, or surface area) can only be considered favourable if there is a stable or positive trend and the value is above a certain threshold. This threshold is the so-called favourable reference value (FRV). FRVs are essential for assessing the CS and must be based on scientific knowledge. They can only be decided after sufficient comparative studies have been conducted.

The CS for a species (DocHab-04-03/03 rev.3) is assessed (in relation to the FRV) based on:

- Distribution area
- Population
- Habitat (extent, quality, and trend)
- Future prospects

For habitats, the criteria are slightly different (surface area instead of population size and quality instead of habitat extent, but habitat also concerns quality). The specified criteria are:

- Distribution area
- Surface area;
- Quality;
- Future prospects

This assignment addressed these aspects for several of the most important habitats and species (threatened, key, and indicator habitats and species) for which sufficient knowledge is available. Unfortunately, due to the structural lack of knowledge and monitoring of most of the biodiversity in the Caribbean Netherlands, a quantitative report for most species and species groups was still not possible in this second reporting round. Nevertheless, major strides have been made with as a result that baseline CS reporting this time has become possible for 9 new species (e.g., Bridled Quail-dove) or species groups (e.g. bats) for which no reporting could be provided in 2018.

Thus, in this assignment, the CS of habitats and some important species or species groups (threatened, key, and indicator species) for which sufficient knowledge exists is reported. The report provides insight into, among other things, population dynamics aspects of the involved species, the area of distribution and habitats, and trends in threatening factors.

The purpose of this report was to:

- Report on the SoN in the Caribbean Netherlands and where possible compare the 2024 CS to the 2017 CS for certain habitats, species groups or species.
- Gain insight into the key pressures (i.e., threats) affecting the CS of species and habitats so as to help prioritize conservation efforts.
- Make an essential contribution to the required five-year evaluation and definition of objectives for the nature policy plans for the Caribbean Netherlands.
- Provide insights for the adjustment and/or expansion of monitoring indicators for future reporting purposes.
- Largely fulfil reporting obligations arising from the Netherlands' involvement in the SPAW Protocol of the Cartagena Convention and the CBD Convention.

Limitations of this Report

In the European Netherlands and Europe, reports regarding the SoN are driven by extensive monitoring programs. These do not exist in the Caribbean Netherlands with few exceptions (like sea turtles and coral reefs). Most biological-scientific research conducted there since the 1960s has been (and still is) primarily descriptive and taxonomic in nature or motivated and funded by broader scientific interest. As a result, simple applied management-oriented quantification, which is necessary to monitor the status of threatened elements, is generally unavailable barring a few exceptions. Jongman et al. (2009) previously pointed out the pressing need for even the most basic inventories. Practical policy questions are usually not of key interest to the main, pure-science funding channels (such as NWO, WOTRO, NSF) because they do not involve "cutting-edge" scientific questions. European Member States report, for example, quantitatively on all breeding birds at the species level (EEA, 2015). This varies from country to country, and the number of reported species ranges from 27 (Malta) to 340 (Spain) (EEA, 2015). Reports like those which are routinely produced for the Netherlands, and for instance in which 76 species can be meaningfully treated quantitatively (e.g., Ottburg and van Swaay, 2014), are simply impossible based on the existing short-term, project-based funding for the Caribbean Netherlands. With a few notable exceptions, there is practically no quantitative biological monitoring taking place. Nevertheless, thanks to substantial funding from the Netherlands Ministry of Agriculture, Fisheries, Food Security and Nature, there have been major advances since 2010 in basic quantitative management-oriented research and knowledge. Also, due to the time required for analysis and writing of monitoring reports, it often happens that in the meantime new data, new aerial photographs or new measurements become available that cannot be included in the analysis due to time constraints. It is normal that by the time a monitoring report comes out, there are new developments, or new data available to report on that have not yet been possible to incorporate into the assessment. The SoN report is intended to inform and to provide input for nature policy and management but is not itself a policy- or implementation plan.

Another limitation of this report is that nature can not only be managed merely through nature policies but requires incorporation by and integration through other policy areas. Therefore, other policy areas and related issues (to a greater or lesser extent) affect the CS of habitats and species discussed in this report (such as land use, spatial planning, agriculture, waste(water)management, tourism, immigration, economic development). However, the focus in this report is on those policy issues affecting CS and which normally fall inside the scope of nature management.

Terms, Concepts, and Definitions

The question addressed by this report concerns whether the nature of the Caribbean Netherlands is in a "favourable" or "unfavourable" CS. It is crucial to have a clear understanding of the term CS and how it should be determined and scored. We therefore used a methodology that closely aligns with CS as defined by the Habitat Directive. In the Netherlands, the term CS relates to specific species and habitat types under the European Habitat Directive. While these guidelines do not apply to the Caribbean Netherlands, the definitions and approach used have served as a guideline for determining the CS of Caribbean nature.

The term CS can refer to the overall condition of a species or habitat and is also used to describe the condition within a smaller area. Given that Caribbean Netherlands often deals with relatively small natural areas and species with large habitats or strong migratory patterns, it is important to make a clear distinction in their treatment.

Habitats
According to the Habitat Directive, the CS of a natural habitat is considered "favourable" when (Ministry of Economic Affairs, 2014):

a) The natural distribution area of the habitat and the area of that habitat within that area are stable or increasing, and
b) The specific habitat structure and functions necessary for long-term conservation exist and are likely to continue to exist in the foreseeable future, and
c) The CS of the species typical for that habitat is favourable.

Species
According to the Habitat Directive (HD), the CS of a species (see Table 1.1) is considered "favourable" when (Ministry of Economic Affairs, 2014):

a) The species concerned remains a viable component of the natural habitat in which it occurs and is likely to remain so in the long term, and
b) The natural area of occurrence of that species is not decreasing or is not likely to decrease in the foreseeable future, and
c) There exists and is likely to continue to exist a sufficiently large habitat to maintain populations of that species in the long term, and
d) The species has "future prospects" based on the above three conditions.

The criteria used for habitats and species are therefore aligned as closely as possible. To provide a well-founded assessment, data on ecology and population dynamics, information on the natural distribution area, and the size of the available habitat are necessary. This approach necessarily relies on a "favourable reference" against which distribution area and population status must be compared. Each aspect can have four relative values: "favourable", " unfavourable-inadequate", "unfavourable-bad", or "unknown" (EEA, 2015).

Even with such a structured approach, there are underlying concepts that need to be defined, such as "favourable reference values", "viable", "natural habitat", "long term", and "natural distribution area". Often, experts assessing a particular species or category provide detailed evaluations of these concepts. Hence, there are no uniform definitions of the underlying concepts. We will therefore not develop them here for the Caribbean Netherlands. There is considerable scientific knowledge about most (European) Habitat Directive species. However, this is not the case for Caribbean species. While the choice of assessments must be based on scientific knowledge (i.e. data), in many cases, "expert judgment" cannot be avoided (see, for example, Ottburg and van Swaay, 2014). Of course, dependence on "expert judgment" is even greater in the case of Caribbean Netherlands nature, due in many cases to the lack of essential data.

In the many cases where knowledge is lacking, so-called "rules of thumb" are often used in the Netherlands. The choice of "favourable reference value" (FRV) illustrates how, for birds in the Netherlands, a "reference year" was chosen quite practically and arbitrarily without any ecological or scientific basis. For birds in the Netherlands, the "reference year" is 1990 purely because "breeding bird population trends 'generally' begin in 1990" (Teunissen et al., 2015). Since there are hardly any quantitative population counts of species in the Caribbean Netherlands, it is not very useful to establish a similar reference year. We have attempted to apply the same method as with the HD by using an FRV, however, even for birds, due to lack of data, this has rarely been possible.

Table 1. *Systematics for the assessment of the Conservation State (CS) of a species (Ministry of Agriculture, Nature and Food Quality, 2006).*

Parameter	Favourable	Unfavourable – inadequate	Unfavourable - bad	Unknown
Distribution range	Range stable or increasing. Not less than the 'favourable reference".	Between favourable and very unfavourable.	Range loss of more than 1% per year, or range more than 10% less than "favourable reference".	No or insufficient reliable information.
Population	Population equal to or greater than the favourable reference. Reproduction, mortality, and age structure not worse than normal.	Between favourable and very unfavourable.	Population decline of more than 1% per year. Below the favourable reference. Population more than 25% lower than the favourable reference. Or reproduction, mortality, and age structure much worse than normal.	No or insufficient reliable information.
Habitat quality	Habitat is sufficiently large (and stable or increasing). The quality is suitable for the long-term survival of the species.	Between favourable and very unfavourable.	Habitat is clearly insufficient in size for the long-term survival of the species. Or the quality is clearly unsuitable for the long-term survival of the species.	No or insufficient reliable information.
Future prospects	The main threats are not substantial. The species will be viable in the long term.	Between favourable and very unfavourable.	Strong negative impact of threats on the species. Very poor outlook. Long-term viability at risk.	No or insufficient reliable information.
Overall Assessment of Conservation State	All green or three green and one unknown.	One or more orange, but no red.	One or more red.	Two or more unknown combined with green.

Working Definitions

Minimum Viable Population Size
A “minimum viable population” (MVP) means a 95% probability of survival over the next 100 years (Frankham et al., 2014; Traill et al., 2007). To determine the Favourable Reference Values (FRVs), the “Minimum Viable Population” (MVP) concept is used. This concept refers to the "minimum effective population size needed for viability based on genetic parameters." Sensitivity to inbreeding depression varies greatly between species and depends on many factors. For the “effective” population size (i.e., without genetic losses), the values typically range between 50 and 1000 “effective” adult individuals (Frankham et al., 2014). Populations below 50 individuals are at high risk of short-term extinction, while populations below 500 or 1000 are at risk of long-term extinction. To achieve an effective population size of 500, 526 to 50,000 individuals are needed, depending on the randomness of mating (Ottburg and van Swaay, 2014). Since (even in Europe) the required genetic information is often lacking, the rule of thumb is generally to set MVP at 1000 adult animals per subpopulation of vertebrates” (Ottburg and van Swaay, 2014). For invertebrates, Ottburg and van Swaay (2014) recommend using Traill et al. (2007), which suggests several thousand adult animals. In this report, we follow Ottburg and van Swaay (2014). FRVs were rarely available due to the absence of such studies. Tracking population sizes of species is a priority in conservation and key trend indicator (Geldmann et al., 2023). Population size is considered an Essential Biodiversity Variable (EBV) that reflects essential processes such as reproductive success, carrying capacity, susceptibility to extinction, and a species’ role in the functioning of ecosystems (Kissling et al., 2018).

Viable Component
A population of a species in its natural habitat that is "minimally stable" and of "sufficient size" to withstand "population fluctuations." As previously indicated, except for a few exceptions (such as the Caribbean Flamingo and the Yellow-shouldered Amazon), there are no time series of population estimates for rare species. At best, there are some snapshots spread across several decades. Even for the best-studied group, namely corals, there are no population estimates, though this is notoriously difficult because these organisms are clonal animals and a relationship between size and age is likely to be seriously flawed. Therefore, it is usually not possible to provide concrete estimates of population size and/or stability. The best that can be done is to extrapolate from small sample plots to the entire habitat to develop trend lines. The justification for these derived trend lines is scientifically accepted, as evidenced by the many scientific publications produced in this way.

Long Term
The time period in which the "future prospects" of the species can be "reasonably" foreseen.
For EU reporting, this is now set at 2 reporting periods = 12 years. Developments on the islands of the Caribbean Netherlands often proceed rapidly, mainly due to economic pressures, small habitat areas, and small numbers of animals/plants involved. This means that even small and unpredictable events can have a significant and unforeseen impact on a species or habitat. Therefore, it is especially difficult to determine the time period for which the "future prospects of the species can be reasonably foreseen." In this study, we base our statements mainly on the average lifespan of the respective species. For the Caribbean Flamingo and Yellow-shouldered Amazon, this can be estimated at perhaps 15 years, while for the Lesser Antillean Iguana on St. Eustatius, it is closer to 25 years.

Natural Range
Geographical area where a species has established itself "independently" and "permanently".
Because endangered and vulnerable species typically start from abnormally low population densities, it's not surprising that a species might have never been recorded in many habitats where it could potentially occur. This poses the risk that the geographic distribution based on local data significantly underestimates the potential range. Statements regarding this are therefore based on insights and knowledge from the respective experts of each species or category, drawing from their understanding of the ecological literature on the species, species group, or specific habitat.

Natural Habitat
A habitat determined by specific abiotic and biotic factors where the species lives during "one of the stages" of its life history. Over the centuries, human activities such as deforestation, cultivation, and the introduction of exotic grazers and predators have heavily influenced nature on all three islands. For many species, the "natural" habitat has been extensively altered due to human impact. In addition, to the same considerations outlined for "natural range," the question arises of how current habitat use compares to former natural habitat use. An example is the Lesser Antillean iguana, which on St. Eustatius often appears to choose habitat and reaches highest densities in human-inhabited areas. The species also seems scarcely present in the highest parts of the island above 300 meters above sea level. It's unclear whether this is due to habitat preference or suitability, given that the species is known elsewhere from 300 m or higher, which currently seem not to be used on St. Eustatius. Similar to situations in mainland Netherlands (Ottburg and van Swaay, 2014), "expert judgment" has been used in these cases to balance scarce local knowledge against available international scientific knowledge.

Typical Species
The approach using typical species as an indicator of habitat condition has been developed for the monitoring of the CS of Natura 2000 habitat types in the EU. To some extent, this approach can also be applied for monitoring habitats in the Caribbean Netherlands, although most habitats are in reality a collection of different habitat types. The use of species with which to monitor habitats is based on many assumptions and requires a level of detailed ecological understanding that is not yet really feasible for widespread use in the Caribbean Netherlands. An example that could be partially useful might be to use the relative cover of *Thalassia testudinum* seagrass as an indicator for seagrass bed habitat health.

Typical species meet the following criteria (Ministry of Economic Affairs, 2014):
a) The species is a good indicator of the favourable CS of the habitat type and should be able to be measured non-destructively and inexpensively;
b) The composition of the list of typical species per habitat type should remain stable in the (medium to) long term.

Typical species are preferably defined as follows: "typical species are species that cannot be separated from the habitat type, other than the species with which the habitat type is defined." For the Netherlands, there are two categories of typical species (Ministry of Economic Affairs, 2014):

- **Exclusive and characteristic** species, i.e., species whose ecological requirements occur only or primarily in the respective habitat type;

- **Constantly present** species, i.e., species present in every area with the respective habitat type, but not limited to the habitat type itself.

Notes

The concept of CS is applied in Europe to an individual habitat type or an individual species of the HD. In the case of the Caribbean Netherlands, this concept is also applied to birds and clusters of related species. While little can often be said about individual species in the Caribbean Netherlands, it is sometimes possible to say something about groups of species that are ecologically comparable. In this report, this has been done, for example, for groups such as turtles and terns. This deviates from the systematics of the BD and HD. We prefer reporting per species but report now for certain species groups due to lack of data on individual species.

The HD focuses on specific habitat types, such as grey dunes or white dunes. A similar specification could also be possible in the Caribbean, but we have had to limit ourselves to generic habitats such as 'Coral Reefs' and 'Dry tropical forests'. In other words, the HD makes a clear distinction between the terms "habitat" and "habitat types", which we do not make here. The use of "typical species" gives a rough simplification of the system designed for monitoring trends. Therefore, they are not a good substitute for a true understanding of what is actually happening in an ecosystem. Not only typical species, but the ecosystem itself should ideally be monitored to better understand what is happening in the system.

Threats

Finally, this report also addresses several of the most significant threats to nature, which have broad consequences for many species and habitats. These are threats which are not temporary or minor but that are structural and growing or which periodically impact populations or habitats to the point at which long-term survival of one or more species is in danger. These are discussed separately in detail. The state of certain threats largely determines the CS of nature and represents an ecosystem approach rather than an individual "species approach". These issues such as "invasive species", "free-roaming livestock", and "climate change" are discussed as such in this report. The factor of "overfishing" is also a major issue deserving separate reporting but is fairly covered in our fish stock report. Additionally, factors such as coastal development, erosion, and eutrophication due to wastewater should not be overlooked (e.g., Debrot and Sybesma, 2000).

Acknowledgements and Guide to Reading

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In this chapter, we have discussed the limitations in assessing the SoN, and the chosen approach to doing this as applied to the sections devoted to habitats and species/species groups. The habitats section begins with a GIS map of the different areas and corresponding surface coverages as a basis for the description of ten terrestrial and marine habitats on the islands of Bonaire, St. Eustatius, and Saba. Habitats form the living space for species. The CS of each discussed habitat is described in separate chapters, ranging from the terrestrial vegetations of the islands to the open sea and deep sea. A total of ten chapters are dedicated to one or more specific habitats. Only highly rare and endangered freshwater habitats (e.g., Debrot, 2003) and *Lithothamnion* reefs (Zaneveld, 1958; Foster et al., 2013) are not treated in this report. Each paragraph first provides an indication of the international legal protection status before giving a description and assessment of the CS. Each chapter is then concluded with a bibliography, compactly bringing together most key information currently available per habitat.

An overview of more than 100 policy-relevant species is presented in Appendix 1. After habitats, the next section discusses a selection of species and species groups which only represent a part of the total biodiversity of these islands based on data availability. The format used is broadly similar to the approach in the habitat section. A total of 14 species/species groups are treated, nine of which for the first time, while five were already discussed in the 2018 SoN report. The species and/or species groups treated previously but not this time are as follows: the Yellow-shouldered Amazon and flamingo of Bonaire, the Queen conch and the Cetaceans of the Caribbean Netherlands. For a sufficiently up-to-date treatment of these groups, we refer to our 2018 report. The third section of this report discusses three major threats to the biodiversity of the islands while the fourth section presents some key conclusions regarding the overall SoN for the biodiversity in the Caribbean Netherlands in 2024.

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Part 1: Habitats

The Caribbean Netherlands is part of the habitat of at least 143 species of international policy relevance (Annex 1). and home to approximately 130 endemic species. Such species richness can only develop and be maintained if there is sufficient habitat of good quality present. Without sufficient and suitable habitats, many species would disappear and biodiversity would be reduced to a minimum of a few generalist species. When that happens, overall ecosystem resilience also declines. The Caribbean Netherlands possess many habitats which allow the islands to support a rich biodiversity. This is in part due to the fair habitat quality, habitat coverage and habitat inter-connectedness. Habitat inter-connectedness is particularly critical as most species will need more than one habitat during the course of their life cycle. Nevertheless, it must be noted that across the board, habitat quality is under threat and changing and in some cases habitat coverage is also changing for the worse.

Habitats, communities, and ecosystems each have their inherent diversity (termed Alpha-diversity and referring to the number species of found in that habitat). In this respect, coral reefs for instance are one of the most species-rich and diverse habitats in the world. The differences in diversity, or species composition, between habitats is referred to as Beta-diversity (Whitaker, 1972). This is based on the simple fact that different species typically thrive under different habitat conditions, and hence certain species will be limited to different habitats. For instance, the species living in a saline lake will be totally different from those living on a coral reef. Finally, Gamma-diversity refers to the combined species abundance across habitats at a geographic scale (Hunter et al., 2012).

In general, habitat CS changes at a slower rate than does species CS. Habitats are generally more stable than species as they are the summation of many species and because the most essential component of habitats is the physical environment (as defined by climate and geology) and the strongly associated "vegetation formation", which typically responds at a slower rate than the many individual species that depend on them. For this reason, habitat monitoring at longer intervals can be sufficient whereas species monitoring should typically be done at shorter intervals.

In this section, and for the purposes of this second report on the State of Nature in the Caribbean Netherlands, several habitats are highlighted where sufficient information is available to make a substantiated assessment. There are certainly more habitats for which enough is known to allow for meaningful reporting, even if only in summary form (e.g. freshwaters, anchialine and subterranean waters). But these have not yet been included and the selection presented here is purely pragmatic, based on the quality and availability of data.

Habitat coverage, quality and connectedness are key for monitoring of priority species (Verweij et al., 2015) and are among the most important research questions. However, for the Caribbean Netherlands, there are very few legally defined habitat-oriented environmental obligations (Jongman et al., 2009). In that respect, most biodiversity treaties are too species-focussed and only few (e.g., Ramsar Convention) account for habitats.

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1 Overview of Habitats, Habitat Coverage and Maps

Verweij, P. J. F. M. and Mücher, C. A. 2025. From: State of Nature Report for the Caribbean Netherlands, 2024. WUR report C001/25.

Introduction

Global biodiversity is declining due to habitat destruction and degradation, mainly because of land use and climate change (Hanssen et al., 2004; Mücher et al., 2009). Within the EU, many habitat types are protected under the European Habitat Directive because they form the essential conditions for the protection of flora and fauna. In the Caribbean, habitats do not have international protected status but are under significant pressure due to the intensification of land use (particularly urbanization, tourism development and overgrazing) and climate change.

Caribbean Netherlands is part of the European Overseas Territories and concerns Bonaire, St. Eustatius and Saba. The European "Voluntary Scheme for Biodiversity and Ecosystem Services in Territories of European Overseas" (BEST) aims to strengthen nature and biodiversity conservation in these overseas territories and has designated important biodiversity areas for which ecosystem profiles have been developed (Figure 1). These areas are based on nationally designated marine and terrestrial parks, RAMSAR sites (RAMSAR, 2024) and Important Bird Areas (Birdlife, 2024).

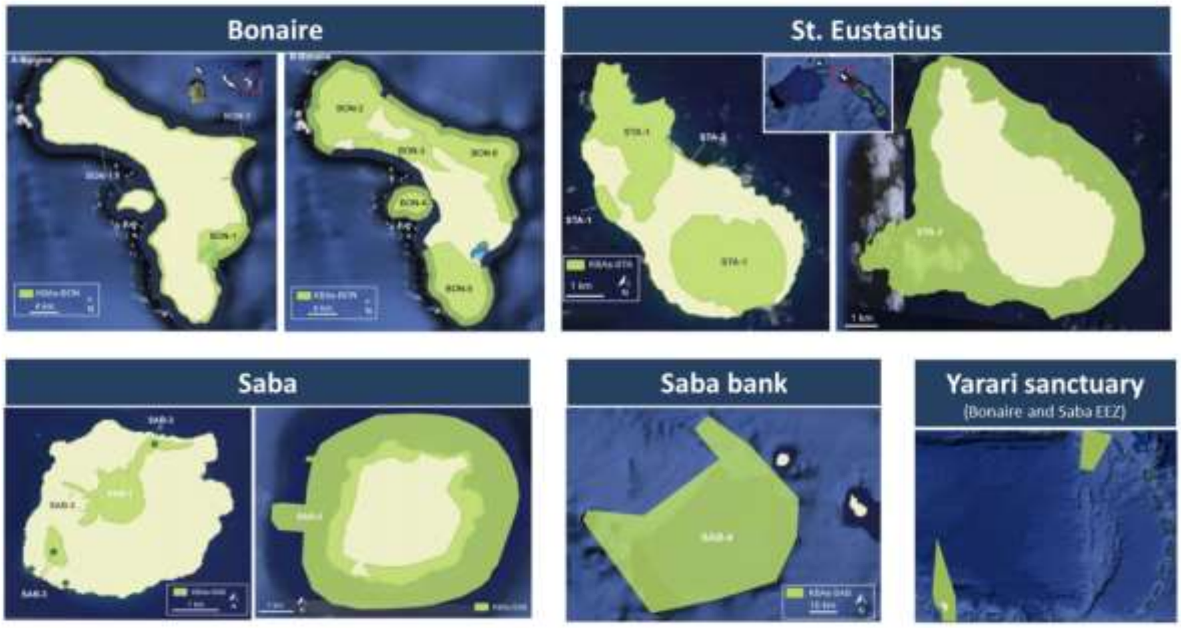


Figure 1. *Important biodiversity areas designated by BEST (Source: REP-CR, 2016).*

Within the BEST ecosystem profiles, the following habitats for the Caribbean Netherlands are distinguished (REP-CR, 2016) (from high to low elevation): cloud forests, montane forests, dry tropical forests, dry shrubland and grassland, caves, beaches, salt pans and saline lakes, mangrove forests, seagrass beds, seaweed beds, coral reefs, and deep sea. These habitats are not only important for the survival of the many species that depend on them, but also for humans due to the many ecosystem services they provide, such as recreation and tourism, coastal protection, water purification, food supply, etc. Freshwater is a habitat type of critical importance on semi-arid islands, but no reports have been made about it (sources, cave waters, permanent water pools, etc.) as very little is known about it.

Table 1. *Habitats of the Caribbean Netherlands (REP-CR, 2016; Verweij et al., 2015). "x": habitat present, "x": habitat rare and "-": habitat not present.*

Habitats	Impression	Bonaire	St.Eustatius	Saba
Elfin Forest – Elfin or "cloud" forests are rainforests characterized by high humidity and are often shrouded in mist. Tree trunks and branches are covered with thick layers of mosses, ferns, bromeliads, and orchids, among others. These forests typically grow on mountain slopes between 1,500 and 3,000 meters, but on St. Eustatius and Saba, they grow at much lower elevations. On Saba, they are distinguished by the tall-growing *Mountain Mahogany*.		-	(x)	x
Montane forest – These forests occur in warm, humid climates on the higher part of the slopes of mountains. Tree height varies as result of wind exposure. In wind protected gullies, three canopy layers can be found and trees are typically much taller and wider. Epiphytes are found in the areas exposed to humid air.		-	x	x
Lowland tropical rainforest - grows on flat lands at elevations generally below 1000m, is rare in the Caribbean and often consists of more than five canopy layers. Trees can reach heights of 40 meter, while emergent trees are covered in epiphytic ferns, orchids and bromeliads		-	x	-
Dry Tropical Forests – One of the most endangered tree-dominated habitats in the entire Caribbean. The trees and scrub shed their leaves during the dry season, allowing light to reach the ground, which results in dense ground vegetation. Here, fruit-eating bats, parrots, parakeets, crested caracaras, the Lesser Antillean iguana, and the Red-bellied Grass Snake find their food. On Bonaire, however, cacti and thorny scrub dominate due to grazing by free-roaming livestock		x	x	x
Dry shrubland and grassland - are fairly open areas with (thorny) shrubs and grasses adapted to grazing animals and nutrient-poor soils, such as steep, rocky and eroding slopes. Often there is a mosaic pattern of shrubs and patches of grass.		x	x	x

Habitats	Impression	Bonaire	St.Eustatius	Saba
Caves – Caves are primarily found in relatively soft limestone, which dissolves fairly easily under the influence of water. On Bonaire, they are home to unique life forms and serve as a crucial habitat for many species of bats and shrimp (freshwater caves and sea caves). Additionally, some of these caves contain rock paintings from the indigenous people. Saba and St. Eustatius have far fewer caves of different geological origins.		x	(x)	(x)
Beaches – Beaches have little or no vegetation and are primarily composed of coral rubble or sand and volcanic sand in the Caribbean Netherlands. They form an important part of the habitat for land crabs, hermit crabs, shrimp, and are nesting sites for sea turtles. Saba has very little beach habitat.		x	x	(x)
Salt Pans and Saline Lakes (Saliñas) – A saline lake is an inland body of water with no open connection to the sea. The term "salt pan" refers to an artificial lake created for salt extraction. These areas are important for (migratory) birds (including the Caribbean Flamingo) and crabs.		x	-	-
Mangrove Forests – These are dense, dark, mosquito-prone tropical coastal forests. Mangroves are a nursery for many reef fish and a breeding ground for (water) birds.		x	-	-
Seagrass and Seaweed Beds – These beds are often found next to coral reefs and provide shelter for juvenile coral fish and habitat for the Queen Conch. They occur in shallow, calm waters and serve as a food source for the Green Sea Turtle.		x	x	x
Coral Reefs – Coral reefs occur in tropical shallow, clear seas (up to about 60m) and are built by coral polyps.The sea provides a constant supply of food in nutrient-poor waters, supporting a rich and varied ecosystem of soft and hard corals, sponges, turtles, parrotfish, surgeonfish, sea bass, sharks, rays, and more.		x	x	x
Open Ocean and Deep Sea – The open ocean and deep sea are the largest habitats on Earth. Very little is known about life in the deep sea. No sunlight reaches these depths, and the water pressure is extremely high. Yet, life exists there, including anemones, worms, sea cucumbers, crabs, shrimp, and brittle stars.		x	x	x

Surface areas

There is little known about the current (2024) boundaries and quality of the habitats, while it is known that pressures from economic growth, population growth and urban expansion are increasing. Many satellite image interpretations and habitat surveys are outdated and consequently the coverage numbers do not necessarily reflect the current situation. Figures 2, 3, and 4 provide the most recent spatial interpretation from the various sources. Table 2 provides an overview of the areas of all habitats in the Caribbean Netherlands.

Table 2. *Total habitat area at the Dutch Caribbean islands. Data acquisition date is included between brackets.*

Habitat	**Bonaire [ha]**	**St. Eustatius [ha]**	**Saba [ha]**	**Saba Bank [ha]**	**Totaal [ha]**
Elfin forest	-	2-4 (2020)	6 (combi 2010, 1999, 2020)	-	~9
Montane forest	-	157 (combi 1999, 2011)	703 (combi 2010, 1999)	-	105
Lowland tropical rainforest	-	34 (combi 1999, 2011)	-	-	34
Dry tropical forest	6.820 (combi 1998-1999, 2014-2016)	806 (combi 1999, 2011)	14 (combi 2010, 1999)	-	17.573
Dry shrubland and grassland	16.941 (combi 1998-1999, 2014-2016)	718 (combi 1999, 2011)	348 (combi 2010, 1999)	-	4447
Caves	>3* (2017)	< 1 (2017)	< 1 (2017)	-	>3
Beaches	9 (1998-1999)	4 (combi 1999, 2011)	<< 1 (2017)	-	13
Salt pans and salt lakes (saliñas)	3.279 (2014)	-	-	-	3.279
Mangroves	236 (2014)	-	-	-	236
Seagrass beds	215 (2022)	124 (2012-2013)	20 (2013)	-	359
Macroalgal fields	475 (2017)	578 (2012-2013)	42 (2013)	5.529 (2012-2016)	6.596
Coral reef	866 (2013, expert estimates for the east coast)	1.027 (2012-2013)	308 (2013)	14.200 (2012-2016)	16.401
Open sea and deep sea EEZ	1.297.000 (2010)	215.000 (2010)	728.400 (2010)	-	~2.240.400 ‡

*Caves Bonaire: this is a rough estimate.
‡ Source: Soons (2011).

Caves are mainly found in the limestone of Bonaire. The area is roughly estimated at over 3 hectares. On the entirely volcanic islands Saba and St. Eustatius caves are rare. Cloud and rainforest are found only on St. Eustatius and Saba. These habitats are among the rarest and are likely the most vulnerable to climate change. All habitats show some form of degradation.

Beaches are mainly found on Bonaire (98% of the total in the Caribbean Netherlands) and to a much lesser extent on St. Eustatius. On the steep island of Saba, there are hardly any beaches. Salt pans and saline lakes (saliñas) are only found on Bonaire, especially in the south, where they are used for sea salt production. Mangrove forests are also only found on Bonaire. Mangrove trees can be found in various locations, but true mangrove forest is only present in Lac Bay. The maps do not differentiate between seagrass beds and algae beds. Nearly 60% of the seagrass beds in the Caribbean Netherlands are found on Bonaire, and just over 20% on St. Eustatius. No seagrass grows on the Saba Bank, but large algae beds are present there. The extent of the algae beds on the Saba Bank is unknown, but it is likely larger than the area covered by coral reefs. An estimated 98% of the algae beds in the Caribbean Netherlands are located on the Saba Bank. Around 92% of the coral reef area of the Caribbean Netherlands can also be found here. The famous fringing reefs of Bonaire are still about 30 times smaller in size than the reefs of the Saba Bank. With over 2 million hectares, the open ocean and deep sea form the largest habitat in the Caribbean Netherlands.

Figure 2 shows the spatial distribution of the different habitats on Bonaire. Dry tropical forests are derived from the landscape-ecological vegetation map of Freitas et al. (2005). "Undulating hills," "Escarpments" (E), "Higher terraces" (TH), "Middle terraces" (TM), and "Lower terraces" (TL) have been interpreted as dry tropical forests, except for TL1, TL2, TL3, TL4, TL5, TM1, TM2, and TM5, which are classified as degraded tropical forests. Caves were provided by F. Simal (pers. comm. 2017). The estimates for mangroves originate from Mücher and Verweij (2020). Beaches are again based on Freitas et al. (2005), using the B1, B2, and B3 classes. Salt pans and saline lakes are based on the S2 and W classes of Freitas et al. (2005), with additions from E. Dijkman (pers. comm. 2013). Coral reefs on the west coast, south coast, and Klein Bonaire are sourced from van Duyl (1985) and Dijkman et al. (2012). The extent of the deep sea is based on the size of the Exclusive Economic Zones (EEZ) (Meesters et al., 2010), reduced by the area of coral reefs, seagrass beds, and shallower seabeds.

Figure 3 shows the spatial distribution of habitats on St. Eustatius. The terrestrial habitats are based on a combination of the fieldwork-based vegetation map (Freitas et al., 2012) and land use data derived from high-resolution satellite images (Smith et al., 2013). While land use patterns typically change slowly, for detailed vegetation assessments (see Van Proosdij et al., chapter 5) the most recent aerial images from 2018 and 2024 were used. The vegetation class M1 (*Myrcia*) is represented as rainforest. 'Lowlands,' 'hills,' and 'mountains' are classified as dry tropical forests, except for the classes L2 and M9, which indicate degraded tropical forest. Invasive flora-dominated areas (e.g., Coralita) are also included as degraded. The presence and location of caves and beaches were estimated based on expert advice from A. Debrot (2017). Marine habitats are based on Debrot et al. (2014). The extent of the deep sea is based on the size of the EEZ (Meesters et al., 2010), reduced by the area of coral reefs, seagrass beds, and shallower seabeds.

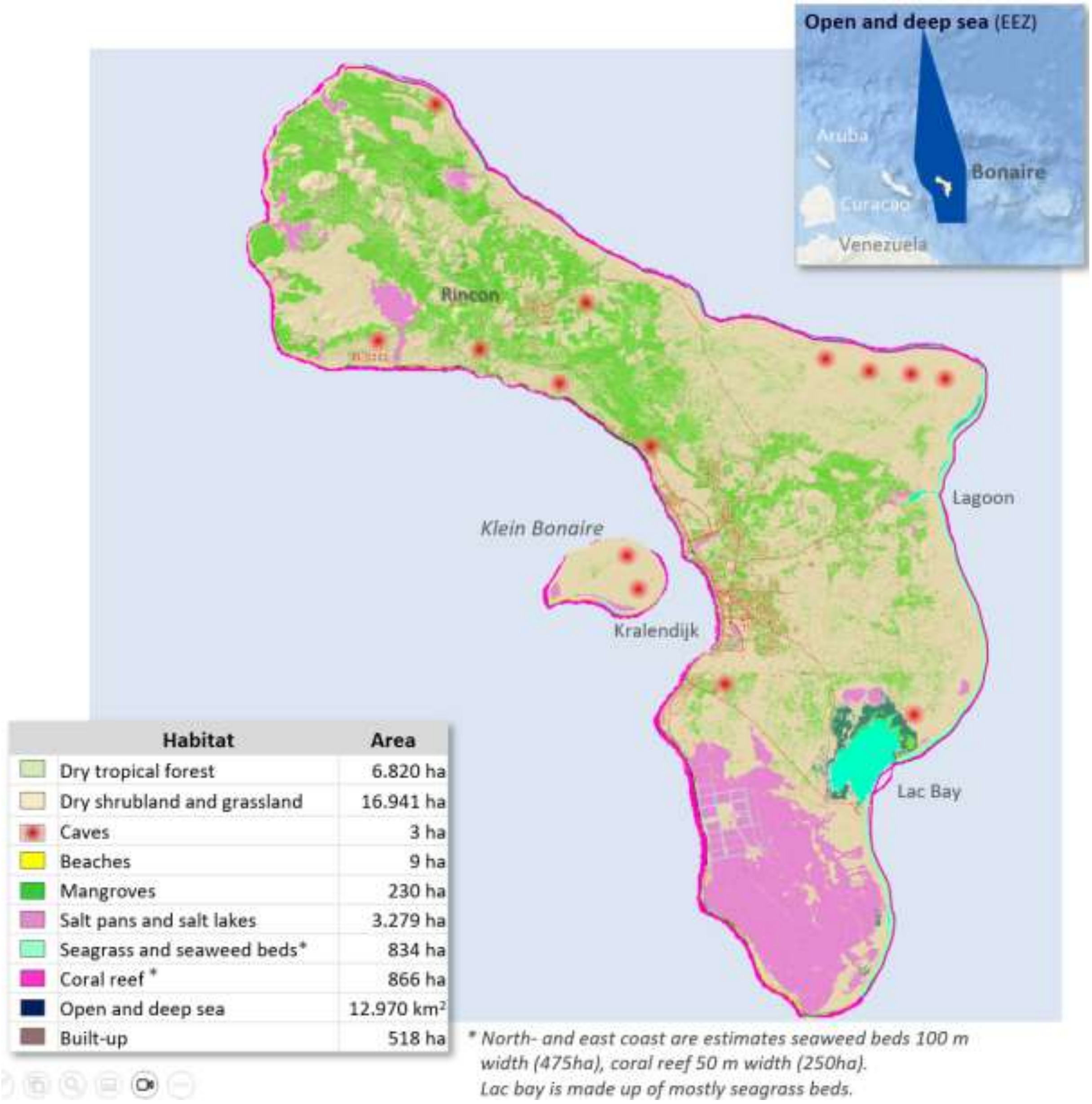


Figure 2. *Habitats of Bonaire.*

Saba and Saba Bank

Figure 4 shows the spatial distribution of habitats on Saba. The terrestrial habitats are based on a combination of the fieldwork-based vegetation map (Freitas et al., 2016) and more recent land use data derived from high-resolution satellite images (Smith et al., 2013). The vegetation class M1 (*Heliconia*) represents cloud forest, while class M2 (*Philodendron Marcgravia*) is classified as rainforest. The remaining vegetation classes represent dry tropical rainforest. Classes M7 and M8 represent degraded dry tropical rainforest. Invasive flora (e.g., Coralita) is interpreted as degraded. The presence and location of caves and beaches were estimated by A. Debrot (2017). Marine habitats are based on Kuramee and van Rouendal (2013). The extent of the deep sea is based on the size of the EEZ (Meesters et al., 2010), reduced by the area of coral reefs, seagrass beds, and shallower seabeds. The habitats of the Saba Bank have been mapped by Meesters et al. (2024).

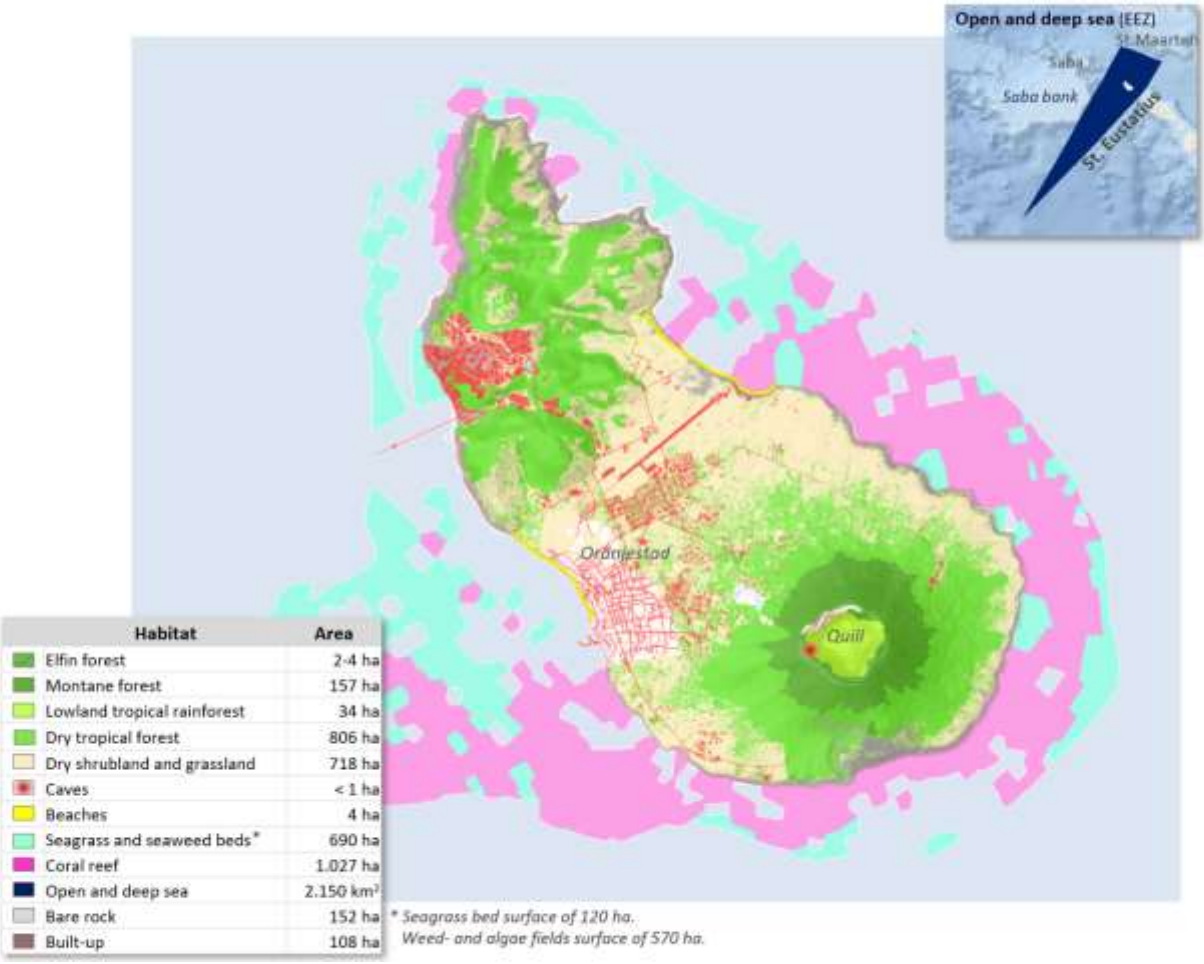


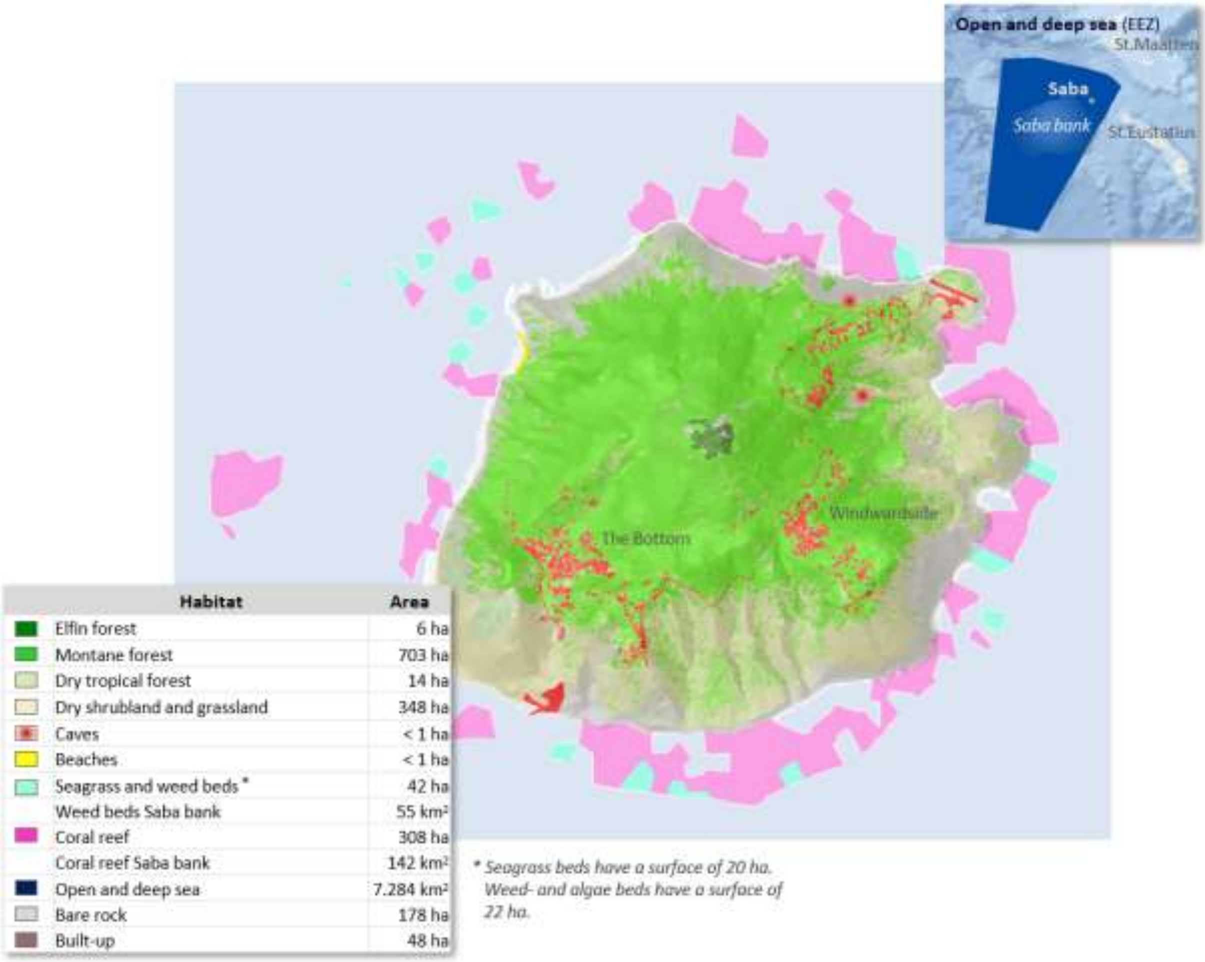
Figure 3. *Habitats of St. Eustatius.*

It is known that in the colonial past, agriculture occurred almost everywhere on Saba where the land was not excessively steep, amounting to about 200 ha of land (de Palm, 1985), such that very little primary vegetation left, except on the steepest slopes. Areas not converted to fields provided forage and wood for various uses. However, by 1985 only 65 ha were still in use as agricultural fields (de Palm, 1985). Over the last thirty years, agriculture has largely disappeared, and vegetation is now recovering throughout the island.

Comparison to the 2018 State of Nature Report

No major changes in habitat “availability” have taken place since the 2018 report but habitat “quality” (for instance in the relative density of seagrass in Lac Bay) has changed for several habitats (as discussed individually in the following chapters).

Figure 4. Habitats Saba and Saba Bank.



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2 Conservation State of the Terrestrial Vegetations Washington-Slagbaai and Klein Bonaire

Van Proosdij, A. S. J., Janssen, J. A. M., Houtepen, E. A. T., Eckrich, C., Bertuol, P., Francisca, R. and Beukenboom, E. 2025. From: State of Nature Report for the Caribbean Netherlands, 2024. WUR report C001/25.

Vegetation and Habitat Diversity

On Bonaire the changes in terrestrial habitats have been studied by a resurvey of vegetation plots made in the 1990's, which were used for the Landscape Ecological Vegetation Map (De Freitas et al., 2005). For this, keep in mind that "vegetation" (as a characteristic composition of plants associated with certain meteorological and geological parameters) forms a key habitat parameter for all animals and most individual plant species. Therefore, in the field of ecology, terrestrial "habitat type" is often synonymized with "vegetation type". On Bonaire, plots (325) were made for that map in 1999, and so far, (end of 2024) about half of the plots have been resurveyed. The other half will be repeated in 2025. Therefore, the here presented results are preliminary, until a complete resurvey of Bonaire has been completed. For this chapter we selected two parts of the island for which the resurvey has already been completed: (i) the Washington-Slagbaai NP (National Park), and (ii) the island of Klein Bonaire. Both are interesting sites, from a nature conservation point of view, considering that the main pressure for the terrestrial biodiversity is over-grazing by free-roaming animals, specifically goats and donkeys. The island of Klein Bonaire is free of goats (and donkeys) for more than 60 years. It serves as an example of how vegetation restoration takes place in a coastal limestone area. In the Washington-Slagbaai NP the last ten years have been used to create goat-free areas in the former Slagbaai plantation (Fig. 1). Some of these areas show the first signs of vegetation restoration, while large parts of the park are still over-grazed by both goats and donkeys.

The main diversity in habitats on the island of Bonaire is between azonal and zonal vegetation types. Azonal types refer to those in which the species composition is mainly determined by one or more extreme conditions unrelated to climate (like mangroves) while zonal types are more uniform across locations within a climatic zone. Consequently, azonal types typically have greater geographic similarity in species composition compared than zonal vegetation types. Azonal types for Bonaire are for instance mangroves, saliñas, dunes, fresh-water wetlands, and temporary riverbed (rooi) vegetation, while zonal vegetations of limestone and volcanic bedrock (Washikemba formation) are especially determined by climatic conditions depending on rainfall and humidity as related to altitude. In the two assessed areas, Klein Bonaire and Washington-Slagbaai NP, six (sub)habitats are found. Klein Bonaire consists largely of a low limestone terrace, covered by a low, sparse shrubland (habitat 1, Limestone shrubland). Smaller sections along the shores of the island are covered by azonal saliña vegetation, with some low mangroves (habitat 2) and by dune and beach vegetation (habitat 3). The main habitat in Washington-Slagbaai NP is dry shrubland and forest on volcanic soil (habitat 4, Volcanic woodland; part of the broader habitat Dry tropical forest). In contrast, the higher, more wind-exposed slopes are sparsely vegetated with an open grassland community on screes (habitat 5, Volcanic grassland; part of the broader habitat Dry shrubland and grassland).

The north coast and parts of the west coast are sparsely vegetated limestone soils with azonal coastal vegetation (habitat 6), while smaller sections of the coast consist of saliña (habitat 2) and dune vegetation (habitat 3). Marginal parts of the areas are made up of other azonal habitats, like mangroves, rooi vegetation and freshwater bodies. The three main azonal habitats are less well represented in the data for the two studied areas. For the three zonal habitats, enough data were available to analyse changes in species composition and structure. These are described in the next section.

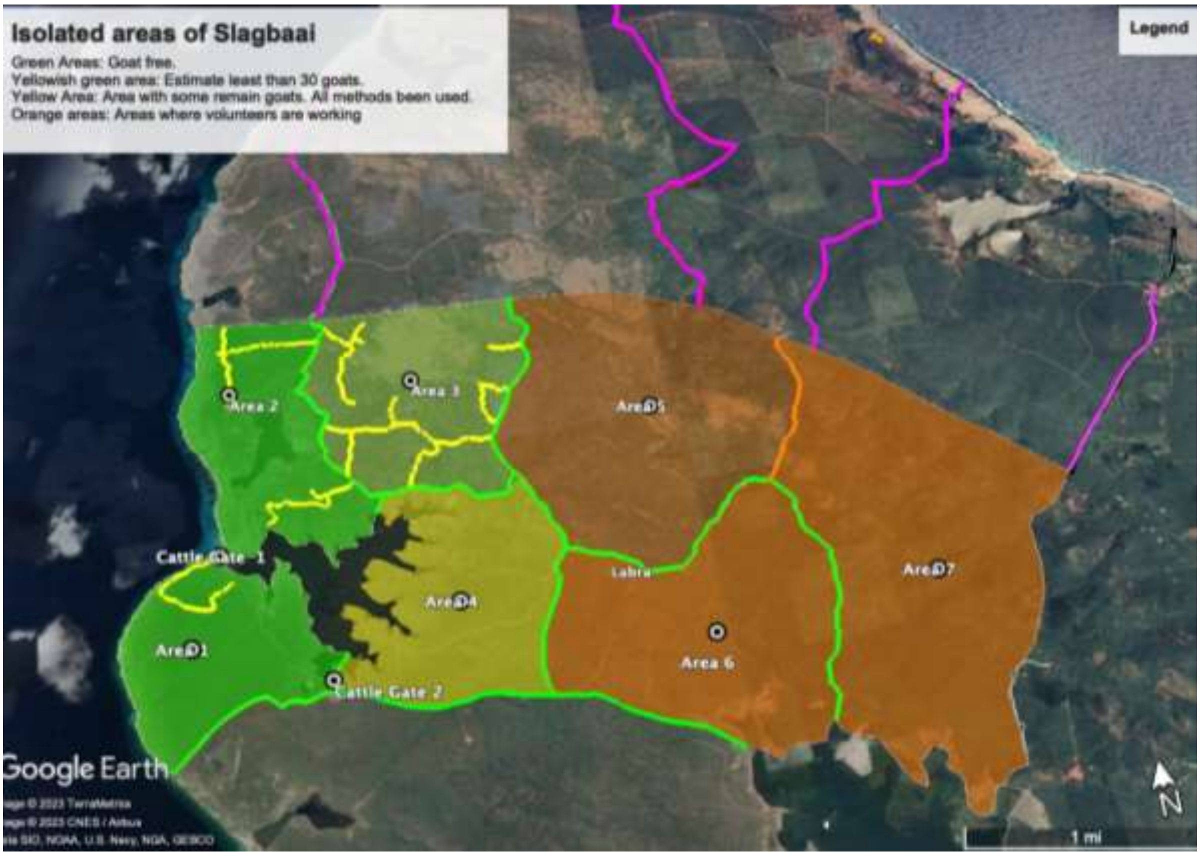

Figure 1. *Recently fenced areas in the Washington-Slagbaai National Park, of which Area 1 and 2 are free of goats and Area 3 and 4 the number of goats has been reduced substantially.*

Characteristics of the Four Habitat ("Vegetation") types

Limestone shrubland (Klein Bonaire): Large areas of Klein Bonaire are covered by low, shrubby vegetation, growing on karstic soils. Here, the vegetation is more sheltered from wind than on the north coast of Bonaire, so both height and cover of the vegetation on this sheltered limestone terrace are higher. This difference to the more open 'limestone pioneer vegetation' is also the result of the island Klein Bonaire being cleared of goats and donkeys several decades ago. In large parts of the island a limestone shrubland of about one meter height is found. The most common species of the shrubland are *Lantana involucrata, Corchorus hirsutus, Capraria biflora, Condalia henriquezii, Cordia curassavica* and *Jatropha gossypifolia*. A broad range of low herbs, sedges and grasses are found, including *Euphorbia thymifolia, Cyperus fuligineus, Sporobolous pyramidatus, Eragrostis urbaniana* and *Fimbristylis cymosa*. In the center of Klein Bonaire and in a narrow strip along the east coast, this type has already developed into a several meters high woodland where small trees are present, e.g. *Sideroxylon obovatum, Haematoxylum brasiletto* and *Zanthoxylum flavum*. In a broader sense this local habitat is part of the (natural) habitat Dry shrubland and grassland, but here it is restricted to a variety on limestone. Besides, it is unclear how much of this habitat should be considered as degraded Dry tropical forest, and how much represents a climax stage in places where no forest will grow due to salt-spray and strong wind.

Volcanic grassland (Washington-Slagbaai National Park): This is a relatively open grassland or pioneer habitat on volcanic scree, which is mainly found on relatively steep, exposed and eroding slopes (Fig. 2). The characteristic species include several grasses, sedges and small herbs, e.g. *Aristida adscensionis*, *Euphorbia thymifolia*, *Lithophila muscoides*, *Cyperus amabilis*, *Anthephora hermaphrodita* and *Chloris barbata*. In some places the low shrub *Croton conduplicatus* (in Van Proosdij, 2012 as *Croton flavens*) has high cover, indicating succession towards a shrubland on volcanic soil. On the higher slopes of the Brandaris hill (the highest point of Bonaire), the fern species *Doryopteris concolor* occurs in this type. *Opuntia caracassana* is shared with the Volcanic woodland type. In the Volcanic grassland type a few (near-) endemic species are present, of which the distribution outside the ABC islands is largely unknown, e.g. *Paspalum bonairense, P. curassavium* and *Opuntia curassavica*. In a broader sense this local habitat is part of the (natural) habitat Dry shrubland and grassland, but it also includes areas that are severely degraded and with undisturbed succession may develop into Dry tropical forest.

Volcanic woodland (Washington-Slagbaai National Park): This is the most common and widespread habitat in the Washington-Slagbaai National Park. It is widespread on volcanic soils of the Washikemba formation, on all middle and lower slopes and in dry rooi beds. Dominant high shrub species are *Prosopis juliflora*, *Casearia tremula*, *Phyllanthus botryanthus*, *Randia aculeata*, *Quadrella odoratissima* and the columnar cacti *Stenocereus griseus* and *Cereus repandu*s. *Opuntia caracassana* is shared with the Volcanic grassland type. Most of these species are protected against grazing by either not being palatable or having spines or thorns. Just recently, the first areas of the park have been cleared of goats and donkeys (areas 1 and 2, Fig. 1), and here immediately regeneration of juvenile trees can be seen. However, none of the old plots are situated in area 1 and 2. Several however, are situated in the even more recently fenced areas 3 and 4 from which many but not all goats have been removed. In a broader sense this local habitat is part of the Dry tropical forest habitat type, but it includes both well-developed and degraded stages. In the Washington-Slagbaai National Park the presence of Dry tropical forest type is largely restricted to the here described variety on volcanic soils.

Figure 2. *Wind-exposed volcanic slopes with grassland (background) and low shrubland (foreground) on the high slopes of the Brandaris in the Washington-Slagbaai National Park. In the lower, more-sheltered depressions in the back volcanic woodland is seen. This area has been recently fenced and since then, the number of goats has been strongly reduced. Photo John Janssen.*

Relative Importance Within the Caribbean

Limestone shrubland on Bonaire occurs on Klein Bonaire as well as in several other areas on Bonaire, e.g. Lima, Karpata, Bolivia and is also present on Aruba and Curaçao. This type resembles shrubland and grassland on limestone on St. Eustatius and Sint Maarten, although large differences are observed in species composition. Overall, this type is widespread in the wider region of the Caribbean and coast of Central and northern South America, however large regional differences exist and are not yet well understood. The type present on the ABC islands seems to be restricted to the dry tropics of La Guajira, Paraguaná Peninsula and the islands near the South American north coast. Based on the limited distribution of this subtype, the international importance of the Limestone shrubland on Bonaire is medium (see Table 1).

Volcanic grassland is common in many areas of Bonaire, as well as Aruba and Curaçao. This type is widespread in the wider region of the Caribbean, Central and South America. Regional differences exist but are not yet well studied. The distribution of endemic grass species remains largely unknown, but their presence indicates a higher international importance of this type. With respect to these knowledge gaps, the international importance of this type cannot be identified yet.

Volcanic woodland is widespread in all six Dutch Caribbean islands as well as in the wider Caribbean region, Central and northern South America. The diversity in dry forest communities within the wider Caribbean is not yet well described. However, the plant communities in the leeward islands Aruba, Bonaire and Curaçao differ strongly from those in most of the volcanic islands of the Caribbean arcs. The main reason is the different climate, which is much drier in the area situated in the rain shadow of the Venezuelan coastal mountain range (Cordillera de la Costa). The dry forests of the leeward islands, as well as the derived shrublands, resemble the forests on the most northern part of the South American mainland, especially those in the peninsula La Guajira, Colombia (Rieger, 1976; Rangel, 2012), on the Paraguaná Peninsula, Venezuela (Matteuci, 1987) and in the adjacent mainland. Also, Isla de Margarita and other Venezuelan islands are situated in this dry tropical region and have similar forest and shrubland communities. Within this region, there is a clear difference in species composition and structure between the woodlands on limestone and those on volcanic soils (Stoffers, 1956; De Freitas et al., 2005). Important is that on a high level of classification, the Dry tropical forest is amongst the most threatened forest ecosystems of the Neotropics (Ferrer-Paris et al., 2019). Consequently, the international importance of the Volcanic woodland on Bonaire is high.

Table 1. *Relative importance of the habitat types on Bonaire within the wider region. Distribution and surface areas on Bonaire are derived from the map of De Freitas et al. (2005).*

	Area (ha)	Worldwide range	International importance
Limestone shrubland	±525 ha (Klein Bonaire)	Caribbean, Central and northern South America, but this subtype is limited to La Guajira, Paraguaná Peninsula and the islands near the South American north coast	Medium(?)
Volcanic grassland	±330 ha (Washington-Slagbaai NP)	Caribbean, Central and northern South America, but this subtype may be limited to the dry tropical region of the north-coast of South America.	Unknown
Volcanic woodland	±2765 ha (Washington-Slagbaai NP)	Caribbean, Central America and northern South America, but this subtype is limited to La Guajira, Paraguaná Peninsula and the islands near the South American north coast	High

Developments and Trends

Pressures and threats
The continuous and severe over-grazing of the understory by goats and donkeys has led to a vegetation where the herb and shrub layer is largely reduced, and regeneration of woody species has stopped. In vast areas of the Washington-Slagbaai NP saplings of tree species are rare. To restore nature values, some areas have been cleared from free-roaming goats in the past few years and the remaining areas are to be freed of goats in the next years as well. The NEPP for the Caribbean Netherlands assigns a high priority to culling uncontrolled roaming livestock (Min. LNV et al., 2020). All free-roaming animals have been taken from Klein Bonaire some 60 years ago and since then the vegetation has recovered immensely (see below). Urbanization, leading to destruction of natural vegetation is an important pressure on most if not all Caribbean islands. However, with respect to Klein Bonaire and the Washington-Slagbaai NP, this is not the case as both areas are a national park.

Climate change may be a major threat to both areas. Sea level rise is expected to severely impact Klein Bonaire, either directly through incidental or permanent flooding (Van Oosterhout et al. 2023), but also by reducing the freshwater table in the soil. The resulting net effect on species composition of both plant and animals is yet unknown. The Washington-Slagbaai NP may face longer periods of drought and more intense rainfall on other moments, which is expected to have a negative impact on the present Volcanic woodland type and to a lesser extent on the Volcanic grassland.

Finally, several invasive species are present on Bonaire, of which several are found on Klein Bonaire and in the Washington-Slagbaai NP. On Klein Bonaire, *Scaevola taccada* has become established on sand dunes and coastal shingle walls, where it outcompetes other species (Fig. 3) while the rubber vine, *Cryptostegia grandiflora*, is also found in many places on the island (Debrot, 1997).

Trends
The trends in structure and species richness and composition have been assessed by repeating in 2024 a vegetation survey at sample plots that were recorded before in 1999 (for the landscape ecological vegetation map by De Freitas et al., 2005). All plot data are stored in the CACTUS database (Janssen et al., 2023). The following trends in structure, functioning and species composition have been analysed from statistical comparison of the two data sets (Van Proosdij et al., in prep). So far, about 50% of the original 325 plots have been resurveyed, and therefore only a preliminary assessment is possible. For the areas Klein Bonaire and Washington-Slagbaai NP all old plots have been resurveyed, and the trends in those areas are presented here. As the three habitat types discussed here are also present in other areas of Bonaire for which the sample plots were not yet repeated, we here limit the discussion to trends in structure and species composition for Klein Bonaire and the Washington-Slagbaai NP.

Trends in Washington-Slagbaai NP
The **Volcanic grassland**, on the exposed and open slopes with scree did not change significantly in cover. An exception to this are the goat-free areas 1 and 2, where a resurgence of grasses and increase in tree seedling density and diversity is observed. The total species diversity increased, which was largely attributed to species in the herb layer. Among the 'winners' (species with increased occurrence) in this habitat are the grasses and sedges *Cyperus amabilis*, *Chloris barbata* and *Cyperus nanus*, the fern *Doryopteris concolor*, the cactus *Melocactus macracanthos*, the herbs *Bastardia viscosa* and *Spermacoce confusa*, the shrub *Croton conduplicatus* and juveniles of the tree species *Quadrella odoratissima* and *Libidibia coriaria*. A similar trend was found when comparing fenced and still-grazed exposed slopes. However, the number of plots was too low to apply robust statistical tests and the here discussed trends for this type are mainly based on qualitative analysis. Several explanations for these trends are possible. First, the development into more species-rich and better developed vegetation may be the result of fencing, even though this restoration measure was only carried out recently. Secondly, changes may be accounted for by seasonal effects. The rainy seasons of recent years, and especially those of 2022/2023

and 2023/2024, were extremely wet, which may have caused a higher vegetation cover. As a result of the more abundant crop, the relative impact on the vegetation of grazing by goats and donkeys may have been lower (see also next type). A third explanation is based on the abiotic conditions of the grasslands.

Figure 3. *Scaevola taccada forming dense stands and outcompeting other species in dune vegetation on Klein Bonaire (Photo John Janssen).*

The Volcanic grassland type represents the most exposed and highest slopes of the park. Here the goats are easily seen, and the hunting may have caused a change in behaviour, with goats preferring the more sheltered woodlands rather than the exposed grasslands. For that reason, the open slopes outside (but close to) the fenced areas, may have become less intensively grazed as well. The difference between the currently grazed and non-grazed sites indicates that fencing already had some impact. On the other hand, the positive changes in some of the still-grazed plots indicates the overall effect from the higher precipitation. Therefore, it is likely that a combination of both relatively wet years and the establishment of exclosures has resulted in the positive trend in this habitat.

The trend in the **Volcanic woodland** differs for different parts of the park. The number of plots in fenced areas that are free or largely free of goats is limited. However, by including additional, new plots made in 2024 in fenced, (nearly) ungrazed areas and comparing these with plots in grazed areas, differences in vegetation structure and species richness could be quantified. Overall, an increase in vegetation cover was found, which could be contributed largely to a significant higher cover of the shrub layer. Also, the cover of the tree and herb layer increased. The species diversity of this habitat type increased for both grazed and ungrazed sites, when compared to the 1999 data, but the magnitude of change is different. Current total species richness on ungrazed sites is more than 60% higher compared to 1999, with woody species richness more than 80% higher and herb species richness more than 120% higher. On grazed sites, total species richness and woody species richness did not differ significantly from the situation in 1999, but herb species richness was more than 70% higher compared to 1999. Apparently, herb species have increased in both grazed and ungrazed sites, whereas woody species only increased in ungrazed sites. Among the 'winners' (species with increased occurrence) in this habitat are the herbs *Nama jamaicensis*, *Elytraria imbricata*, *Bastardia viscosa* and *Rivina humilis*, the trees *Phyllanthus botryanthus*, *Guapira pacurero* and *Quadrella odoratissima a*nd the vine *Passiflora suberosa*. Two species showed declining numbers: the sedge *Bulbostylis curassavica* and the shrub *Lantana camara*. The increase of herb species richness in both grazed and ungrazed sites can partly be explained by the extremely wet rainy seasons of recent years, especially 2022/2023 and 2023/2024 (see also the previous type). However, the much stronger increase of herb diversity and increase of woody species diversity in (largely) ungrazed sites can be regarded as a success of the removal of free-roaming goats. In contrast to the positive trends for the (largely) ungrazed areas of Slagbaai, other parts of the national park show different trends. At the north coast, between Malmok and the park entrance, this habitat is represented by relatively species-poor *Vachellia tortuosa* and *Prosopis juliflora* dominated scrub, sometimes with high cover of *Opuntia caracassana*. The Washington plantation has a higher density of goats than Slagbaai (goat census, Stinapa, pers. comm). In the areas nearest to the coast, a regression of the vegetation towards more open stands with grasses took place. The higher volcanic shrubland of the adjacent hills, slightly more inland, showed few changes. Many of these are relatively species-poor stands and some of them became even more species-poor. Here, some species showed positive trends, but others (like *Melochia tomentosa*, *Cereus repandu*s, *Heliotropium angiospermum*) had negative trends. Overall, the first signs of vegetation recovery and habitat restoration are noticed in the few areas that have been (largely) cleared of free-roaming goats. Areas that are still under severe, long-term overgrazing showed even further loss of species and degradation of the habitat. These areas, however, may be expected to recover too, once they are freed of free-roaming goats. The presence of a local species pool that is much larger than the one on Klein Bonaire (see below) and the presence of many sheltered sites offer opportunities for rapid recolonisation and increase for many species.

Trends on Klein Bonaire

Klein Bonaire has been intensely grazed by free-roaming goats for more than a century. In the 1960s (almost) all goats were removed from the islet, creating opportunities for the vegetation to recover. The last few goats were removed in the 1980s (E. Newton, pers. comm.). Since 1999 the **Limestone shrubland** has substantially increased in height (Fig. 4). Overall cover has not changed significantly, but changes in cover are strongly impacted by seasonal and annual variations. Species richness however has increased with 9%, although due to the low number of plots this is not significant. The increase in height and species richness illustrate the recovery of the ecosystem on Klein Bonaire and hence the success of goat removal from Klein Bonaire. The data quantify the low speed of vegetation recovery on the harsh, coastal limestone plateaus (karst soil) and the slow, but steady succession from sparsely vegetated limestone vegetation towards shrubland and patches of forest (see also Debrot, 1997). Seven species present in plots in 1999 were not observed in the plots by 2024, although all of these were observed outside the chosen plots. In contrast, fourteen species were observed in plots by 2024 that were not recorded in plots in 1999 and of which it is unknown if these were present on other parts of Klein Bonaire in 1999. Overall, more species increased than decreased in presence as well as in cover. Especially, the following species substantially increased: *Cordia curassavica*, *Fimbristylis cymosa*, *Rhynchosia minima*, *Anthephora hermaprodita* and *Stemodia maritima*.

The two other habitat types on Klein Bonaire, the Saliña vegetation with some low mangroves (habitat 2) and the Dune and beach vegetation (habitat 3) also show a recovery. The Saliña vegetation did not change much in height, cover and species richness. The Dune and beach type shows an increase in height and an increase of species richness, but these are not significant due to the very limited number of plots. The speed of vegetation recovery and recolonization by species is hampered by the harsh conditions and limited local species pool on Klein Bonaire. Natural colonization of Klein Bonaire plays an important role. As seeds of several species are dispersed by wind or birds from Bonaire. The establishment of some of these help develop the ecosystem by stabilising sand, providing shelter against strong winds and providing food for insects and birds.

Figure 4. *Changes in vegetation at Tanki Kalbas on Klein Bonaire between approx. 1956 (left) and 2024 (right). Note the presence of Typha in the water and the many, large columnar cacti in 2024, both absent in the 1950s.*

For example, *Paspalum vaginatum* was newly recorded for Klein Bonaire (outside plots) and this species plays an important role in stabilising sand on karst plateaus and in low dunes. Since 2006 a reforestation project is being carried out on Klein Bonaire (Debrot, 2013). The above results are based on plots where no planting or seeding of species has taken place. The reforestation is concentrated on the northern part of Klein Bonaire. In this area, several species were planted that were by then not yet present on Klein Bonaire. They were either endangered, like *Sabal lougheediana*, or bear fruits that serves as food for birds, like *Bourreria succulenta, Jacquinia arborea* and *Metopium brownei* (Debrot, 2013). It is expected that these (re-)introduced species will naturally disperse to other parts of Klein Bonaire and boost the recovery of the Limestone shrubland and other vegetations on Klein Bonaire. Both natural recolonization and reforestation through planting and seeding play an important role in ecosystem restoration. A full inventory of Klein Bonaire can render data to quantify the relative contributions to habitat restoration of each factor.

Two exotic species play an important role on Klein Bonaire. *Scaevola taccada* forms large stands in the Dune and beach vegetation (Fig. 3). Although it stabilises dune sand and beaches, it outcompetes other plant species including the native *Scaevola plumieri* that is present on Sorobon but not yet (or no longer?) on Klein Bonaire. The second exotic species is *Tabebuia heterophylla*, a tree native to the Caribbean that is widely planted as ornamental tree and rapidly spreads across Bonaire. This species grows on coastal limestone biotopes similar to the environment on Klein Bonaire. It’s natural distribution in the Caribbean region as well as its potential invasiveness on Bonaire (and Aruba and Curaçao) are insufficiently understood and this knowledge gap hampers effective management measures for Klein Bonaire. Finally, the presence of small patches of *Cryptostegia grandiflora*, an extremely invasive vine, need to be addressed before it gains major foothold (Debrot, 1997).

Assessment of Conservation State

For Bonaire, no complete assessment of Conservation State of terrestrial habitat types has been carried out by us yet, as only half of the historical data points have been repeated so far. However, it was possible to analyze trends in structure and function ('habitat quality’), based on changes in species composition for the Washington-Slagbaai NP and for Klein Bonaire. For the three main habitat types Limestone shrubland, Volcanic grassland and Volcanic woodland sufficiently large data sets were available which enabled the analysis of robust results. The main conclusions are:

- On Klein Bonaire since the removal of goats (which began in the 1960s and was finalized in the early 1980s; E. Newton, pers. comm.; Debrot, 1997), a slow but steady succession from sparsely vegetated limestone vegetation towards shrubland and patches of forest took place. Vegetation height substantially increased in the Limestone shrubland and in the Dune and beach habitat type. Only for the Limestone shrubland sufficient data are available to analyze trends. Species richness in the Limestone shrubland habitat increased by 9% since 1999. Although species numbers remain low and the recovery proceeds slowly, this increase of overall species richness is in sharp contrast to the dramatic decline of species richness observed in continuously overgrazed habitat types on e.g. Saba and St. Eustatius. Both natural recolonization and reforestation through planting and seeding play an important role in ecosystem restoration. A full inventory of Klein Bonaire can render data to quantify the relative contributions to habitat restoration of each factor.

- In the Washington-Slagbaai NP both Volcanic grassland and Volcanic woodland types show an overall small increase of species richness. However, large contrasts exist between fenced areas that are (largely) free of goats and unfenced areas that still face long-lasting, intense overgrazing by free-roaming goats (and to a lesser extent, donkeys). Volcanic woodland in goat-free areas shows a strong increase of species richness, both in herb species as well as in woody species. Whereas an increase of herb species richness can partly be accounted for by several extremely wet rainy seasons (2022/2023 and 2023/2024), the relatively higher level of increase of herb species and the increase of woody species in the ungrazed areas is a clear sign of vegetation recovery and habitat restoration that follows the removal of free-roaming goats in certain areas of Slagbaai. In contrast, vegetation changes in other areas including the northern part of the park, especially the coastal areas, but also the northern low hills, indicate little recovery or even a decline in structure and species richness.

The developments in both Washington-Slagbaai NP and on Klein Bonaire provide strong evidence that the removal of free-roaming goats and donkeys leads to improvement of the vegetation structure and an increase of the species diversity. It will also lead to more stability of the soil, which is an important factor to prevent erosion during heavy rains. The development on limestone is much slower than on volcanic soils, but on both soil types the removal of grazing non-native mammals is a successful approach to habitat restoration. We strongly recommend further investment in this restoration measure, as removal of over-grazing provides the best option for creating resilience of the vulnerable island ecosystems against effects of climate change. Its positive impact goes far beyond the nature values itself, as it prevents erosion and through that decreases the pressures on corals, it improves the water storage capacity of the soil and it lowers surface temperatures by the development of higher and denser forests.

Table 2. *Summary overview of the status of the Limestone shrubland vegetation of Klein Bonaire, Caribbean Netherlands, in terms of different conservations aspects.*

Vegetation Klein Bonaire	2024
Distribution	Favourable
Area	Favourable
Quality	Unfavourable-inadequate
Future Prospects	Favourable
Overall Assessment of Conservation State	**Unfavourable-inadequate**

Comparison to the 2018 State of Nature Report

The 2018 report presented an overall assessment for the whole island of Bonaire. Overall, no major improvements can be reported except that for Klein Bonaire definite improvements in the CS of the vegetation are clear and that early improvements for the WSNP, following removal of goats from certain sections of the park appear evident. Only for Klein Bonaire can we give a meaningful assessment of CS as there the improvements are long-term and sustainable, albeit gradual (Table 2). For the Washington-Slagbaai NP any improvements have been too recent, incomplete and uncertain, given the proven vulnerability of goat removal programs.

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3 Conservation State of the Terrestrial Vegetations of Saba

Janssen, J. A. M. and van Proosdij, A. S. J. 2025. From: State of Nature Report for the Caribbean Netherlands, 2024. WUR report C001/25.

Vegetation and Habitat Diversity

The vegetation (c.q. "habitat types") of Saba is developed along a steep gradient from sea levels to the highest point of Mount Scenery. De Freitas et al. (2016) distinguished nine vegetation types in this gradient, which for the assessment of the Conservation State have been generalized into four broader, natural habitats, all with transitions to the next type. From high to low elevation these are: Elfin forest, Montane forest, Dry tropical forest and Dry shrubland and grassland. These four types will be described here, and their status and trends will be assessed. A trend analysis of plot data from a vegetation survey in 1999 (De Freitas et al., 2016) and a resurvey of these in 2020 (Janssen and Van Proosdij, in prep.) provided relevant insight in the status and trends of the habitat types on Saba. All plot data are stored in the CACTUS database (Janssen et al., 2023). However, although the vegetation in question is still in a recovery phase and quantifiable data is lacking, it is worth mentioning that these surveys predate the start of the goat control project on Saba. According to island authorities, since 2020, over 90% of roaming goats have been removed from the island. If roaming goat densities can indeed be maintained at such low levels, then within a few years we predict that tree seedling densities and diversity, and herbaceous cover should measurably improve.

Although it is the smallest island within the Dutch Caribbean, Saba has the highest number of vascular plant species (772 taxa, according to Axelrod, 2021). This high diversity is a result of the steep gradient in elevation, causing a broad range of micro-climates and micro-habitats with different expositions to sun and wind, and different precipitation, humidity and temperature. In addition to the environmental diversity, the relatively low human impact on the island is a subsidiary factor explaining the high species diversity. This is illustrated by the 1963 topographic map, where the slopes of Mount Scenery show patches of low vegetation (grassland or scrubs). Some of these are lands that are used for crops or grazing, but in the higher elevations such patches form a minor part of the slopes. The (relatively) low impact of land-use is strongly related to the environment: because of the steepness of many slopes, these are hard to use for agriculture or building. A possible contributing factor could be the complexity of land ownership on the island. Many parcels of land remain undeveloped because of ownership issues resulting from poor legal and notarial services in the past. De Freitas et al. (2016) refer specifically to the northern and north-western parts of the island as being less disturbed by man, except for roaming feral goats. Although there is a slow increase of built-up areas over the last decades (figure 1), urbanization and land-use historically have caused much lower pressures on the natural environment than in any other of the Dutch Caribbean islands.

 

Figure 1. *A view of the Bottom from the west slope of St. John's Hill in the 1950s (left; Stoffers 1956) and 2020 (right; photo: John Janssen) shows the increase of buildings within the villages. Note the increase of vegetation cover on Parish Hill and Great Hill on the left and Castle Hill on the right.*

Characteristics of the Four Habitat Types

Elfin forest on Saba is characterised by the tree known as the mountain mahogany (*Freziera undulata*) and the shrub *Miconia purpurea* (= *Charianthus purpureus*). This forest type is easily recognized by the relatively low height of the gnarled trees, the tangled branches, the crowns deformed by wind with dead twigs, and especially the huge amounts of mosses, liverworts and filmy ferns (*Hymenophyllaceae*) that cover stems and branches of the woody species, and together with ferns, orchids and other vascular plants create a very species-rich epiphytic flora. It is the most species habitat type, on average (per plot) slightly richer than the montane forest and dry forest. Within the Dutch Caribbean, many plant species are largely restricted to this habitat type, amongst others the rare epiphytes *Voyria aphylla*, *Utricularia alpina*, *Notopleura guadalupensis*, *Ornithidium reflexum*, *Peperomia hernandiifolia, Peperomia emarginella* and *Werauhia urbaniana* (= *Vriesea antillana*). The entire summit of Mount Scenery is covered by Elfin forest, but the species composition varies between the exposed rocky sites of the highest ridges (the typical site of the mountain mahogany) and the more sheltered, shallow depression of the crater floor. The most important ecological factor is the high air humidity, which is caused by the clouds that develop almost every day around the top of Mount Scenery. Because of the clouds, Elfin forest is also referred to as Cloud forest or Mist forest. Because of the relatively shrubby appearance of trees, the plant community is also referred to as Elfin woodland. The altitude of occurrence (between 750 and 850 m) is not the most relevant abiotic factor: if you compare similar elfin forests on other Caribbean islands, it is found mostly in higher altitudes, but always on the summit or on the highest ridges of the volcanic mountains (Beard, 1949). Also, on St. Eustatius some patches of Elfin forest are found, with low trees and shrubs of which the stems and tangling branches are completely covered by mosses and liverworts. These patches grow on Mazinga, the highest part of the crater rim of the Quill volcano, which is also often covered in clouds and therefore has a high air humidity. The species composition of the Elfin forest on St. Eustatius is less typical however: most of the species are found in other parts of the rim as well. However, in those places the typical structure of moss-covered stems and branches is lacking. The occurrence at lower altitudes on Saba and St. Eustatius, as compared to other Caribbean islands, is probably caused by the steepness of the mountains and proximity to the ocean, forcing moist air from sea to rise, which results in the formation of clouds.

Montane forest forms the second-highest belt on the steep slopes of Mount Scenery, directly below the Elfin forest. The habitat occurs roughly between 500 and 750 m altitude, with exact ranges depending on the slope exposition. The trees in general grow taller than in Elfin forest, and the structure in most places is more layered, with a distinct shrub layer. Especially in sites that are sheltered from the wind, such as in gullies on the north slope, trees grow tall and different tree layers can be distinguished. The epiphytic flora is much less conspicuous than in the Elfin forest, with branches not fully covered by bryophytes or other epiphytic plants. It has many characteristic species in common with the Elfin forest, including two species of tree fern (*Cyathea* species), the palm *Prestoea acuminata*, the trumpet tree (*Cecropia peltata*), the epiphyte *Vriesea ringens* and the low herb *Begonia retusa*. Some species from the Elfin forest grow downwards into the higher levels of the Montane forest, e.g. the low shrub *Rubus rosifolius* and the fern *Nephrolepis rivularis*. In the lower belts of the Montane forest, species are found that are characteristic of the Dry tropical forest, such as the woody species *Myrcia splendens*, *Coccoloba diversifolia*, *Cordia sulcata*, *Clusia major*, *Citharexylum spinosum* and *Miconia laevigata*. The elephant ear (*Philodendron giganteum*) and the fern *Blechnum occidentale* are found in all three forest habitats. This habitat type is also present on St. Eustatius.

Dry tropical forest forms the lower forest belt on the slopes of Mount Scenery and occurs roughly between 200 and 500 m altitude. In many placed this forest is degraded, resulting in shrubland or even grassland, but as such vegetation types also have natural occurrences, these degraded stages have been described as a separate habitat type. The Dry tropical forest has many species in common with the previous forest type, but the following trees and shrubs are more-or-less restricted to this habitat: *Bursera simaruba*, *Pisonia subcordata*, *Coccoloba uvifera*, *Inga laurina*, *Casearia decandra*, *Eugenia axillaris*, *Myrcianthes fragrans*, *Randia aculeata*, *Maytenus laevigata* and *Guapira fragrans*. In the understorey *Peperomia magnoliifolia* and *Peperomia myrtifolia* are relatively abundant, as they are not eaten by goats. The Dry tropical forest resembles forest at the same zonation belt on Sint Maarten and St. Eustatius. This forest type has some species in common with the best-developed dry forests of the leeward islands, but overall, differences in species composition between the dry forests of the windward and leeward islands are large.

Figure 2. A view on Swan Gut from the Dancing Place Trail. It is remarkable that the patterns between low shrub vegetation and grasslands on these slopes have nearly not changed between the 1950s (left; Stoffers 1956) and 2020 (right; photo: John Janssen). Before the removal program for roaming livestock began in 2021, the slopes were heavily grazed by goats, but as the patterns differ on both slopes they seem mainly the result of exposure to the strong trade winds, resulting in grassy patches on the rim and higher, exposed parts of the slopes and shrubby vegetation with some trees in the gullies and lower, more sheltered slopes.

Dry shrubland and grassland occurs partly as primary vegetation and partly due to degradation of Dry tropical forest. The (over)grazing by goats helps such secondary occurrences to endure, as it prevents succession towards forest. The natural (primary) occurrences of these low vegetation types are found in extreme environments, such as on steep, rocky and eroding slopes and in places that are exposed to strong winds or to salt spray from the sea. In these extreme environments, it forms the climax vegetation. It is difficult to distinguish between the primary and secondary sites, but figure 2 provides some insight: probably Dry shrubland and grassland on many sites on the wind-exposed part of the island are natural. This habitat is rather heterogeneous, even within the shrubland and within the grassland. Important and dominant shrub species in the shrublands are *Croton astroites*, *Lantana camara*, *L. involucrata*, *Mitracarpus polycladus*, *Plumbago scandens*, *Jatropha gossypiifolia* and *Pentalinon luteum*. The grasslands are dominated by *Aristida adscensionis*, *Dactyloctenium aegyptium*, *Stylosanthes hamata*, *Desmodium triflorum*, *Chloris barbata* and the non-native hurricane grass (*Botriochloa pertusa*). On extreme steep cliffs close to the sea, this habitat is rather inaccessible, and some variation of this habitat, for example with stands of *Coccoloba uvifera* shrubs, are for that reason not represented in the plot sample.

In conclusion, the subtle gradient of vegetation types along the elevation of Mount Scenery has been generalized in four broad habitat types. Of these, the Elfin forest is unique within the Dutch Caribbean islands and it harbours the highest number of plant species that are restricted to a single island. Within the six Dutch Caribbean islands, well-developed Elfin forest is only present on Saba, although a much less typical form is present on St. Eustatius. The three forest types are much more species-rich than the shrubland and grassland. Montane forest is also present on St. Eustatius, although the species composition differs. The Dry tropical forest resembles forest at the same zonation belt on Sint Maarten and St. Eustatius. It also has some species in common with the best developed forests of the Dutch Leeward Islands. The Dry shrubland and grassland is a mixture of primary and secondary low vegetation. It is climax vegetation on steep, exposed slopes, but many parts are degraded forms of Dry tropical forest. The Dry shrubland and grassland habitat is highly similar to those present on St. Eustatius and Sint Maarten. Many species are shared with shrubland and grassland types on the leeward islands, but large differences can be observed as well.

Relative Importance Within the Caribbean

Within the Dutch Kingdom the **Elfin forest** is unique to Saba and St. Eustatius, although that on St. Eustatius is a different, albeit less species-rich form. The type is characterized by the tree *Freziera undulata*, while the accompanying *Charianthus* species varies between the different islands. The Elfin forest type dominated by mountain mahogany is restricted to the inner arc of the lesser Antillean islands, from Saba to Grenada (Beard, 1949). The Elfin forest is known to have a relatively high level of endemism, with different dominant species on each island. The area of this habitat is small on every Caribbean island, as it is restricted to ridges and high summits (Beard, 1949). Therefore, the area on Saba (the whole summit covers more than 7 ha in De Freitas et al., 2016), is relatively important. *Charianthus purpureus* is known only from Saba, Saint Kitts and Montserrat. On Saba it was described as an endemic variety *cirinus* in the past (Stoffers, 1956; Rojer, 1997). Also, the recently discovered endemic fern species *Amauropelta sabaensis* (Axelrod, 2021) grows in this habitat on Saba. Many other plant species of the Elfin forest have a limited distribution range, e.g. *Begonia retusa* and *Cyathea muricata*.

The **Montane forest** apart from Saba, is found on the higher slopes of the Quill volcano on St. Eustatius. The distribution in the wider range of the lesser Caribbean islands and Central and South America is unclear and deserves further studied. The distribution of the most characteristic species indicates that the montane forest community of Saba and St. Eustatius is more widespread than the Elfin forest. Within the Lesser Antilles, it is restricted to the islands of the inner arc, as the outer islands are older and relatively low. However, it is likely that the range of this habitat is more widespread, as several characteristic species range towards the larger Antillean islands and some others towards Central America or northern South America. The Montane forest is structurally the most developed plant community on Saba, especially in more sheltered places, where it may have developed relatively undisturbed over a long time. In other, more wind-exposed places, this habitat has the character of secondary forest, recovering from hurricanes. In such sites, tree ferns (*Cyathea* sp.), mountain palms (*Prestoea acuminata* var. *montana*) and elephant ears (*Philodendron giganteum*) have high cover.

Dry tropical forest is much more widespread than Montane forest. The vegetation is found extensively on St. Eustatius (Boven NP, Signal Hill, lower slopes of the Quill) and on the hills of Sint Maarten. The diversity in dry forest communities within the wider Caribbean is not yet well described, and therefore it is difficult to assess the international importance of the communities on Saba, St. Eustatius and Sint Maarten. Within the Lesser Antilles, this habitat is found in both the inner arc of volcanic islands and the outer arc of older, lower islands. The characteristic tree, *Pisonia subcordata,* is restricted to the Lesser

Antilles, Puerto Rico and Jamaica and *Maytenus laevigata* is restricted to the Lesser Antilles. Many other species are much more widespread, for example, *Bursera simaruba*, *Guapira fragrans*, *Randia aculeata*, *Senna bicapsularis* and *Wedelia calycina* occur also in Central America and northern South America. These species' distribution data suggest, therefore, that this habitat type is much more widespread than the Montane forest. The best developed dry forest communities of the leeward islands (Aruba, Bonaire, Curaçao) share many species with the dry forests on Saba, St. Eustatius and Sint Maarten, however, each group of islands has a distinct set of species not present on other islands. Based on these differences in species composition and functioning, the dry forest habitat on the leeward islands should be considered as one or two different habitat types, probably separating those on limestone and volcanic soils (see for instance Beers et al., 1997; De Freitas et al., 2005). Important is that on a high level of classification, the dry tropical forest is amongst the most threatened forest ecosystems of the Neotropics (Ferrer-Paris et al., 2019).

Dry shrubland and grassland is widespread in the Caribbean and is found on all windward and leeward islands. The species composition differs between the leeward and windward islands for the dry shrubland, but both island groups also share many species, e.g. *Lantana camara* and *L. involucrata*. Dry shrublands are widespread on the leeward islands but on the windward islands are restricted to lower, dry slopes. Dry shrubland on Aruba, Bonaire and Curaçao resembles those on Saba, St. Eustatius and Sint Maarten in structure and species composition, although each island group has its own unique set of species too. This is illustrated by the *Croton* species occurring as characteristic species in the dry shrubland types: *Croton astroites* is more-or-less restricted to the lesser Antillean islands and Puerto Rico, *C. flavens* occurs throughout the Caribbean and Mexico, and *C. conduplicatus* is largely restricted to South America including the ABC islands (www.worldfloraonline.org). As their functioning and species composition differ, one may distinguish several subtypes of Dry shrubland, both for the windward and leeward islands, and for volcanic and calcareous soils. The distribution of this habitat or its sub-habitats in the wider region is unknown. The grasslands on both island groups have the same dominant species (*Aristida adscensionis* and *Bothriochloa pertusa*) and can be considered as one and the same (sub)habitat. Both dominant grass species and many other characteristic species (*Desmodium triflorum*, *Dactyloctenium aegyptium*, *Euphorbia serpens*, *Stylosanthes hamata*, *Tragus berteroninus*) are widespread in the Caribbean including the ABC islands, as well as in Central and South America, illustrating the wide distribution of the grassland habitat.

Table 1. *Relative importance of habitat types on Saba within the wider region. Surface areas on Saba and St. Eustatius are derived from the map of De Freitas et al. (2016; 2014).*

Habitat type	Distribution (1x1 km-grids)	Area (ha)	Worldwide range	International importance
Lowland tropical rainforest	Saba (0) St. Eustatius (1)	Saba (0) St. Eustatius (34)	Caribbean, Central America, South America (with regional differences)	High
Elfin forest	Saba (1) St. Eustatius (1)	Saba (7.2) St. Eustatius (2-4*)	Inner arc of the lesser Antillean islands	High
Montane forest	Saba (6) St. Eustatius (7)	Saba (200) St. Eustatius (166)	Inner arc of the lesser Antillean islands, possibly also Greater Antillean islands	High
Dry tropical forest	Saba (19) St. Eustatius (29)	Saba (400) St. Eustatius (813)	Lesser and Greater Antillean islands, Central America and northern South America (leeward islands have different subtypes)	Medium

Dry shrubland and grassland	Total Saba (24) St. Eustatius (35) Limestone subtype Saba (0) St. Eustatius (3) Volcanic subtype Saba (24) St. Eustatius (25) Coastal scrub subtype Saba (1?) St. Eustatius (30)	Total Saba (470) St. Eustatius (1456) Limestone subtype Saba (0) St. Eustatius (20) Volcanic subtype Saba (470) St. Eustatius (1157) Coastal scrub subtype Saba (< 1) St. Eustatius (278)	Lesser and Greater Antillean islands, Central America and northern South America (leeward islands have different subtypes)	Low

Developments and Trends

In the 1950s, Saba had many small agricultural fields, in which people used to grow vegetables and fruit for their own consumption. These are depicted roughly on the vegetation map made by Stoffers (1956). Since then, most of these fields have been abandoned, which resulted in a slow recovery of forest. However, at the same time, the need to limit roaming livestock had been ignored and in recent decades this had become a major problem on both Saba and St. Eustatius (Debrot et al., 2018). On Saba, not only do goats form a threat, but also many chickens roam free, the latter especially in the higher parts of the island. By eating fruits and seedlings they prevent the regeneration of many woody species, as well as herbs and grasses. However, the Public Entity of Saba (PES) has started a reforestation project in 2022 with the aim to outplant 5,000 trees, focusing on erosion prone areas. Invasive species control measures are ongoing and currently focus on goats, cats, rats, non-native iguanas and chickens. Initiatives for reforestation and reduction of these pressures are further mentioned under the heading of "future prospects".

Urbanization is a relatively minor pressure on Saba. On the vegetation map of De Freitas et al. (2016) about 8.6% of the island was indicated as urbanized area. This is a relatively low number, compared to many other Caribbean islands.

Climate change is especially a future threat. So far, it has not resulted in a noticeable shift of climatic belts and consequently any shift of vegetation belts. However, in several (neo)tropical mountain regions, increasing temperatures and rising cloud levels have already caused distributional shifts (mainly upslope) of montane biota, leading to alterations in biodiversity and ecological functions (Mata-Guel et al., 2023). It remains unclear to what extent such effects are to be expected in the future for Saba and St. Eustatius, where no upward shift for the Elfin forest is possible. Debrot and Bugter (2010) expect severe future impact in this habitat.

Trends in distribution and area
The trends in distribution and area have been studied by visual comparison of satellite images from 1991 and 2018, comparison of old and new photographs and maps, and by field observations. The distribution

of the four distinguished habitat types (in terms of 1x1 km-grid cells) did not change significantly over the past decades. Distribution is strongly related to altitude and exposure to wind.

The area of the Elfin forest and Montane forest has not changed significantly since 1991. In contrast, the area of Dry tropical forest has slightly increased since the 1950s, which is mainly the result of land abandonment. Small former agricultural fields have become overgrown with forest since abandonment, like along the Sandy Cruz trail and in the vicinity of Little and Big Rendezvous. Also, on some lower slopes, small patches of dry forest have recovered since the 1950s. The latter is illustrated in figure 1 by the slope of Bunker hill on the left, which in the 1950s was much opener than in 2020. The same trend is observed on other lower slopes surrounding the Bottom. On Parish Hill, one plot indicates a succession from shrubland in 1999 towards dry tropical forest in 2020. An explanation for these changes is, that also on these slopes land-use has changed, for example by a reduction of small-scale wood cutting and gathering, which – despite the pressure of goats – has led to succession from (secondary) grassland or shrubland towards dry forest. Notwithstanding these local changes, the area of the primary stands of grassland and shrubland remain unchanged.

Trends in structure, function and species composition
Overall, the trend analysis of the plot data shows an increase in vegetation height, indicating a trend towards higher forest. The total cover significantly increased for Dry forest and Dry shrubland and grassland. Total cover seems to have decreased for Elfin forest and Montane forest, although the trend is not significant. It should be noted that changes in cover can be caused by seasonal variation or observer bias. As in 1999, for several plots data for the tree and shrub layer was combined or data for shrub and herb layer was combined, a full analysis of trends in individual strata was not possible. Comparison of individual plot data however does render an indication of the changes in structure. Species richness decreased dramatically; all plots together showed a loss of approximately one third of the number of species. Roughly the same level of species diversity loss is observed in each individual habitat type, although trends are only significant for all plots together and for the Montane forest and the Dry tropical forest separately. This decline in species richness can be explained by succession and by overgrazing by free-roaming goats and chickens. The trends in structure and species composition are discussed in more detail for each habitat type below.

In the **Elfin Forest,** the data showed a trend towards higher forest, however, with slightly lower species diversity in woody species. The characteristic *Freziera undulata* has increased in numbers and dominance. De Freitas et al. (2016) describe how in 1999 this type was found in a completely degraded state after the hurricane George of 1998. Most large trees were destroyed and *Freziera undulata* was no longer the dominant tree species. The 2020 data indicate a recovery of this forest type. However, the recovery is not yet up to a climax situation, as is indicated by the photographs in figure 3. The recovery of the Elfin forest goes along with a decline in overall species richness as light-demanding species have difficulty in surviving under the gradually closing forest canopy. In addition, loss of species is worsened by the continuous overgrazing of the forest floor by goats and chickens.

Data for the **Montane forest** mainly indicate ongoing succession. The tree and shrub layer have become slightly higher (but not statistically significant), while the shrub and herb covers have slightly decreased (also not significant). Overall species richness declined significantly by more than 40%. This dramatic loss of diversity can only partly be attributed to the succession in forest structure (the forest becoming more closed and darker). Observer bias from different researchers may play a (minor) role too. However, neither of these two factors can account for such dramatic levels of species loss. Most likely, the steep decline in species diversity is primarily caused by the continuous and increasing over-grazing of the understory by goats and chickens. This is particularly worrying as many animal species are dependent on the availability of nectar and fruits of montane forest species, including the Bridled Quail-dove (*Geotrygon mystacea*) and the Saban Green iguana (*Iguana iguana* subsp. *melanoderma*).

The **Dry tropical forest** shows an increase in tree and shrub height (but not statistically significant), and a significant increase in overall cover. However, this increase, which can largely be attributed to a higher cover of the herb layer, goes along with a strong (significant) decrease of species richness, especially in the woody species. The decline in species richness of approximately one third of the species can only partly be explained by succession of the forest and consequent increase of tree layer cover. Similar to the other forest types, the continuous and increasing overgrazing by free-roaming goats and chickens is considered the main factor for the dramatic loss of species in the Dry tropical forest habitat.

Large changes were found in the low vegetation types of the **Dry shrubland and grassland**. Some of these patches have become overgrown by non-native species. In a shrubland area at Fort Hill, as well as at other localities, a dominance of the liana *Cryptostegia madagascariensis* was observed (figure 4), an invasive species that was not recorded in the 1999-data. In other Dry shrubland and grassland plots an increase of hurricane grass (*Bothriochloa pertusa*) was recorded. The increasing dominance of hurricane grass is probably the reason for the 20% increase in the total cover, which is made up of a 32% increase of the herb layer in combination with a 12% decrease (not significant) of the shrub layer. In addition, Guinea grass (*Megathyrsus maximus*) and Mother-of-thousands (*Kalanchoe* spp.), are present at many sites. Species richness of the Dry shrubland and grassland vegetation types is much lower compared to the forest types, but here too, a loss of approximately one third (but not significant) of the species was recorded between 1999 and 2020. Most of the species lost are woody species. As with the other types, here too, the continuous overgrazing of goats and chickens is considered the main factor for species loss. A second major explanatory factor for species loss is the increasing presence and cover of non-native plant species including hurricane grass.

Figure 3. *Elfin forest before (top left) and after (top right) hurricane George in September 1998 (photos: Tom van het Hof). Before the hurricane the dominant Freziera undulata trees had an estimated height of 10 to 15 m, when compared to the tree fern. In 2020 (bottom right, photo: John Janssen) the Freziera trees are recovering to a height of on average 9 m.*

Reference Values

Favourable reference values (FRVs) represent a situation in which a habitat type has enough distribution (in terms of 1x1 km-grids) and area (in terms of ha or km^2) for long-term survival. This may be analyzed by historical data or by modelling (see Bijlsma et al., 2019). Within the reporting for Natura 2000 in Europe, reference values are set for the evaluation of distribution and area (European Commission, 2017).

The distribution and area of the Elfin forest and Montane forest have not changed significantly since the 1950s (map by Stoffers, 1956). Similarly, the distribution of the Dry tropical forest and of the Dry shrubland and grassland has not changed significantly either. The area of the Dry tropical forest has declined historically due to clearing of forest for building activities and establishment of agricultural lands, but since the 1950s it has increased slightly, due to abandonment of small agricultural lands. As a reference for these three habitat types, we use the situation from 1999, as is indicated on the map of De Freitas (2016). For the Dry shrubland and grassland we use only the "natural" part of the current distribution and area, i.e. the primary sites. We assume that the distribution of natural stands is probably more-or-less the same as the overall distribution. The reference value for the shrubland and grassland area is roughly estimated at between 60 and 80% of the current area.

Elfin Forest: FRV distribution: 1 km-grid, FRV area: 7.2 ha
Montane forest: FRV distribution: 6 km-grids, FRV area: 200 ha
Dry tropical forest: FRV distribution: 19 km-grids, FRV area: 400 ha
Dry shrubland and grassland: FRV distribution: 24 km-grids, FRV area: 280-380 ha (rough estimate)

Figure 4. *The lower slope of Fort Hill is dominated by the non-native invasive shrub Cryptostegia madagascariensis in 2020 (photo: John Janssen).*

Assessment of Conservation State

Assessment of distribution:
For the assessment of the Conservation State the distribution should not be below the favourable reference value and not have a negative trend.

- Elfin Forest: good (stable and at reference value)
- Montane forest: good (stable and at reference value)
- Dry tropical forest: good (stable and at reference value)
- Dry shrubland and grassland: good (stable and at reference value)

Assessment of area:
For the assessment of the Conservation State the area should not be below the favourable reference value and not have a negative trend.

- Elfin Forest: good (stable and at reference value)
- Montane forest: good (stable and at reference value)
- Dry tropical forest: good (slightly positive trend and at reference value)
- Dry shrubland and grassland: good (slightly negative trend, but mainly for secondary sites; at reference value)

Assessment of structure and function, incl. species composition:
- Elfin Forest: poor, positive trend for structure (recovering from hurricane damage), but negative trend for functioning including species composition (decline of species richness).
- Montane forest: poor, positive trend for structure (continuous succession), but negative trend for functioning including species composition (decline of species richness).
- Dry tropical forest: poor, negative trend (structure and functioning heavily degraded by over-grazing, only few sites recovering from former land-use, decline of species-richness).
- Dry shrubland and grassland: poor, negative trend (structure and functioning heavily degraded by overgrazing and invasive plant species, decline of species richness).

Assessment of future prospects:
A positive prospect for the future is the fact that since 2020 a start has been made with tackling the problem of roaming livestock, especially goats. The NEPP for the Caribbean Netherlands assigns a high priority to culling uncontrolled roaming livestock (Min. LNV et al., 2020). It is stated by local government that since then most of the population has been removed. Additionally, some experiments with reforestation have started, including one exclosure near Sulfur Mine. The PES has an ongoing reforestation program and has already started with the outplanting of trees that have been grown from local seeds or seedlings. Outplanting focuses on erosion-prone areas. We expect recovery of the three forest habitats because of the combination of these management actions. Especially the dry tropical forest may benefit. However, no detailed monitoring is being carried out yet of the effects of goat removal on the four habitat types. Another positive development is the establishment of the Mt. Scenery National Park in 2018, which will contribute in the long-term conservation of the habitats present including the species that live there.

On the contrary, a future threat is climate change. It is likely that an increase of number and strength of hurricanes will occur, and – as has been shown from past data – this is a serious threat to the forest habitats on the island (see Fig. 3). Also, ambient temperatures are rising, and the amount of precipitation may change, resulting in an overall drier climate, which can have severe negative impact on the higher forest habitats as well. Especially the Elfin forest may be facing extinction on Saba in case climate change results in loss of cloud cover and decline in precipitation.

- Elfin Forest: unknown (positive prospects from goat removal and negative prospects from climate change)
- Montane forest: unknown (positive prospects from goat removal and negative prospects from climate change)
- Dry tropical forest: unknown (positive prospects from goat removal and negative prospects from climate change)
- Dry shrubland and grassland: unknown (positive from goat removal and negative from continuing threats from invasive species)

Conservation and Monitoring Objectives

Overall, our comparison between how the vegetation was in the late 1990s and how it was in 2020 showed large deterioration in all four habitat types, that they were in a poor state and faced severe threats from continuous overgrazing and from climate change (Table 2). Fortunately, positive prospects are present too, as exemplified by the current removal of free-roaming goats from practically the entire island, the post-hurricane recovery of the Elfin forest and the continuous succession of the Montane forest. Targeted removal of goats started in 2021 while island-wide removal started in 2023. In 2020, in terms of structure, the Elfin forest and Montane forest were in better Conservation State than the habitat types of the lower elevations. For conservation efforts it is important to focus on these types from high elevation, as these are most species-rich, contain the most (near) endemic species and have a higher importance from an international perspective. The main issues that should be tackled:

- Immediate and total removal of all remaining free-roaming goats and chicken.
- Continuation of the plot monitoring every 5 to 10 years with an increased number of plots for Elfin forest and for the subtypes of Dry shrubland and grassland to assess the long-term effects of climate change, removal of free-roaming grazers and other factors.
- Monitoring on the development of juvenile trees, herbs and grasses in exclosures and other goat-free areas to assess the effects of early stages of habitat restoration. Even on rocky and generally steep Saba there are plenty of locations where small exclosures and control plots can be established.
- Population monitoring of key species in each habitat, in particular (near-)endemic species and species essential for rare and endangered animal species.
- Assessment of the currently present non-native species to identify priorities in the management of invasive species.
- Immediate and total removal of those invasive species that pose the highest threats and are still in an early stage of invasiveness (e.g. *Cryptostegia madagascariensis*, *Azadirachta indica*, *Antigonon leptopus*, *Kalanchoe spec.*).

Table 2. *Summary overview of the status of the terrestrial habitats of Saba in terms of different conservations aspects.*

Habitat	**Elfin Forest**	**Montane forest**	**Dry tropical forest**	**Dry shrubland and grassland**
Distribution	Favourable	Favourable	Favourable	Favourable
Area	Favourable	Favourable	Favourable	Favourable
Structure, function & species composition	Unfavourable-bad-bad	Unfavourable-bad	Unfavourable-bad	Unfavourable-bad
Future prospects	Unknown (positive and negative developments)	Unknown (positive and negative developments)	Unknown (positive and negative developments)	Unknown (positive and negative developments)
Overall Assessment of Conservation State	**Unfavourable-bad**	**Unfavourable-bad**	**Unfavourable-bad**	**Unfavourable-bad**

Comparison to the 2018 State of Nature Report

Overall, the CS of terrestrial vegetations and the habitats they define on Saba had remained equally worrisome in 2020 compared to the 2018 assessment. Fortunately, since 2020 the PES has been making rapid headway in terms of goat culling with practically all goats removed by end of 2024. If goat densities can be maintained low for a longer period of time (hopefully permanently) then improvements in the vegetation should soon become apparent and monitoring follow-up is recommended with which to document the expected changes.

Data Quality and Completeness

Trends were analyzed based on comparing 1999 and 2020 vegetation surveys in 34 plots over the island, and by additional information from remote sensing images, maps, field observations and literature. For the Elfin forest habitat few plots were available, but extra plots have been recorded for future monitoring. For the Montane forest and the Dry tropical forest relatively many plots were resurveyed, but some data were recorded in a different way than in the past and, consequently, are not well comparable between the two periods. In the Dry shrubland and grassland habitat, a low number of plots were resurveyed, especially when considering the large diversity in terms of dominant species, including non-native species. The 2020 vegetation survey, including the additional plots made in the Elfin forest, may be used as a baseline when assessing the effects of the goat removal that started that same year.

No data exists on rocky cliff-vegetation. No data exists on recovery of vegetation and restoration of habitats. The distribution of plant communities in the wider region, and therefore the international importance of the habitats, is yet unknown.

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4 Conservation State of the Terrestrial Vegetations of St. Eustatius

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Vegetation and Habitat ("Vegetation") Diversity

On the island of St. Eustatius, three distinct regions can be distinguished: the "Northern Hills", an area of older volcanic hills in the northern part of the island; the "Quill", the younger, steep volcano in the South, and; the flat area in between called the Central Plain or 'Cultuurvlakte' (Fig. 1). On the far south end of St. Eustatius, White Wall and Sugar Loaf form a fourth distinct geological element. This limestone complex was uplifted and tilted by volcanic activities and is the only limestone formation present on St. Eustatius (Westerman and Kiel, 1961). Land use and vegetation differ between the four regions. The majority of the Central Plain is covered by the roads, houses and other man-made structures of Oranjestad, the airport, as well as agricultural fields, with little area remaining for natural vegetation. The Northern Hills, comprising Boven, Bergje, Gilboa, Mary's Glory, and Signal Hill, are largely covered by natural vegetation, except for the oil terminal which covers approximately one fifth of the area. The Quill shows a complex mix of largely primary vegetation on the steep upper slopes and crater rim, and secondary forest on the lower slopes and crater floor, that have been used for agriculture in the past, and on sites that have been severely hit by hurricanes during the last decades. The present vegetation on both the Quill and The Northern Hills is developed along a steep gradient from sea level to the summits. De Freitas et al. (2014) distinguished thirteen vegetation types. These are generalized into five broader natural habitats: Elfin forest, Lowland tropical rainforest, Montane forest, Dry tropical forest, and Dry shrubland and grassland. For the Dry shrubland and grassland three subtypes are distinguished, two for volcanic soils and one for limestone soils. These five types are described here, and their status and trends are assessed. A trend analysis of plot data from a vegetation survey in 1999 (De Freitas et al., 2014) and a resurvey of these in 2023 (Van Proosdij et al., in prep.) provides relevant insight in the status and trends of the terrestrial habitat types on St. Eustatius.

With a surface of only 21 km^2, St. Eustatius is the second smallest island of the Dutch Caribbean, after Saba. The top of the Quill reaches 601 m above sea level. A total of 626 different vascular plant taxa have been reported from St. Eustatius (Axelrod, 2017), although during recent field work approximately 20 additional species have been recorded (van Proosdij et al., in prep). This high diversity is a result of the steep gradient in elevation, causing a broad range of micro-climates and micro-habitats with different expositions to sun and wind, and different precipitation, humidity and temperature. In contrast to Saba, the human impact on St. Eustatius is much larger. The island currently has approximately 3,200 inhabitants, which is much lower than the historic numbers. In the 18^{th} century, the population peaked at more than 25,000 and large parts of the island were used to produce cane sugar, cattle breeding as well cultivation of other crops (Teenstra, 1977).

Felling of trees for firewood and charcoal took place on the higher slopes of the Quill. The Central Plain, large parts of The Northern Hills and even the crater floor of the Quill were used for agricultural activities in that era. After the decline of trade in the first half of the 19^{th} century, population numbers dropped quickly to below 3,000 and even further in the 20^{th} century. As a result of these historical land uses, much of the currently present forests on the slopes of The Northern Hills and lower slopes and crater floor of the Quill are secondary forests. In the past decades, particularly after 2010, urban development took place on many places in the Central Plain and lower slopes of the Quill.

Figure 1. *View from Boven on Gilboa towards the Central Plain "Cultuurvlakte" and the Quill. In the background, St. Kitts covered in clouds, photo André van Proosdij.*

Characteristics of the Five Habitat ("Vegetation") Types

Elfin forest on St. Eustatius is limited to the highest part of the Quill crater rim. There is ongoing debate if this type is still present as well as to its past and current area of occupancy. According to Stoffers (1956), this type was only present on the highest part of the crater rim. During the vegetation survey in 1999, the Elfin forest was not mapped (De Freitas et al., 2014). De Freitas and Debrot (in Debrot et al., 2017), estimated the area covered by Elfin forest at 4.5 ha, but expressed worries that it might have been completely lost. The Elfin forest was indicated by Debrot et al. (2017, Fig. 2.1.3) to occur on the NW part of the crater rim around Panorama Point, which today is no longer the case as that area suffered a documented forest fire in the past (De Freitas et al., 2005)**.** In fact, it occurs on the much higher eastern part of the crater rim around Mazinga Peak (Van Proosdij and Janssen, pers. obs.). It can be recognized on and around Mazinga Peak from the stems and branches of gnarled trees and shrubs that are covered in thick layers of mosses, in particular large quantities of *Orthostichopsis tetragona* (Wiersma, 1984). This moss is restricted to the Elfin forest and is a typical feature of this type (Fig. 2).

Species composition on St. Eustatius differs from the Saban Elfin forest and many of the typical Elfin forest species of the Saban Elfin forest are lacking on St. Eustatius, e.g. *Freziera undulata* and *Miconia purpurea* (= *Charianthus purpureus*). Based on new vegetation relevées from 2022-2023, dominant woody species in the Elfin forest on St. Eustatius include *Ternstroemia peduncularis* (not known from Saba), *Coccoloba swartzii*, *Clusia major* and *Cestrum citrifolium*. Some fern species are restricted to the Elfin forest, e.g. *Elaphoglossum martinicense* and *E. petiolatum*. According to Stoffers (1956), *Begonia retusa* and *Tillandsia usneoides* are limited to Elfin forest as well, but these have not been observed in the past 70-115 years. In conclusion, although Elfin forest on St. Eustatius is often regarded as a more species-poor form than on Saba, it contains several rare and narrow-ranged species including some not present on Saba. It therefore represents a different, albeit less species-rich form of Elfin forest.

Figure 2. *Elfin forest showing typical high cover of stems and branches by the moss species Orthostichopsis tetragona, photo John Janssen.*

The most important ecological factor is the high air humidity, which is caused by clouds that develop almost every day around the top of the Quill. Because of the clouds, Elfin forest is also referred to as Cloud forest or Mist forest. Because of the relatively shrubby appearance of trees, the plant community is also referred to as Elfin woodland. The altitude of occurrence on St. Eustatius (above approx. 550 m) is not the most relevant abiotic factor: similar elfin forests on other Caribbean islands are mostly found at higher altitudes, but always on the summit or highest ridges of volcanic mountains (Beard, 1949). Its occurrence at lower altitudes on Saba and St. Eustatius, as compared to other Caribbean islands, is probably caused by the steepness of the mountains and proximity to the ocean, forcing moist air from sea to rise, which results in the formation of clouds.

Montane forest covers the higher slopes and rim of the Quill, roughly between 200 and 500 m altitude. Its exact range depends on the slope exposition. On exposed sites, the vegetation is often only 5-10 m high, whereas in sites that are sheltered from the wind, such as in gullies, trees grow much taller and different tree layers can be distinguished. The epiphytic flora is much less conspicuous than in the Elfin forest, with branches not fully covered by bryophytes or other epiphytic plants. *Chionanthus compactus*, *Damburneya coriacea* (syn. *Nectandra coriacea*), *Maytenus laevigata* and *Coccoloba swartzii* dominate the tree layer and *Ardisia obovata* is characteristic for the shrub layer. Some species are only found in this type: *Cordia sulcata*, *Exothea paniculata*, *Sideroxylon foetidissimum*, *Inga laurina*, *Byrsonima spicata* and the epiphytes *Werauhia ringens*, *Serpocaulon triseriale* and *Vittaria lineata*. Some are shared with the Elfin forest, e.g. *Ternstroemia peduncularis* and *Clusia major*. Other species are also found in the Dry tropical forest but have their optimum in the Montane forest type: *Citharexylum spinosum*, *Gyminda latifolia* and *Pleopeltis polypodioides*. Several species are shared with the Lowland rainforest, e.g. *Myrcia splendens*, *Peperomia magnoliifolia* and *Anthurium cordatum*. Montane forest on St. Eustatius resembles the Montane forest type on Saba in structure and many species are shared, although differences exist between the two islands. Montane forest is not present on other Dutch Caribbean islands.

Lowland tropical rainforest on St. Eustatius is restricted to the crater floor, where the surrounding, steep slopes of the volcano create a sheltered, moist microclimate. Here, the forest is protected from most strong winds. Trees in this type can reach heights of 40 m and impressive diameters. Emergent trees are covered with large numbers of epiphytic ferns, orchids and bromeliads. Some species are restricted to this type including *Quararibea turbinata*, *Coccoloba venosa*, *Piper reticulatum*, *Hirtella triandra*, *Faramea occidentalis*, *Asplenium cristatum*, *Philodendron lingulatum* and *Miconia impetiolaris*. Emergent tree species in this type are *Ceiba pentandra*, *Cedrela odorata* and *Spondias mombin*, whereas *Myrcia splendens* and *Quararibea turbinata* dominate the lower tree layer. Liana diversity is high (e.g. *Hyperbaena domingensis*, *Pisonia aculeata* and *Smilax guianensis*) and lianas can have substantial cover. *Theobroma cacao* is a persistent species that was formerly cultivated on the crater floor. The surface of the crater floor is covered with large boulders with shallow, organic-rich soils in between. Boulders and stems of old and dead trees are covered with mosses, ferns and other epiphytes. The herb and shrub layers are sparse, except for areas with gaps in the tree canopy. In 2017, hurricane Irma caused severe damage to this type. Several emergent trees were damaged or lost, creating large canopy gaps that have subsequently been filled with lianas and shrubs. This type is not present on other Dutch Caribbean islands.

Dry tropical forest forms the lower forest belt on the slopes of the Quill and the higher parts of The Northern Hills and occurs roughly between 50 and 300 m altitude. In several places this forest type is degraded, resulting in shrubland or even grassland, particularly in The Northern Hills, but as such vegetation types also have natural occurrences, these degraded stages have been described as a separate habitat type. Typical, dominant tree species in the Dry tropical forest are *Pisonia subcordata*, *Guettarda scabra*, *Malpighia emarginata*, *Randia aculeata* and *Bourreria succulenta*. This type shares many species with the lower parts of the Montane forest type, e.g. *Eugenia axillaris*, *Guettarda scabra*, *Guapira fragrans*, *Bursera simaruba*, *Krugiodendron ferreum* and *Myrcianthes fragrans*. Trees grow up to 5-10 m high and the understory is often quite sparse. The Dry tropical forest on St. Eustatius resembles forests at the same zonation belt on Sint Maarten and Saba. This forest type has some species in common with the best developed dry forests of the leeward islands (Aruba, Bonaire and Curaçao), but overall, differences in species composition between the dry forests of the windward and leeward islands are large.

The Dry shrubland and grassland type consists of three subtypes. **Dry shrubland and grassland on limestone** occurs only on White Wall and Sugar Loaf. This subtype is easily recognized in the field by the white soil (Fig. 3). As the limestone is free-draining and this type is located on the south-facing slope of the Quill, water availability is strongly limiting the height and cover of the vegetation. The herb layer is sparse. Dominant species include the woody species *Stenostomum acutatum*, *Dodonaea elaeagnoides*, *D. viscosa* and *Crossopetalum rhacoma* as well as the grass species *Aristida adscensionis* and *Paspalum laxum*. *Pisonia subcordata* is shared with the Dry tropical forest, and many herb and grass species are shared with either the Dry tropical forest type and/or the Dry shrubland and grassland on volcanic soils. This subtype resembles coastal scrub and grassland types on Sint Maarten and the leeward islands, although large differences are observed in species composition.

Figure 3. *White Wall (left) and Sugar Loaf recognizable by the white calcareous soil, photos Jethro van't Hull.*

The **Dry shrubland and grassland on volcanic soils** subtype occurs partly as primary vegetation and partly due to degradation of Dry tropical forest. The (over)grazing by goats helps such secondary occurrences to endure, as it prevents succession towards forest. The natural (primary) occurrences of these low vegetation types are found in extreme environments, such as on steep, rocky and eroding slopes and in places that are exposed to strong winds or to salt spray from the sea. In these extreme environments, it forms the climax vegetation. It is difficult to distinguish between the primary and secondary sites, but figure 4 provides some insight: probably Dry shrubland and grassland on many sites on the wind-exposed part of the island are natural, but they may be much more restricted in ungrazed situations. This habitat type is rather heterogeneous, even within the shrubland and within the grassland, and often a mosaic pattern of grasslands and shrubland is observed. On St. Eustatius, large parts of this habitat are covered by the invasive Coralita vine (*Antigonon leptopus*), which can cover the vegetation completely and smother trees and shrubs. Where *Antigonon* is absent or not present in low cover, the shrub layer is dominated by *Rauvolfia viridis* and *Randia aculeata* and the herb layer is dominated by *Jatropha gossypiifolia*, *Sidastrum multiflorum*, *Sida cordifolia* and *Tragus berteronianus*. In the grassland parts of this subtype, the invasive grass *Botriochloa pertusa* is by far the most dominant species, followed by *Pentalinon luteum*, *Solanum bahamense*, *Lantana involucrata*, *Croton flavens*, *Vachellia tortuosa*, *Aristida adscensionis* and *Cynodon dactylon*. The shrubland and grassland vegetations share many herb and vine species of the Fabaceae and Malvaceae families, all of which occur with low cover. This subtype occurs on exposed sites on the lower slopes of the Quill and the Northern Hills. The Dry shrubland and grassland on St. Eustatius are very similar to shrubland and grassland types on Saba and Sint Maarten. They also share many species with Dry shrubland and grassland on the leeward islands, although differences in species composition are larger.

Figure 4. *Overgrazed vegetation on Boven Hill, grassland/scrub with few shrubs on exposed sites and open forest on more sheltered sites, photos Jethro van't Hull.*

The final subtype, **Coastal scrub on volcanic soil**, occurs on St. Eustatius on the steep cliffs but also on beaches and exposed lower slopes, close to the sea. Vegetation cover on these exposed sites is low and the number of species present is lower than for the Dry shrubland and grassland on volcanic soil subtype. *Coccoloba ufivera* is the most dominant species, followed by *Conocarpus erectus* and *Thespesia populnea*, both forming patches of shrubby vegetation. Other species are present in much lower numbers and cover and include the woody species *Jacquinia armillaris*, *Volkameria aculeata*, *Vachellia tortuosa* and *Ernodea littoralis*, as well as the herb species *Plumbago scandens*, *Ruellia tuberosa* and *Capraria biflora*. Several herb species are shared with the Dry shrubland and grassland on volcanic soil subtype. This subtype is also present on Saba and Sint Maarten, although on Saba only at very few places. This subtype is rare on the leeward islands, as these have karst plateaus along the coastline instead of volcanic soils; however, a similar vegetation may be found in sandy stretches and estuaries ("bocas").

In addition, two other habitat types are present on St. Eustatius, that cannot be described and assessed. Along the coast of the Northern Hills, steep cliffs are present that are covered with **coastal cliff vegetation**. These are situated much higher than the Coastal scrub on volcanic soil type. Due to the inaccessibility of this type, no data are available on its area, structure and species composition. The second type is formed by vegetation on the steep inner slopes of the Quill. An exploratory field survey in 2023 to this inland cliff vegetation identified two vegetation types limited to the almost vertical inner slopes of the Quill that are distinct from other habitat types on St. Eustatius, one on sun-exposed slopes and another on sheltered, shady slopes.

In conclusion, for St. Eustatius a total of seven terrestrial habitat types are described which together capture the ecological gradients along the elevation of both the Quill and the Northern Hills as well as the abiotic differences caused by soil type (limestone vs. volcanic soil) and exposition to wind and salt spray (coastal cliffs, exposed slopes and sheltered crater floor). Of these types, the Tropical lowland rainforest is unique within the Dutch Caribbean islands and the Elfin forest and Montane forest are shared only with Saba, though with a different species composition. The Dry tropical forest resembles forest at the same zonation belt on Sint Maarten and Saba and in a different form on the leeward islands. The Dry shrubland and grassland type is a mixture of primary and secondary low vegetations. It is climax vegetation on steep, exposed slopes, but many parts are degraded forms of Dry tropical forest. Dry shrubland and grasslands are highly similar to those present on Saba and Sint Maarten. Many species are shared with shrubland and grassland types on the leeward islands, but large differences can be observed as well. Rocky cliff-vegetation along the coast of the Northern Hills and rocky outcrop vegetations on the steep inner slopes of the Quill have been identified on St. Eustatius but due to lack of data not yet properly described.

Relative Importance Within the Caribbean

Within the Dutch Caribbean the **Elfin forest** is unique to Saba and St. Eustatius. The Elfin forest type dominated by mountain mahogany, is restricted to the inner arc of the lesser Antillean islands, from Saba to Grenada (Beard, 1949). The Elfin forest is known to have a relatively high level of endemism, with different dominant species on each island. The area of the habitat is small, on every Caribbean Island, as it is restricted to ridges and high summits (Beard, 1949). Although Elfin forest on St. Eustatius is often regarded as a more species-poor form, it contains several rare and narrow-ranged species including some not present on Saba. It therefore represents a different, albeit less species-rich form of Elfin forest. Based on the differences in species composition compared to Saba and the rarity of this type in the Caribbean, the importance of the Elfin forest on St. Eustatius is high.

Like the Elfin forest, within the Dutch Caribbean, the **Montane forest** is also found exclusively on St. Eustatius and Saba. The distribution in the wider range of the lesser Caribbean islands and Central and South America is unclear and needs further studied. The distribution of the most characteristic species indicates that the montane forest community of Saba and St. Eustatius is more widespread than the Elfin forest. Within the Lesser Antilles, it is restricted to the islands of the inner arc, as the outer arc islands are older and relatively low. However, it is likely that the range of this habitat is more widespread, as several characteristic species range towards the larger Antillean islands and some others towards Central America or northern South America. The Montane forest is the most species-rich plant community on Saba and the second most species-rich on St. Eustatius (after the Lowland tropical rainforest), especially in more sheltered places, where it may have developed relatively undisturbed over a long time. In other, more wind-exposed places, this habitat has the character of secondary forest, recovering from hurricanes. Based on the rarity of this type in the Caribbean and the high species-diversity, the international importance of the Montane forest on St. Eustatius is high.

Within the Dutch Caribbean the **Lowland tropical rainforest** is only present on St. Eustatius. This type is widespread in the Caribbean Region and into Central and South America. Several of the dominant tree species are distributed on the Greater and Lesser Antilles as well as in Central and South America, e.g. *Cedrela odorata*, *Ceiba pentandra* and *Spondias mombin*. Other species however are restricted to the Caribbean. As such, the Lowland tropical rainforest in the Caribbean differs from similar Central and South American types. Lowland tropical rainforest is present on few islands in the Caribbean, the area of occupancy is small and knowledge about its subtypes is limited. Given the rarity of this type in the Caribbean and particular the Lesser Antilles, the importance of the Lowland tropical rainforest on St. Eustatius is high.

Dry tropical forest is much more widespread than the other forest types and covers more area. The vegetation is found extensively on Saba and St. Eustatius and on the hills of Sint Maarten. The diversity in dry forest communities within the wider Caribbean is not yet well described, and therefore it is difficult to assess the international importance of the communities on Saba, St. Eustatius and Sint Maarten. Within the Lesser Antilles, this habitat is found in both the inner arc of volcanic islands and the outer arc of older, lower islands. The characteristic tree *Pisonia subcordata* is restricted to the Lesser Antilles, Puerto Rico and Jamaica and *Maytenus laevigata* is restricted to the Lesser Antilles. Many other species are much more widespread, for example, *Bursera simaruba*, *Guapira fragrans*, *Randia aculeata*, *Senna bicapsularis* and *Wedelia calycina* which are found also in Central America and northern South America. The impression is therefore that this habitat type is much more widespread than the Montane forest. The best developed dry forest communities of the leeward islands (Aruba, Bonaire, Curaçao) share many species with the dry forests on Saba, St. Eustatius and Sint Maarten, however, each group of islands has a distinct set of species not present on other islands. Based on these differences in species composition and functioning, the Dry tropical forest habitat on the leeward islands should be considered as one or two different habitat types, probably separating those on limestone and on volcanic soils (see for instance Beers et al., 1997; De Freitas et al., 2005). Important is that on a high level of classification, the Dry tropical forest is amongst the most threatened forest ecosystems of the Neotropics (Ferrer-Paris et al., 2019). Consequently, the international importance of St. Eustatius Dry tropical forest is medium.

Dry shrubland and grassland on limestone occurs only on White Wall and Sugar Loaf on St. Eustatius. This subtype is absent from Saba but resembles coastal scrub and grassland types on limestone on Sint Maarten and the leeward islands, although large differences are observed in species composition. Overall, this type is widespread in the wider region, but regional differences are not well studied.

Dry shrubland and grassland on volcanic soil is widespread in the Caribbean and is found on all windward and leeward islands. The species composition between the leeward and windward islands differs for the Dry shrubland, but both island groups also share many species, e.g. *Lantana camara* and *L. involucrata*. Dry shrublands are widespread on the leeward islands, but on the windward islands restricted to lower, dry slopes. Dry shrubland on Aruba, Bonaire and Curaçao resembles those on Saba, St. Eustatius and Sint Maarten in structure and species composition, although each island group has its own unique set of species too. This is illustrated by the *Croton* species occurring in the Dry shrubland types: *Croton astroites* is largely restricted to the lesser Antillean islands and Puerto Rico, *C. flavens* occurs throughout the Caribbean and Mexico, and *C. conduplicatus* occurs in South America and the ABC islands. As their functioning and species composition differ, one may distinguish several subtypes of Dry shrubland, both for the windward and leeward islands, and for volcanic and calcareous soils. The distribution of this habitat or sub-habitats in the wider region is unknown. The grasslands on both island groups have the same dominant species (*Aristida adscensionis* and *Bothriochloa pertusa*) and can be considered as one and the same (sub)habitat. Both dominant grass species, and many other grassland-type species (*Desmodium triflorum*, *Dactyloctenium aegyptium*, *Euphorbia serpens*, *Stylosanthes hamata*, *Tragus berteroninus*) are widespread in the Caribbean including the ABC islands, as well as in Central and South America, illustrating the wide distribution of the grassland habitat.

Coastal scrub on volcanic soil can be found throughout the Caribbean and into Central and South America, but usually, this type covers only small areas. In addition, many sites on and near beaches have been lost due to urban development. Based on the wide distribution in the Caribbean and the lower number of species, the St. Eustatius Dry shrubland, grassland and scrub on limestone and volcanic soils are of lower international importance than the previous types.

Table 1. *Distribution, area and relative international importance of habitat types on Saba and Statia. Surface areas on Saba and St. Eustatius are derived from the map of De Freitas et al. (2016, 2014). *Elfin forest on St. Eustatius estimated from field observations.*

Habitat type	Distribution (1x1 km-grids)	Area (ha)	Worldwide range	International importance
Elfin forest	Saba (1) St. Eustatius (1)	Saba (7.2) St. Eustatius (2-4*)	Inner arc of the lesser Antillean islands	High
Montane forest	Saba (6) St. Eustatius (7)	Saba (200) St. Eustatius (166)	Inner arc of the lesser Antillean islands, possibly also Greater Antillean islands	High
Lowland tropical rainforest	Saba (0) St. Eustatius (1)	Saba (0) St. Eustatius (34)	Caribbean, Central America, South America (with regional differences)	High

Dry tropical forest	Saba (19) St. Eustatius (29)	Saba (400) St. Eustatius (813)	Lesser and Greater Antillean islands, Central America and northern South America (leeward islands have different subtypes)	Medium
Dry shrubland and grassland	Total Saba (24) St. Eustatius (35) Limestone subtype Saba (0) St. Eustatius (3) Volcanic subtype Saba (24) St. Eustatius (25) Coastal scrub subtype Saba (1?) St. Eustatius (31)	Total Saba (470) St. Eustatius (1467) Limestone subtype Saba (0) St. Eustatius (20) Volcanic subtype Saba (470) St. Eustatius (1157) Coastal scrub subtype Saba (< 1) St. Eustatius (290)	Lesser and Greater Antillean islands, Central America and northern South America (leeward islands have different subtypes)	Low

Developments and Trends

Pressures and threats

In the 17th and 18th century, population numbers on St. Eustatius were much higher than today, reaching a maximum of 25,000 inhabitants (Teenstra, 1977). After the decline of trade, population numbers rapidly fell by the first half of the 19th century and further dropped to a minimum of 970 by 1950. Following the establishment of the oil terminal in 1982, population increased again to over 2,000 by 1996 and since 2010 further increased to the current number of 3,200 (Central Bureau of Statistics, 2024). Consequently, during the 17th and 18th century, most of St. Eustatius was used for production of food and cattle, and wood was collected from the Northern Hills and higher slopes of the Quill, resulting in a substantial loss of Dry tropical forest. During the first half of the 19th century, there was still a large production of sugar cane on the Cultuurvlakte and in the Northern Hills and bananas, soursop, coffee and cacao were grown on the crater floor of the Quill (Teenstra, 1977). By the early 1950s, there were still extensive production fields on the Cultuurvlakte and goats were grazing in the Northern Hills (Veenenbos, 1955). The lower slopes of the Quill were no longer used for agriculture, but instead were the domain of free-roaming cattle. By the turn of the millennium, only a small part of the Cultuurvlakte was still used for growing crops, whereas all parts of the island except the increased oil terminal were used for grazing by goats, chicken and cattle (De Freitas et al., 2014). Today, only small areas are still used for growing crops. However, the grazing pressure is intense. Especially free-roaming goats, with an estimated number of 7.600 (Madden, 2020), severely damage the vegetation and increase erosion (Debrot et al., 2018). In addition, many sheep and chickens roam free in all areas of the island including the slopes and even crater floor and rim of the Quill. Both on Saba and St. Eustatius, regeneration of many species, both woody and herbs is hampered by chickens eating fruits, seeds and seedlings. Urbanization is a minor pressure on St. Eustatius. Although new houses are being built at an increasing speed, this is mostly done in urban areas and on former agricultural lands (Fig. 5). Climate change is especially a future threat. It is unclear if it has already resulted in a noticeable shift of climatic belts and consequently, a shift of vegetation zonation on St. Eustatius. However, in several (neo)tropical mountain regions increasing temperatures and rising cloud levels have already caused distributional shifts (mainly upslope) of montane biota, leading to alterations in biodiversity and ecological functions (Mata-Guel et al., 2023). It remains unclear to what extent such effects can be expected in the future for Saba and St. Eustatius, where no upward shift for the Elfin forest is possible.

Debrot and Bugter (2010) expect severe future impact in this habitat. De Freitas and Debrot (in Debrot et al., 2017) expressed their worries that the Elfin forest as observed by Stoffers in the 1950s may have been lost already. The lack of records for some rare and typical species of the Elfin forest in the past 70-115 years (*Begonia retusa* and *Tillandsia usneoides*) could be the result of climate change, free roaming goats and chickens or a combination of these factors. As such, it’s unclear if climate change has already affected the Elfin forest on St. Eustatius. Other habitat types may be affected as well, due to vegetation degradation and loss of trees decreasing the islands resilience to storms.

Figure 5. *Oranjestad shown from the viewpoint on the Quill in 1980 (left, photo Anton Stoffers) and in 2023 (right, photo John Janssen). Over more than 50 years, large parts of the agricultural fields in the ‘cultuurvlakte’ have undergone urbanization, with an increase of trees between the buildings. On the lower slopes of the Quill regeneration of dry forest took place, while the hills of Signal hill (in the back) became more sparsely vegetated due to over grazing.*

Trends in distribution and area

The trends in distribution and area have been studied by visual comparison of aerial images from 1991, 2011, 2018 and 2024, comparison of old and new photographs and maps, and by field observations. The distribution (in terms of 1x1 km-grid cells) of the five distinguished habitat types and subtypes has not changed significantly over the past decades. Distribution is strongly related to altitude, exposure to wind and historical land use.

The area of the Lowland tropical rainforest and Montane forest has not changed significantly since 1991. Similarly, the area of the Dry shrubland and grassland on limestone subtype has not changed. The area of this subtype is limited by the extent of White Wall and Sugar Loaf. The Coastal scrub subtype has not changed much either, and is still limited to a small strip along the coast. Identifying the borders of the Dry tropical forest and shrubland on volcanic soil subtype from satellite images is challenging. In the field, often transition zones are observed. Similarly, changes in vegetation cover are notoriously difficult to identify and quantify from aerial photos, especially due to large seasonal differences (Fig. 6). Based on visual comparison between aerial images of 1991 and 2024, in many sites, open, shrubby vegetation (Shrubland on volcanic soil) appears to have become more open, whereas at other sites it has become denser. This is best illustrated by the area of Venus Bay (Fig. 7), where the valley has become much more densely covered and the hill slope west of Venus Bay has become more open in larger areas but in other areas shows an increase in cover. The boundaries of Dry tropical forest do not appear to have shifted, but this type too has become more open in some places and denser in others. The area of Dry tropical forest was substantially reduced in the 18th and 19 century but does not seem to have changed significantly since then. Similarly, notwithstanding many local changes, the area of Shrubland and Grasslands on volcanic soil does not appear – on average – to have changed significantly. The area of the Elfin forest type is not known. The estimated surface of approx. 4.5 ha by De Freitas and Debrot (in Debrot et al., 2017) is probably an overestimation. Based on the presence of the typical moss species *Orthostichopsis tetragona*, the area around Mazinga covered by Elfin forest is estimated at 2-4 ha (van Proosdij and Janssen, field observations).

Trends in structure, function and species composition

The trends in structure and species richness and composition have been assessed by repeating a vegetation survey in 2023 at 73 sample plots that were recorded before in 1999 (for the landscape ecological vegetation map by De Freitas et al., 2014). All plot data are stored in the CACTUS database (Janssen et al., 2023). The following trends in structure, functioning and species composition have been analyzed using a statistical comparison of the two data sets (Van Proosdij et al., in prep).

Overall, the plot data show an increase in vegetation height, indicating a trend towards higher forest. The total cover significantly decreased for Dry forest. Trends in cover for other types were not significant, but cover seems to decrease as well for Lowland tropical rainforest, Coastal scrub and Shrubland and Grasslands on volcanic soil, and negligibly increase for Montane forest and Limestone scrub.

Figure 6. *Seasonal differences in vegetation cover near Billy Gut where the invasive Coralita vine (Antigonon leptopus) demonstrates an immense increase of cover between April (left) and November (right) 2023, photos John Janssen (left) and André van Proosdij (right).*

Figure 7. *Aerial photo of Venus Bay by 2024 (left, Google Earth) and 1991 (right) showing changes in vegetation cover.*

It should be noted that changes in cover can be caused by seasonal effects (see Fig. 6), annual variation and observer bias. As in 1999, for several plots data for tree and shrub layer was combined or data for shrub and herb layer was combined, a full analysis of trend on individual strata was not possible. Comparison of individual plot data however does render an indication of the changes in structure.

Species richness decreased dramatically; all plots together showed a loss of approx. 20% of the total average number of species. Species losses were significant and substantial for Montane forest (-30%) and Dry tropical forest (-26%), but not significant for other types. When only herb species are considered, all plots together show an increase of species richness, which also is the case for all individual habitat types. However, increase of herb species richness is only significant for the Montane forest, Shrubland and Grasslands on volcanic soil and Coastal scrub. When only woody species are considered, all plots together show a decrease in species richness. Montane forest (-44%), Dry forest (-27%) and Shrubland and Grasslands on volcanic soil (-42%, but based on overall low numbers of species) show dramatic losses of species. This decline in woody species richness can be explained by lack of regeneration due to long-term, severe overgrazing by free-roaming goats and chickens. The trends in structure and species composition are discussed in more detail for each habitat type below.

The **Elfin forest** was not included in the trend analysis as this type was not surveyed during the vegetation mapping in 1999. Consequently, not much can be said on the structure, function and species composition of this type. It should be noted though that several species previously categorized as typical for this type have not been found in the past 70-115 years and this can be regarded as a sign indicating a decline of the species richness and functioning of this type. A thorough exploration of the area potentially occupied by this type and the species occurring in it is needed to combat this long-existing knowledge gap.

The **Montane forest** type has faced a dramatic decline in species richness, especially woody species. This may indicate ongoing succession. The tree and shrub layer have become significantly higher and thus the forest could have become darker. However, the increase of herb species richness indicates that light may not be the most limiting growing factor for species. Overall species richness declined significantly with 30% and woody species even with 44%. This dramatic loss of diversity can only partly be attributed to the succession in forest structure (the forest becoming more closed and darker). Observer bias of different researchers may play a (minor) role too. However, neither of these two factors can account for such dramatic levels of species loss. Most likely, the steep decline in species diversity is primarily caused by the continuous and even increasing over-grazing of the understory by goats and chickens. This is particularly worrying as many species are dependent on the availability of nectar and fruits of montane forest species, including Bridled Quail-dove (*Geotrygon mystacea*) and Lesser Antillean iguana (*Iguana delicatissima*).

The **Lowland tropical rainforest** is restricted to the crater floor of the Quill. Hurricane Irma severely damaged this forest in 2017, creating large gaps in the canopy. In the following years, gaps were filled with herbs and lianas and today, seven years post-hurricane, these gaps are largely filled with woody lianas and large shrubs. Young trees have been established too. The trend analysis did not provide significant results as there were only two plots of which the exact location is uncertain. However, based on field observations (Van Proosdij pers. obs., 2017; 2022; 2023), a post-hurricane recovery of the vegetation is evidently in progress. It is expected that the forest will close again over the next decades, followed by a change in species composition in the herb and shrub layers. Pre- and post-hurricane inventory data of Bridled Quail-dove (*Geotrygon mystacea*), show a strong decline of this species since 2017 (Rivera-Milán et al., 2021) and yet unpublished results from 2023/2024 surveys are even more worrying. This is attributed to direct structural vegetation damage and food limitation caused by hurricane Irma, but also to ongoing negative impacts of free-roaming chickens and goats on food availability and loss of eggs and chicks to cats and rats. As such, species composition and population sizes of understory species (including ecologically important taxa) are in a poor state. As for the Elfin forest, also for the Lowland tropical rainforest, data on its species composition and population sizes are sparse, especially for the lower parts of the steep inner slopes. A thorough exploration of the area occupied by this type and the species occurring in it is needed to address this long-existing knowledge gap.

The **Dry tropical forest** shows a small but significant increase in tree and shrub height, but a large, significant loss of overall cover (-22%). The loss of species richness in this type is seen for both herb species and woody species. Whereas loss or gain of herb species may be subject to seasonal effects, this cannot explain the observed dramatic loss of woody species in the shrub and tree layer (-27%). In the field, in all areas with this forest type, free-roaming goats were observed. Loss of vegetation cover and trampling of the soil results in erosion, as was observed at many places (Fig. 8). As for the other forest types, the continuous and increasing overgrazing by free-roaming goats and chicken is the main factor for the dramatic loss of species in the Dry tropical forest habitat.

Dry shrubland and grasslands on volcanic soil is dominated by many species that are not (frequently) eaten by goats, e.g. *Croton flavens*, *Lantana camara* and *L. involucrata*, *Pentalinon luteum* and *Jatropha gossypifolia*. This type is often present as a mosaic of grassland and patches of shrub vegetation. It is therefore not easy to quantify changes in the amounts of individual subtypes. This type shows a negligible change in overall species richness. However, the increase in herb species richness and decrease in woody species richness are both significant. In the Grasslands subtype the invasive Hurricane grass (*Bothriochloa pertusa*) has become the dominant species. The invasive Coralita vine (*Antigonon leptopus*) covers large areas of this type (both Dry shrubland and the Grasslands subtype) and smothers all plants it covers. The continuous overgrazing by free-roaming goats and chicken and the increased cover by Coralita hamper regeneration of woody species and drive the shift from woody towards herbaceous species.

The **Dry shrubland and grassland on limestone** subtype shows a minor increase in overall species richness. However, this increase is not significant and based on only a few plots. Here too, a shift from woody species towards herbaceous species is noticed, although these changes are not significant. As with the other types, the continuous overgrazing by especially free-roaming goats hampers regeneration of woody species. As this type is only found on White Wall and Sugar Loaf and covers only 20 hectares, several rare species restricted to limestone are vulnerable including e.g. *Ernodea littoralis*.

Coastal scrub on volcanic soil has shown an increase in species richness, almost entirely of herb species. Here too, changes are not significant, and the trend is based on small number of plots only. The average cover of this type has not changed significantly. This type is facing increased erosion due to the presence of free-roaming goats. Small clusters of *Coccoloba uvifera* can be found at several locations along the coast. The presence of *Conocarpus erectus* is limited to a single stand at Venus Bay and a small patch in Oranjebaai, and neither show any regeneration.

Reference Values

Favourable reference values (FRVs) represent a situation in which a habitat type has enough distribution (in terms of 1x1 km-grids) and area (in terms of ha or km^2) for long-term survival. This may be analysed by historical data or by modelling (see Bijlsma et al., 2019). Within the reporting for Natura 2000 in Europe reference values are set for the evaluation of distribution and area (European Commission, 2017). The distribution and area of the Elfin forest, Montane forest and Lowland tropical rainforest have not changed significantly since the 1950s (map by Stoffers, 1956). As a reference for Montane forest and Lowland tropical rainforest we use the situation from 1999, as is indicated on the map of De Freitas et al. (2014). For the Elfin forest, we use the approximated area and distribution based on recent field observations.

Figure 8. *Dry tropical forest above White Wall where due to overgrazing the herb layer is almost completely absent and erosion takes place, photo André van Proosdij.*

The distribution of the Dry tropical forest and of the Dry shrubland and grassland has not changed much since the 1950s. The area of the Dry tropical forest has declined historically due to clearing of forest for building activities, harvest of wood and establishment of agricultural lands, but since the 1950s it has increased slightly, due to abandonment of agricultural lands. Currently, parts of The Northern Hills and lower slopes of the Quill that historically were covered by Dry tropical forest are now covered by the Dry shrubland and grasslands on volcanic soil subtype. Consequently, the reference area for Dry tropical forest is larger than the current area (813 ha) and is estimated at 1000-1200 ha.

For the Dry shrubland and grassland type and subtypes we use only the “natural” part of the current distribution and area, i.e. the primary sites. We assume that the distribution of natural stands is probably more-or-less the same as the overall distribution. The reference value for the Dry shrubland and grassland area is the sum of the reference areas for the three subtypes. We consider the Dry shrubland and grassland on limestone and Coastal scrub on volcanic soil subtypes to currently cover their historic area and therefore for these subtypes we use the situation from 1999, as is indicated on the map of De Freitas et al. (2014). Dry shrubland and grasslands on volcanic soil currently cover a larger area than historically, as Dry forest on parts of The Northern Hills and lower slopes of the Quill has been degraded into Dry shrubland and grasslands. Therefore, we estimate the reference value for area of the Dry shrubland and grasslands on volcanic soil subtype at between 40-70% of the current area. The reference value for area of the three Dry shrubland and grassland subtypes together is then summed to 900-1100 ha.

Elfin Forest: FRV distribution: 1 km-grid, FRV area: 2-4 ha
Montane forest: FRV distribution: 7 km-grids, FRV area: 166 ha
Lowland tropical rainforest: FRV distribution: 1 km grid, FRV area: 34 ha
Dry tropical forest: FRV distribution: 29 km-grids, FRV area: 1000-1200 ha
Dry shrubland and grassland: FRV distribution: 35 km-grids, FRV area: 900-1100 ha
Dry shrubland and grasslands on volcanic soil: FRV distribution: 25 km-grids, FRV area: 600-800 ha

Dry shrubland and grassland on limestone: FRV distribution: 3 km-grids, FRV area: 20 ha
Coastal scrub on volcanic soil: FRV distribution: 31 km-grids, FRV area: 290 ha

Assessment of Conservation State

Assessment of distribution:
For the assessment of the Conservation State the distribution should not be below the favourable reference value and not have a negative trend.

- Elfin Forest: good (stable and at reference value)
- Montane forest: good (stable and at reference value)
- Lowland tropical rainforest: good (stable and at reference value)
- Dry tropical forest: unfavourable-inadequate (stable and at reference value)
- Dry shrubland and grassland (all three subtypes): good (stable and at reference value)

Assessment of area:
For the assessment of the Conservation State the area should not be below the favourable reference value and not have a negative trend.

- Elfin Forest: good (stable and at reference value)
- Montane forest: good (stable and at reference value)
- Lowland tropical rainforest: good (stable and at reference value)
- Dry tropical forest: good (below reference value, but slightly positive trend)
- Dry shrubland and grassland (all three subtypes): good (slightly negative trend, at reference value)

Assessment of structure and function, incl. species composition:
- Elfin Forest: unfavourable-inadequate, negative trend for functioning including species composition (indications of decline of species richness).
- Montane forest: unfavourable-bad, positive trend for structure (continuous succession), but negative trend for functioning including species composition (decline of species richness).
- Lowland tropical rainforest: unfavourable-bad, positive trend for structure (recovering from hurricane damage), but negative trend for functioning including species composition (unfavourable species composition and insufficient population sizes of understory species crucial to Bridled Quail-dove).
- Dry tropical forest: unfavourable-bad, negative trend (structure and functioning heavily degraded by over-grazing, only few sites recovering from former land-use, decline of species-richness).
- Dry shrubland and grassland (all three subtypes): unfavourable-bad, negative trend (structure and functioning heavily degraded by overgrazing and invasive plant species).

Assessment of future prospects:
A positive prospect for the future is the fact that since 2020 a start has been made with tackling the problem of roaming livestock, especially goats. The NEPP for the Caribbean Netherlands assigns a high priority to culling uncontrolled roaming livestock (Min. LNV et al., 2020). However, numbers are high, so removal has not yet resulted in measurable relief in terms of reducing ecological pressure or enabling meaningful recovery of St. Eustatius's natural environment. Additionally, a few exclosures have been established and in some of these, experiments with reforestation have started. We expect recovery of the four forest habitats can result because of these management actions. Especially the dry tropical forest may benefit. However, so far only limited monitoring is being conducted of the effects of goat removal and this is currently only done for exclosures in the Dry shrubland and grasslands on volcanic soil subtype. All habitats and their species are expected to greatly benefit from these two management activities. However, the grazing by chicken will remain and will need additional management actions.

On the contrary, a future threat is climate change. It is likely that an increase of number and strength of hurricanes will occur, and – as has been shown from past data – this is a serious threat to the forest habitats of the island. Secondly, temperatures are rising and the amount of precipitation may change, resulting in an overall drier climate, which can have severe negative impact on the higher forest habitats as well. Especially the Elfin forest may be facing extinction on St. Eustatius in case climate change results in a loss of cloud cover and decrease of precipitation.

The net result of these on future prospects remains unclear for all habitats:

- Elfin Forest: unknown (positive prospects from goat removal and negative prospects from climate change)
- Montane forest: unknown (positive prospects from goat removal and negative prospects from climate change)
- Lowland tropical rainforest: unknown (positive prospects from goat removal and negative prospects from climate change)
- Dry tropical forest: unknown (positive prospects from goat removal and negative prospects from climate change)
- Dry shrubland and grassland (all three subtypes): unknown (positive from goat removal and negative from continuing threats from invasive species)

Data quality and completeness:
Trends were analysed based on comparing 1999 and 2023 vegetation surveys in 73 plots over the island, and by additional information from remote sensing images, maps, field observations and literature. For the Elfin forest no plots and for the Lowland tropical rainforest only two plots were available. For the latter, additional plots have been recorded for future monitoring. For these two types, basic information on area (Elfin forest) and species composition (both types) is largely lacking, forming a worrying knowledge gap typically illustrated by the recent rediscovery after 138 years of large tree ferns (Nature Today 2023). For the Montane forest and the Dry tropical forest types relatively many plots were resurveyed, but some data was recorded in a different way in the past and consequently not fully comparable. In the Dry shrubland and grassland type many plots were resurveyed, although the number of plots was small for the Dry shrubland and grassland on limestone subtype and for the Coastal scrub on volcanic soil subtype. No data exist on rocky cliff-vegetation and only limited data exist on rocky outcrop vegetations on the steep inner slopes of the Quill. No data exists on recovery of forests within exclosures. The distribution of plant communities in the wider region, and therefore the international importance of the habitats, is yet insufficiently known.

Conservation and Monitoring Objectives

Overall, we conclude that the Lowland tropical rainforest is in an unfavourable-inadequate state and the other four habitat types (Elfin forest, Montane forest, Dry tropical forest and Dry shrubland and grassland (all three subtypes) are in a unfavourable-bad state (Table 2). All types face severe threats from continuous overgrazing, some from climate change and some from invasive species. Fortunately, positive prospects are present too. These are the recent start with removal of free-roaming goats from the entire island, the post-hurricane recovery of the Lowland tropical rainforest and the continuous succession of the Montane forest. Considering structure, none of the types are in a good state, but the causes differ, including hurricane damage, overgrazing and effects of invasive species. For conservation efforts it is important to particularly focus on the Lowland tropical rainforest, the Elfin forest and the Montane forest types, as these are most species-rich, contain the most endemic species and have a higher importance from an international perspective. The main issues that should be tackled include:

- Immediate and total removal of all remaining free-roaming livestock: goats, sheep, cattle and chicken.

- Field assessment of the Elfin forest and Lowland tropical rainforest to identify the area (Elfin forest) and species composition (both types) as well as the presence of species that have not been recorded since the 1950s.
- Continuation of the plot monitoring every 5 to 10 years with an increased number of plots for Elfin forest, Lowland tropical rainforest and for the subtypes of Dry shrubland and grassland to assess the long-term effects of climate change, removal of free-roaming grazers and other factors.
- Monitoring on the development of juvenile trees, herbs and grasses in exclosures and other goat-free areas to assess the success of early stages of habitat restoration.
- Population monitoring of key species in each habitat, in particular (near-)endemic species (*Ipomoea sphenophylla*, *Gonolobus aloiensis*, *Begonia retusa*) and species essential for rare and endangered animal species.
- Assessment of the currently present non-native species to identify priorities for invasive species management.
- Immediate and total removal of those invasive species that pose the highest threats and are still in an early stage of invasiveness (e.g. *Cryptostegia madagascariensis*, *Kalanchoe spec.*) and removal of *Antigonon leptopus* and *Azadirachta indica* from conservation areas.

Table 2. *Summary overview of the status of the terrestrial habitats of St. Eustatius in terms of different conservations aspects.*

Habitat	**Lowland tropical rainforest**	**Elfin Forest**	**Montane forest**	**Dry tropical forest**	**Dry shrubland and grassland (all three subtypes)**
Distribution	Favourable	Favourable	Favourable	Favourable	Favourable
Area				Unfavourable-inadequate (positive trend)	Favourable
Structure, function & species composition	Unfavourable-inadequate (positive and negative trends)	Unfavourable-bad (indications of declining)	Unfavourable-bad (positive and negative trends)	Unfavourable-bad (declining)	Unfavourable-bad (declining)
Future prospects	Unknown (positive and negative prospects)	Unknown (positive and negative prospects)	Unknown (positive and negative prospects)	Unknown (positive and negative prospects)	Unknown (positive and negative prospects)
Overall Assessment of Conservation State	**Unfavourable-inadequate**	**Unfavourable-bad**	**Unfavourable-bad**	**Unfavourable-bad**	**Unfavourable-bad**

Comparison to the 2018 State of Nature Report

Overall, the CS of terrestrial vegetations and the habitats they define on St. Eustatius has gotten worse since the 2018 assessment.

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5 Conservation State of the Caves of Bonaire

Henkens, R. J. H. G. and Simal, F. 2025. From: State of Nature Report for the Caribbean Netherlands, 2024. WUR report C001/25.

International Status

Within the EU, caves are a protected habitat type under the European Habitats Directive, primarily due to the presence of bats and other often unique fauna. Unlike many habitats in Europe, caves in the Caribbean do not have an internationally protected status as a habitat.

Characteristics

Caves are primarily found in relatively soft limestone, which dissolves and erodes well under the influence of water and wind. Saba and St. Eustatius have a volcanic origin. On these islands, caves appear to be rare. This contrasts with Bonaire where limestone rock is abundant, and caves and karst formation are common (Wagenaar Hummelink, 1979).

About one-third of Bonaire's geology is of volcanic origin, but the remainder consists of Quaternary limestone. Over time, hundreds of dry and wet caves have formed in this limestone. Caves host unique life forms and serve as a crucial habitat for at least five species of bats. These bats are, as far as known, the only surviving native terrestrial mammals of the Caribbean Netherlands, with two of these species playing a key role in Bonaire's terrestrial ecosystem.

Habitat definition
Bonaire has both dry and wet caves. The water-holding caves are mainly located in the lower parts of the island. Additionally, there are sea caves with entrances located underwater.

Quality requirements
Caves have few to no abiotic conditions. The most important factor is ensuring peace for the fauna and maintaining water quality in the case of water-filled (wet) caves.

Typical Species
The caves are particularly important for bats, which use them as resting and breeding places. Additionally, shrimp and various endemic freshwater crustaceans are found in the water-filled caves. There are nine known species of bats on Bonaire (Table 1), but since the late 1990s, only *M. nesopolus* (Larsen et al., 2012), *M. molossus, M. megalophylla, L. curasoae,* and *G. longirostris* have been observed (Rojer, draft report, in Smith et al., 2012, Simal et al., 2021). The first two are insectivores, and the last two are nectar-dependent and play a crucial role in the terrestrial ecology of Bonaire. They are the only species capable of pollinating the night-blooming columnar cacti (*Subpilocereus repandus, Stenocereus griseus*, and likely *Pilosocereus lanuginosus*; Nassar et al., 2003) on Bonaire. These cactus flowers and fruits form a critical food source for the fauna of Bonaire during the dry season. *Pteronotus davyi* has been observed once and in 2023 a stray Visored bat, *Sphaeronycteris toxophyllum* was also collected (pers. comm. F. Simal).

In water-filled caves, the Blind Shrimp (*Typhlatya monae*) is also found. This shrimp is consumed by *Macrobrachium lucifugum* (another shrimp species), and it is assumed that these two species are found in different cave systems. *T. monae* is mostly below the halocline in anoxic waters (Debrot, 2003a). Table 1 lists some species that might qualify as typical species for caves in the Dutch Caribbean.

Table 1. *Potential Typical Species for Caves on Bonaire.*

Scientific name	Common name	Food	IUCN category	Species group	Endangerment (E)
Molossus molossus	Velvety Free-tailed Bat / Pallas's Mastiff Bat	Insects	LC	Bats	E
Mormoops megalophylla	Peter's Ghost-faced Bat	Insects	LC	Bats	E
Natalus tumidirostris	Funnel-eared Bat	Insects	LC	Bats	E
Myotis nesopolus	Little Brown Bat	Insects	LC	Bats	E
Pteronotus davyi	Naked-backed bat	Insects	LC	Bats	E
Ametrida centurio	Small Leaf-nosed Bat	Fruit	LC	Bats	E
Noctilio leporinus	Greater Bulldog Bat	Vis	LC	Bats	E
Leptonycteris curasoae	Lesser Longnose Bat	Nectar	VU	Bats	E
Glossophaga longirostris	Common Long-tongued Bat	Nectar	DD	Bats	E
Typhlatya monae	Mona Cave Shrimp	-	LC	Shrimps	E
Macrobrachium lucifugum	-	-		Shrimps	E
Ingolfiella putealis	-	Detritus	DD	Fresh water crustaceans	E
Psammogammarus caesicolus	-	Detritus	DD	Fresh water crustaceans	E

Additional Characteristics of Good Structure and Function:
The absence of human disturbance, soil contamination, and groundwater contamination from sewage and oil leaks is essential.

Environmental Quality Requirements:
Ensuring tranquillity is of paramount importance for the protection and preservation of cave fauna.

Relative Importance Within the Caribbean

Bonaire has hundreds of limestone caves, which are particularly important for bats. Recent research indicates that Long-nosed Bats (*Leptonycteris curasoae*) can travel between the Caribbean islands of Bonaire, Curaçao, Aruba, and the mainland of Venezuela (Simal et al., 2015; DCNA, 2014; De Lannoy, 2013). While limestone caves are unique within the Caribbean Netherlands, their significance to the Caribbean as a whole, is limited. Many of these caves are connected to the groundwater and are important habitats for native freshwater and brackish water fish and shrimp (Debrot 2003a, b). Additionally, the waters in these caves are rich in stygofauna. Considerable taxonomic work has been done on the endemic groundwater fauna of Bonaire (Stock, 1976a, b; 1977a, b; Vonk and Stock, 1987; Pesce, 1985), but little is known about the ecology of these species.

Current Status and Reference Values

Little can be said about the regional distribution of caves. However, it is certain that they occur on all the islands. Within the Caribbean Netherlands, they are primarily found on Bonaire, mainly in limestone formations. Cave entrances are often located in or near the slopes of various limestone terraces. Broadly, three limestone terraces can be distinguished. The oldest High Terrace ranges from 138 to 50 meters, much of which has been eroded. The younger Middle Terrace ranges from 15 to 45 meters. The youngest Low Terrace ranges from 4 to 15 meters and encircles the island almost completely. The Low Terrace generally ends in cliffs along the sea, but in the southeast, it is lower than 4 meters and transitions into recently formed sand ridges. The island of Klein Bonaire is entirely composed of limestone, with a central Middle Terrace and a surrounding Low Terrace that transitions into sand ridges (De Freitas et al., 2005). The northern and eastern parts of Bonaire are higher than the southern and western parts of the island. In the higher areas, water-bearing caves are found only in the Low Terrace. In the lower part of Bonaire, they can be found not only in the Low Terrace but also in the Middle Terrace. A large portion of the caves on Bonaire remains to be mapped. Bats and other fauna are not systematically monitored. As a result, reference values are unknown, making it difficult to determine the extent to which the caves and their fauna are improving or deteriorating.

Recent Developments

Since 2011, Bonaire, along with Aruba and Curaçao, has been a member of RELCOM (The Latin American and Caribbean Network for Bat Conservation; www.relcomlatinoamerica.net). One of RELCOM's primary strategies involves designating important bat conservation areas (AICOMs - Áreas de Importancia para la Conservación de Murciélagos) and preserving bat sites, such as bat caves (SICOMs - Sitios de Importancia para la Conservación de Murciélagos). RELCOM has designated one AICOM on Bonaire: Washington Slagbaai National Park (A-ABC-001). Additionally, in 2016, two caves on Bonaire were designated as SICOMs: Watapana and Lima (S-ABC-001).

These caves are currently unprotected but are likely the only roosts with large colonies of two insectivorous bat species: the Curaçao Little Brown Bat (*Myotis nesopolus*) and the Funnel-eared Bat (*Natalus tumidirostris*). The *N. tumidirostris* colony is likely an isolated population with relatively low numbers (<300), which makes the Bonairean population very vulnerable (source: RELCOM).
In 2016, the Caribbean Speleological Society (CARIBSS; www.caribss.org) was established on Bonaire. This organization focuses on exploring, mapping, protecting, and managing caves in the Caribbean. In 2017, the first phase of a project started, aimed at establishing a 'Bonaire Cave and Karst Reserve,' with activities including cave management, guide certification, sealing of bat roosts, and research into bat use of caves.

Several important freshwater caves may be threatened by infiltrating soil pollution and/or infiltration of sewage from nearby habitation (e.g., in Barcadera and Punt Vierkant).

Assessment of the National Conservation State

Trends: Trends in the occurrence of cave fauna and cave conditions are unknown. Since the last assessment for State of Nature reporting in 2018, no major changes in the Conservation State of beaches can be discerned.

Assessment of distribution: Favourable
The cave habitat primarily consists of 'dead' material. Natural distribution is thus not particularly relevant here. This does not apply to cave fauna. Caves on St. Eustatius and Saba are scarce due to the volcanic soil. Bonaire, on the other hand, is riddled with hundreds of caves. For bats, such as the Long-tongued

Bat (*Leptonycteris curasoae*), Bonaire is only part of the overall range, which extends across other Caribbean islands (Bonaire, Curaçao, Aruba) and the mainland of Venezuela, and likely Colombia (Simal et al., 2015; DCNA, 2014; De Lannoy, 2013). The Little Brown Bat (*Myotis nesopolus*) is also genetically close to the South American population (Larsen et al., 2012). The Funnel-eared Bat (*Natalus tumidirostri*s) also has a large range, although the Bonaire population is likely isolated (source: RELCOM). It is still unclear whether this applies to other bat species as well.

The natural range of the cave habitat can be assessed as favourable. However, the assessment for the Bonairean populations of the individual bat species could be less favourable.

Assessment of area: Favourable
Bonaire is filled with caves. For St. Eustatius and Saba, this is much less due to their volcanic origin. Many caves still need to be mapped. It is unlikely that many caves have been destroyed. Therefore, the surface area of caves is assessed as favourable.

Assessment of quality: Unfavourable-inadequate
Abiotic conditions: The abiotic conditions are unknown. The caves are vulnerable to disturbance, damage, and pollution, especially in the case of cave waters.

Typical species: There are indications that about four of the nine bat species may no longer occur on Bonaire. Whether this is indeed the case and whether landscape degradation is responsible will need to be determined through monitoring.

Other characteristics: The preservation of caves as habitats for bats and other fauna is primarily dependent on maintaining tranquillity and the absence of soil pollution and groundwater contamination from sewage and oil leakage.

Given the indications of a decline in bat species, the perceived increase in disturbance due to tourism, proximity to human habitation, and sewage discharge near some of the major cave systems, the quality is currently assessed as unfavourable-inadequate.

Assessment of future prospects: Unfavourable-inadequate
The future prospects for caves and cave fauna remain speculative, especially because cave fauna, such as the Long-tongued Bat (L*eptonycteris curasoae*), is part of a regional population that is not solely dependent on the caves in Bonaire. To protect the threatened *L. curasoae* (VU), caves outside Bonaire will also need protection (Simal et al., 2015).

Developments such as CARIBSS and projects like the proposed 'Bonaire Cave and Karst Reserve' are positive but do not yet address the existing threats. Further degradation of Bonaire's landscape is likely to result in less food for bats. An increase in tourism could lead to (the demand for) more recreational-touristic use and increased disturbance in the caves. The increase in Bonaire's population is likely to result in urbanization and possibly destruction and contamination of caves, with negative consequences for the endemic cave water fauna. Additionally, existing and potentially new wind turbine projects could lead to collision victims. The future prospects are currently assessed as moderately unfavourable.

Table 2. *Summary overview of the status of caves of Bonaire, Caribbean Netherlands, in terms of different conservations aspects.*

Aspect for Caves	**2024**
Distribution	Favourable
Area	Favourable
Quality	Unfavourable-inadequate
Future prospects	Unfavourable-inadequate
Overall Assessment of Conservation State	**Unfavourable-inadequate**

Comparison to the 2018 State of Nature Report

Overall, the CS of the caves of Bonaire has remained fairly unchanged compared to the 2018 assessment.

Recommendations for National Conservation Objectives

National long-term goals: Preservation of distribution and area and improvement of quality for the benefit of cave fauna.

National short-term (5 years) goals: Mapping the cave system and identifying key caves for bats according to the RELCOM strategy.

Key Threats and Management Implications

The strategic bat conservation program 2014-2018 (Simal, 2013) provides crucial input for determining management actions to protect caves as essential for the conservation of bats.

Table 3. *Overview of main threats to the caves of Bonaire, Caribbean Netherlands and implications for management.*

Main threats		**Management implications**
Disturbance	Disruption of roosting sites and maternity habitats	• Identify and Protect Important Bat Roosting Sites: Focus on identifying and safeguarding crucial roosting sites for bats, such as maternity caves, which are essential for their breeding and survival. • Develop a Cave Management Plan: Create a comprehensive management plan for caves that emphasizes sustainable educational and recreational tourism. This plan should balance the need for public access with the preservation of cave ecosystems. • Enforce Legislation to Prevent Habitat Loss: Implement and enforce laws and regulations to prevent habitat loss. This includes ensuring compliance with building permits and monitoring changes in land use that could impact cave environments. • Raise Awareness and Educate the Public: Provide information and education about the importance of caves to human communities and their ecological significance. This can help foster a greater appreciation for cave conservation and encourage responsible behaviour.

	Vandalism, graffiti	• Monitor Compliance with Regulations: Ensure that rules and regulations related to cave conservation are adhered to. This involves regular inspections and enforcement to uphold protective measures.
Pollution	Pollution and salinization of groundwater	• Identify and Protect Water Quality in Key Catchment Areas: Focus on identifying crucial water catchment areas linked to caves and implementing measures to protect and maintain their water quality.

Data Quality and Completeness

There is more unknown than known about the caves and cave fauna on Bonaire. A map of potential caves is available (Smith et al., 2012), and 100 caves have currently been mapped, but the actual entrances of the caves have not been published for various reasons (personal communication with F. Simal). CARIBSS is working on a cadastral-type database for the caves on Bonaire. There is virtually no information available about the ecology of the many endemic cave-dwelling crustaceans, and nothing is known about the vulnerability of cave waters to infiltration of anthropogenic water pollution. Additionally, the presence of fauna and the function of caves as maternity roosts and/or resting places need further investigation. This is particularly important due to the crucial role of nectar-feeding bats, such as *Leptonycteris curasoae* and *Glossophaga longirostris*, in the pollination of columnar cacti and thus in the ecology of Bonaire.

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6 Conservation State of the Salt Pans and Saline Lakes (Saliñas) of Bonaire

Van der Geest, M. and Debrot, A. O. 2025. From: State of Nature Report for the Caribbean Netherlands, 2024. WUR report C001/25.

International Protection Status

Various habitat types of salt marshes are protected under the European Habitat Directive, mainly because of their importance for mudflat and migratory birds. In the Caribbean, salt pans and salt lakes (salt marsh areas, also known as "saliñas") play a similar role. Some saliñas are designated as Ramsar sites, such as the saliñas of Washington Slagbaai National Park, as well as Pekelmeer on Bonaire. Many salt marshlands are also designated as Important Bird Areas (IBAs) by Birdlife International (Geelhoed et al., 2013).

Characteristics

Description
Saliñas are shallow, semi-enclosed bodies of saltwater mainly connected to drainage areas along the coast. These were largely formed at the end of the major ice age as shallow end stages of former inland bays (Boekschoten, 1982). On an annual basis, saliñas undergo strong fluctuations in salinity, ranging from nearly fresh to hypersaline conditions (Jongman et al., 2009). Plants and animals adapted to this environment include seagrass *Ruppia maritima* and various fish species belonging to the family of Cyprinodontidae (Kristensen, 1970), *Mullidae*, *Gerridae*, *Centropomidae*, *Albulidae*, and *Elopidae*. Saliñas also harbor important food sources for the Caribbean flamingo (*Phoenicopterus ruber*) such as brine shrimp, *Artemia* sp. (Kristensen and Hulscher-Emeis, 1972), snails *Cerithidae costata*, *Cerithium variabile*, and *Gemma purpurea* (de Boer, 1979), and the brine fly *Ephydra cinerea* (Rooth, 1965). Various studies provide background information on the ecology of saliña aquatic life (Stephensen, 1933; Koster, 1963; Kristensen, 1964, 1967, 1971; Versichele, 1984; Ecovision, 1996; Debrot and de Freitas, 1999). Strong salinity fluctuations mean that during dry periods, when the saliñas become hypersaline, the fish fauna dies. Subsequently, saliñas develop large densities of small food organisms, which serve as a food source for flamingos and other birds. Saliñas are the main breeding habitat for various ground-nesting seabirds, especially terns and plovers (Wells and Wells, 2006; Debrot et al., 2009). Presumably, these birds choose this type of breeding habitat because terrestrial predators are usually visible from a great distance. In general, it can be stated that the saliña is an example of a fluctuating saline and arid marsh area. Thanks to the highly variable and physiologically stressful conditions, saliñas form a unique niche for species resilient to large differences in salt concentration and capable of escaping aquatic predators and competitors physiologically unable to cope with high concentrations and fluctuations in salinity (Levinton, 1982). This allows such salt- and fluctuation-tolerant species to build up high population densities, which in turn serve as food for waterfowl (such as the flamingo). Saliñas also serve as nursery grounds (at lower salt concentrations) for certain fish species such as tarpon (*Megalops atlanticus*), white mullet (*Mugil liza*), and snook (*Centropumus undecimalus*) (Kristensen, 1964).

Relative importance in the Caribbean region
Saliñas in the Caribbean are concentrated around the Bahamas and the southern Caribbean, including Bonaire. The saliñas of Bonaire are of great international significance as breeding habitats for three

species of regionally threatened terns, namely the common tern (*Sterna hirundo*), the Cabot's tern (*Thalasseus acuflavidus*), and the least tern (*Sternula antillarum*) (Voous, 1983; Halewijn and Norton, 1984; Debrot et al., 2009). Additionally, they are of great importance as foraging areas for migrating and overwintering shorebirds (Voous, 1983; Prins et al., 2009; Debrot et al., 2014). Many of these salt marsh areas of Bonaire fall within the IBAs recognized by Birdlife International (Geelhoed et al., 2013).

Ecological aspects

Habitat: Saliñas are essentially low-lying flat drainage areas and are usually located close to the sea. They vary greatly in size, depth, physical parameters (such as salinity, water clarity, temperature, and nutrient content), and associated fauna (Kristensen 1967, 1970, 1971; Debrot, 2003). Often, they have been adapted to some extent as "salt pans" for salt extraction in the colonial past.

Environmental requirements: Saliñas can form in warm (tropical) flat areas near the sea. Because saliña's vary greatly under the influence of rainfall and evaporation, the abiotic conditions are also highly diverse. The strongly varying abiotic factors are the most important conditions for the formation of saliñas (Table 1).

Table 1. *Outline of the main abiotic conditions necessary for the formation of saliñas where orange and green reflect slightly suitable, very suitable condition, respectively.*

Salinity	Fresh	Slightly brackish	Brackish	Strongly brackish	Saline (33-38 ppt)	Highly saline (38-45 ppt)	Hypersaline (>45 ppt)

Nutrient richness	Oligotropic	Mesotrophic	Eutrophic	Hypereutrophic

Water clarity	Very turbid	Turbid	Slightly turbid	Clear	Very clear

Wave action	Low	Intermediate	High

Water depth	Deep (>2 m)	Shallow (1-2 m)	Lower intertidal zone	Mean sea level	Higher intertidal zone	Terrestrial

Acidity	Alkaline	Neutral	Slightly acidic	Slightly acidic	Acidic	Acidic

Typical species
Table 2 provides some species that may qualify as typical species for saliñas.

Table 2. *Typical species inhabiting saliñas in the Caribbean Netherlands.*

Common name	Scientific name	IUCN category	Taxa	Category[1]
Common tern	*Sterna hirundo*	LC	Birds	..
Least tern	*Sternula antillarum*	LC	Birds	Cb
Royal tern	*Thalasseus maximus*	LC	Birds	Cb
Caribbean Flamingo	*Phoenicopterus ruber*	LC	Birds	E
Brine shrimp	*Artemia salina*	DD	Crustaceans	E
Brine fly	*Ephydra cinerea*	DD	Insects	K
Sheepshead minnow	*Cyprinodon dearborni*	DD	Fish	Cb
Bonefish	*Albula vulpes*	NT	Fish	Cb
Ladyfish	*Elops saurus*	DD	Fish	Cb

[1]Typical species categories are as follows: Ca = constant species indicating good abiotic conditions; Cb = constant species indicating good biotic structure; Cab = constant species indicating both good abiotic conditions and good biotic structure; K = characteristic species; E = exclusive species.

Saliñas are surrounded by salt-tolerant and drought-resistant plant species, often characterized by fleshy leaves (e.g. *Sesuvium portulacastrum*, *Batis maritima*, *Salicornia perennis* (samphire)). Healthy and resilient saliñas provide important ecosystem services, such as:

- Stabilization of sediment and trapping of eroded topsoil (topsoil).
- Protection of coral reefs from sediment loading by trapping sediment particles before they reach the sea.
- Provisioning of nursery and breeding grounds for numerous (commercial) fish species, including the snook (*Centropomus undecimalis*).

Quality requirements of the environment:
For the function as a breeding area, protection against human disturbance is necessary. For other functions, protection against pollution of soil and groundwater from the hinterland drainage area is necessary. Overgrazing by free-roaming livestock, particularly goats, is a serious and persistent ecological problem on Bonaire (Neijenhuis et al., 2015; Lagerveld et al., 2015; Debrot, 2016). This causes extensive erosion and loss of soil nutrients (Vergeer, 2017), resulting in accelerated silting of this important habitat.

Current Occurrence and Reference Values

Within the Caribbean Netherlands, saliñas and saline lakes are only found on Bonaire (Jongman et al., 2009). They are in all flat parts of the island along the coast. The total area amounts to approximately 3,814 ha (Debrot et al., 2018). Reference values are unknown and difficult to define as saliñas are highly variable in space and time.

Assessment of National Conservation State

Trends in the Caribbean Netherlands
Ongoing erosion of sediment from degraded land results in silting and encroachment of saliñas, leading to surface loss and loss of ecosystem services provided by saliñas.

Recent developments
Urbanization around Kralendijk is causing the loss of saliñas in the area.

Assessment of distribution: Favourable
Salt marsh areas are a common habitat along the coast of Bonaire. They do not occur on St. Eustatius and Saba. Saliñas are also prevalent on Curaçao, Aruba, and Venezuela. The distribution of saliñas is considered Favourable.

Assessment of surface area: unfavourable-inadequate
Salt marsh areas are a common habitat along the coast of Bonaire. The largest area is in the low-lying southern part, which consists largely of managed salt pans for industrial salt production. This is the only part of Bonaire where salt pans are actively managed for salt production. In all other locations, salt mining has ceased, and the salt pans have to varying degrees reverted to a natural state. However, silting of saliñas and loss due to urbanization are reducing the surface area of unmanaged saliñas.

Assessment of quality: unfavourable-inadequate
Abiotic conditions: Erosion of sediment from degraded land leads to silting (Debrot et al., 2012). Eventually, a saliña becomes completely landlocked, resulting in the loss of all important ecosystem services as wetlands. As a drainage area, the salt marsh is not only vulnerable to silting, but also to the accumulation of anthropogenic pollution carried by surface and/or groundwater from inhabited areas. Except for Gotomeer, where serious industrial pollution has been demonstrated and proven (Slijkerman et al., 2013, de Vries et al., 2017), nothing is known about the pollution status of the island's saliña areas. In the plantation past, these areas were popular with plantation owners as shooting ranges and hunting grounds for migrating ducks. The potential accumulation of and pollution with lead (Pb) needs to be further investigated.

Typical species: Recreational disturbance is likely to have a negative impact on the breeding success of terns and the foraging and breeding success of the Caribbean flamingo. The presence of the flamingo is an important indicator of habitat quality.
Other characteristics: With the encroachment of a saliña, significant ecosystem services are lost, such as trapping of sediment, nutrients and pollutants. These, in turn, will have negative effects on downstream ecosystems, such as coral reefs. The extent to which this is currently occurring is unknown.

Assessment of future prospects: unfavourable-inadequate
Controlling the issue of free-roaming livestock has proven to be very difficult. Additionally, the urban development planning process has been stalled for years, and the Stichting Nationale Parken Nederlandse Antillen Bonaire (STINAPA-Bonaire) is inadequately equipped to control non-native predators. Furthermore, there is significant pressure from urbanization near Kralendijk on Saliña di Vlijt. Climate change will also influence saliñas through rising sea levels, altered rainfall regimes, and increasing temperature and subsequent evaporation (de Boer et al., 2023), but it is unclear what this will mean for the Environmental Quality Standards. Therefore, the future prospects for saliñas are assessed as unfavourable-inadequate (Table 3).

Table 3. *Overview of the status of saltpans and saline lakes of the Caribbean Netherlands for different ecological aspects.*

Aspect: Salt pans and saline lakes (saliña's)	**2024**
Distribution	Favourable
Surface area	Unfavourable-inadequate
Quality	Unfavourable-inadequate
Future prospects	Unfavourable-inadequate
Overall Assessment of Conservation State	**Unfavourable-inadequate**

Comparison to the 2018 State of Nature Report

Overall, the CS of the salt pans and saline lakes of Bonaire has remained fairly unchanged compared to the 2018 assessment.

Recommendations for National Conservation Objectives

National long-term goals
The target for achieving a favourable Conservation State for saltpans and saline lakes (saliñas) is preservation of the current distribution and area coverage, and the improvement of their quality.

National short-term (5-year) goals
Reduction of overgrazing by free-roaming livestock, management of human disturbance, such as recreational activities, control of invasive predators, particularly cats, and planning security against urbanization of the hinterland drainage areas.

Key Threats and Management Implications

Table 4. *Overview of the main threats to saltpans and saline lakes (saliñas) of the Caribbean Netherlands and implications for management.*

Main threats		**Management actions**
Climate change	Increased rainfall, higher temperatures, and rising sea levels will undoubtedly influence the low-lying saliñas. This can result in flooding (of nests) with fresh or saltwater on one hand and drying out on the other hand.	• Monitoring effects and implementing management measures based on that.
Pollution	Soil and water pollution are washed along with surface and groundwater and accumulate in the saliñas.	• Using urban planning to prevent extensive construction and industrial activity in the upstream drainage areas.
Coastal erosion	Land degradation, particularly due to overgrazing, leads to erosion and terrestrial sediment and nutrient input into the saliñas.	• Reduce overgrazing and implement active management of livestock. • Recover eroded topsoil from saliña areas to restore lost water areas. • Reduce coastal construction and industrial activity in upstream drainage areas of coastal bays and saliñas.
Disturbance	Disturbance of nesting and foraging birds due to uncontrolled recreational activities.	• Zoning and improved visitor management • Supervision and law enforcement (Debrot et al., 2009; Bertuol et al., 2015).
Invasive predators	Total or large-scale negative impact on the breeding success of ground-nesting birds such as tern colonies.	Culling of free-roaming invasive predators (especially cats, but also dogs and pigs).

Data Quality and Completeness

There have been very few studies conducted on the ecology of typical species, little is known about the use of saliñas by migrating shorebirds, and there is limited information about the conditions necessary for the healthy functioning of saliñas.

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7 Conservation State of the Beach Habitats of the Caribbean Netherlands

Henkens, R. J. H. G. and Debrot, A. O. 2025. From: State of Nature Report for the Caribbean Netherlands, 2024. WUR report C001/25.

International Conservation State

The beaches in the region are under significant pressure from climate change, tourism development, population growth, invasive species, pollution, and illegal sand extraction for commercial construction purposes. For native flora and fauna, the beach habitat can be considered highly threatened.

Characteristics

Description
In the Caribbean Netherlands, various types of beaches are present. On Bonaire (and Klein Bonaire), most beaches consist of coral rubble and smaller coral pebbles (primarily skeletal remains of the corals *Acropora palmata* and *A. cervicornis*), with fewer 'white' sand beaches. The beaches on St. Eustatius and Saba are also sandy but consist of dark volcanic sand. The beaches are generally only a few meters wide. Nevertheless, they serve as breeding and/or foraging habitats for various coastal birds such as terns and plovers. Above all, the sandy beaches of the Caribbean Netherlands serve as nesting grounds for various sea turtle species (see Table 1).

For the formation of beaches, waters should ideally be calm and there should be subtidal sand that accumulates in shallow depths to serve as a source of sand for the beach. Calm waters, and shallow accumulations of sand conducive to beach development are relatively rare in the Caribbean Netherlands. Beach development also often occurs where submerged valley systems discharge into the sea. These valley systems are the drainage routes for the hinterland. During the rainy season, small streams provide habitats for native amphidromous fish and shrimp (which spend their juvenile stages in the sea and estuaries but their adult stages in freshwater) (Debrot, 2003a, b). The valley systems are relatively moist, shaded, and sheltered, and together with the beach, they also serve as corridor areas for swarms of large land crabs (such as *Gecarcinus ruricola* and *G. lateralis*) and hermit crabs (*Coenobita clypeatus*) that live far inland but migrate annually from land to sea to reproduce (de Wilde, 1973).

Table 1. *Turtle species that use the sandy beaches of the Caribbean region and specifically the Caribbean Netherlands as nesting sites (Dow Piniak and Eckert, 2011).*

Common name	Scientific name	IUCN Red List status	Number of nesting beaches in the Caribbean (including Bermuda/Brazil)	Nesting beaches (N) and infrequent nesting beaches (IN)		
				Bonaire	St. Eustatius	Saba
Green Turtle	*Chelonia mydas*	EN	593	N	N	IN
Loggerhead Turtle	*Eretmochelys imbricata*	CR	817	N	N	IN
Hawksbill Turtle	*Caretta caretta*	LC	552	N	IN	
Leatherback Turtle	*Dermochelys coriacea*	VU	470	IN	N	

Relative Importance in the Caribbean

Data on the extent of sandy beaches in the greater Caribbean are not available, as far as known. The relative importance of sandy beaches can possibly be inferred from tourism developments. For instance, in Aruba, Curaçao, and St. Maarten, beach tourism is a significant source of income. On Bonaire, this is less pronounced, and on the volcanic beaches/small beaches of St. Eustatius and Saba, it is scarcely developed. Compared to other beaches in the Caribbean, the extent of the beaches in the Caribbean Netherlands is therefore limited. This likely also applies to the ecological function for sea turtles and coastal (breeding) birds, as evidenced by the hundreds of sea turtle nesting sites spread across the Caribbean region. Because sea turtles are listed as high conservation priority on the IUCN Red List, the importance of the rare nesting beaches should not be underestimated.

Ecological Aspects

Vegetation types

De Freitas et al. (2005, 2012, 2016) provide landscape-ecological vegetation maps for Bonaire, St. Eustatius, and Saba, which also include vegetation types of beach habitats. Saba essentially has no permanent beaches (De Freitas et al., 2016), which is why no beach vegetation type has developed there.

Most beach surface area is unvegetated, but different vegetation types can still be distinguished on beaches. De Freitas et al. (2005) identify three types of vegetated beaches on Bonaire:

- *Sesuvium-Lithophila* beach: with vegetation types S*esuvium-Lithophila* and *Lithophila – Euphorbia*;
- *Conocarpus* beach: with the vegetation type *Conocarpus*; and,
- *Lantana* beach: with vegetation types *Lantana – Capraria*, *Euphorbia – Sporobolus*, and S*esuvium – Lithophila*.

On St. Eustatius, the *Coccoloba* beach with the vegetation type *Coccoloba uvifera* is the only type of vegetated beach (De Freitas et al., 2012).

Table 2. *Vegetation types found on beaches in Bonaire and St. Eustatius.*

Vegetation types of beach habitat	
Bonaire	**St. Eustatius**
Sesuvium – Lithophila	*Coccoloba uvifera*
Conocarpus erecta	
Lithophila – Euphorbia	
Euphorbia – Sporobolus	
Lantana – Capraria	

Abiotic conditions

The primary conditions for the development of a beach are the availability of sand near the coast and moderate wave action (Table 3).

On none of the three islands of the Caribbean Netherlands is there much sand available in shallow waters, due to the steep bathymetry. Beaches therefore mainly occur where there are wider shallows near the coast. Additionally, moderate wave action is also essential. Excessive wave action quickly washes sand away to deeper areas, preventing the formation of a beach. Large parts of the coastlines of the islands, mainly the east sides, are entirely unsuitable for beach formation due to rough water. Conversely, insufficient wave action and therefore little water movement result in no sand transport to the coast. Ideally, a combination of turbulence in certain areas creates conditions for sand transport while calm conditions in other areas create conditions for sand deposition. Low wave action leads to sand deposition in the intertidal zone, while moderate wave action forms coarser pebble beaches. This results in significant variations in beach characteristics, including sand depth, stone content in the sand, and the development of beach rock, etc. (see e.g., Debrot and Pors, 1995). Laloë et al. (2016) and Esteban et al. (2018) demonstrate how properties such as the type and colour of sand on St. Eustatius can influence nest temperature and ultimately the sex ratio of sea turtles.

Table 3. *Outline of the key abiotic conditions for beach development.*

Availability sand	High	Moderate	Low	None
Wave action	High (no beach)	Moderate (pebble beach)	Low (sandy beach)	None (no beach)

Typical species

Table 4. *Typical species using beach habitats in the Caribbean Netherlands.*

Common name	Scientific name	IUCN category	Species group	Island
Green Turtle	*Chelonia mydas*	EN	Reptiles	B, E
Loggerhead Turtle	*Eretmochelys imbricata*	CR	Reptiles	B, E
Hawksbill Turtle	*Caretta caretta*	LC	Reptiles	B, E
Leatherback Turtle	*Dermochelys coriacea*	VU	Reptiles	B, E
Shoreline Sea Purslane	*Sesuvium portulacastrum*	DD	Plants (herb)	B, E, S
Bay Cedar	*Suriana maritima*	LC	Plants (herb)	B, E, S
Pygmy Blue	*Brephidium exilis*	LC	Butterfly	B

Other characteristics of a good structure and function
Beaches provide various ecosystem services. Firstly, they are of great tourist and economic importance to many Caribbean islands, although this is to a much lesser extent for the islands of the Caribbean Netherlands. However, beaches also have an important recreational function for the local population. Additionally, beaches serve as a form of coastal protection, as evidenced by the numerous coastal replenishment projects in the Netherlands to protect the hinterland from the sea.

Quality requirements for the environment
Quality requirements primarily concern species for which the beach serves as a growth area, nesting site, breeding, foraging, or resting place. The most important requirement is that the beach does not become inundated for these functions. Low levels of human disturbance, (oil) pollution and artificial lighting by which hatchlings will wander away from the sea, are key to beach quality.

Present Distribution

GIS analysis shows that Bonaire, St. Eustatius, and Saba have approximately 305 ha, 5 ha, and less than 1 ha of beach respectively. Most beach area is on Bonaire where important beaches are Donkey Beach, Klein Bonaire, Te Amo Beach, Bachelor's Beach, and the beach area in Lac Bay (especially Sorobon) which receives excessive numbers of tourist visitors (Debrot et al. 2012). On St. Eustatius the most important beach for nesting turtles is Zeelandia beach. On Saba Well's Bay Beach is mainly of touristic value.

Assessment of National Conservation State

Trends and recent developments
Long-term monitoring data are not available, so a clear trend cannot be determined. However, due to the exceedingly small surface areas occupied by the beach habitat and the high pressure on beaches for development, changes in this habitat can occur on a much shorter time frame than most other terrestrial habitats. Therefore, monitoring needs to be more frequent and vigilant than monitoring in most other habitat types. Since the last assessment for State of Nature reporting in 2018 (Henkens and Debrot, 2018), no major changes in the Conservation State of beaches can be discussed, in part due to lack of monitoring.

Assessment of distribution: Favourable
The beach habitat consists mainly of 'dead' materials such as sand and pebbles. Natural dispersal is not a relevant factor here.

Assessment of area: Unfavourable-inadequate
In general, the beaches of the Caribbean Netherlands are narrow, short, and have a relatively shallow layer of sand. This greatly limits their suitability as nesting sites for sea turtles. Additionally, an excess of natural “beach rock” in the water near the beach can make it unsuitable as a nesting site for turtles (Debrot and Pors, 1995).

A limited beach area, such as on Saba, is primarily due to the limited potential for beach formation. Coastal development can come at the expense of the beach. For example, on St. Eustatius, the wide sandy beach of Lower St. Eustatius has gradually become narrower due to the construction of the port and the resulting changes in currents (Hoogenboezem-Lanslots et al., 2010). Due to losses in the (recent) past, the current surface area is assessed as unfavourable-inadequate.

Assessment of quality: Unfavourable-inadequate

Abiotic Conditions: The current abiotic conditions appear favourable, but the quality of the beaches is strongly influenced by beach pollution and recreational pressure. Illegal sand mining is a well-documented issue on St. Eustatius and Bonaire. The oil industry is a significant economic sector and the risk of oil pollution is always present. Beaches are vulnerable to pollution from both litter and oil (Debrot et al., 2013). Pollution from *Sargassum* seaweed, likely due to climate change and eutrophication, has become a relatively new major problem for many Caribbean islands (CBC News, 2015; The Observers Direct, 2015; Mercopress, 2015; Van der Geest et al., 2024) that covers the sand and creates anoxic conditions, deadly to turtle nests. When beaches are cleared of washed up *Sargassum* seaweed, much sand is also removed.

Typical Species: Potential typical species primarily involve various types of sea turtles. These are relatively well monitored on the different islands. For Bonaire, there was a slight increase in the number of nests up to 2017 but since then there has been an apparent decline. There are several invasive beach plant species, such as Beach Naupaka, *Scaevola taccada*, which can be locally problematic.

Other Features: The beaches vary from white coral sand beaches to dark volcanic sand beaches, and beaches composed of coral rubble and/or pebbles.

Assessment of future prospects: Unfavourable-bad
The beaches in the Caribbean Netherlands are relatively narrow. A predicted sea-level rise of over half a meter due to climate change is expected to submerge most of the beaches due to the local beach structure, where landward migration is hardly possible (Cheetham, 2012). This will result in the loss of habitat for sea turtles, coastal birds, and other flora and fauna. Additionally, invasive species, pollution,

and the increase in tourism and population are expected to have a negative impact on the beach habitat. Therefore, the future prospects are considered as Unfavourable-bad.

Table 5. *Summary overview of the status of the beach habitat of the Caribbean Netherlands (Bonaire, Saba, St. Eustatius) in terms of different conservations aspects.*

Aspect of beaches	2024
Distribution	Favourable
Area	Unfavourable-inadequate
Quality	Unfavourable-inadequate
Future prospects	Unfavourable-bad
Overall Assessment of Conservation State	**Unfavourable-bad**

Comparison to the 2018 State of Nature Report

Overall, the CS of the beach habitats of the Caribbean Netherlands has remained fairly unchanged compared to the 2018 assessment.

Recommendations for National Conservation Objectives

National long-term goals
The target scenario for a favourable Conservation State is to maintain the current distribution and extent of (sand) beaches and to improve their quality.

National short-term (5 years) goals
Improving the quality should primarily be achieved by keeping visitor densities low on protected nesting beaches, preventing extraction for commercial construction purposes, and protecting (unprotected) sections of beaches with important ecological functions. Additionally, the beach should be cleaned of washed-up debris such as oil, eutrophication from leaching sewage, fishing gear, and (potentially) excess *Sargassum* seaweed (resulting from population explosions of this seaweed).

Key Threats and Management Implications

Table 6. *Overview of main threats to the beaches of the Caribbean Netherlands (Bonaire, Saba, St. Eustatius) and implications for management.*

Key threats		Management implications
Climate change	• The IPCC expects a sea level rise of 0.3-1.1 meters by the end of the century (Pathak et al., 2022). Without replenishment of sand or coral rubble, this means that the narrow beaches will "drown" and disappear. • Sargassum pollution of beaches has developed into a major annual problem for sea turtle nesting beaches in the Caribbean Netherlands (van der Geest et al. 2024). • The sex of a turtle is determined by the temperature in the nest. Higher temperatures result in more females. This effect, likely due to climate change, has already been observed in the Hawksbill turtle (Lolavar &	Coastal replenishment to keep up with sea-level rise is an option, but it must not come at the expense of other habitats such as coral reefs and seagrass beds. The need for beach cleanups to maintain beach quality has become structural.

	Wyneken, 2015). This could lead to a disrupted sex ratio with potentially negative effects on the population.	
Disturbance	• Many sandy beaches in the Caribbean are dominated by tourists. Tourism in the Caribbean is experiencing strong growth rates, with a recent average growth of 7%. The growth rates vary significantly between the islands, but they all generally show positive trends (UNWTO, 2016). Many tourists come for the sun, sea, and sandy beaches, making them a particular threat to turtle nesting sites. There are also plans to further develop beach tourism in the Caribbean Netherlands. • The population pressure is expected to increase in the coming decades (Hoogenboezem-Lanslots et al., 2010); CBS, 2023) which will also raise the pressure on the beaches.	Zoning of nesting beaches and important beach habitats for birds. This can also include temporary zoning (only during vulnerable periods, such as the breeding season).
Extraction/ mining	• Natural sand is a valuable construction material. The illegal extraction of sand is a well-documented issue on St. Eustatius and Bonaire.	• Awareness campaigns • Enforcement • Closing beaches to vehicles
Invasive Species	• Certain invasive plant species, such as *Vitex rotundifolia*, colonize beaches, making them unsuitable as nesting sites for sea turtles (Cousins et al., 2010) or as breeding habitats for coastal birds. On Bonaire, this problem is exemplified by the shrub *Scaevola taccada*.	• Removing invasive species before they can reproduce.
Pollution	• Beaches are vulnerable to pollution from litter and oil (Debrot et al., 2013). The oil industry is a significant economic sector in the Caribbean Netherlands, with Bopec on Bonaire and Nustar on St. Eustatius. Industrial accidents are not uncommon, but oil can also come from elsewhere, such as on May 25, 2017, when Bonaire's beaches were contaminated with oil and tar from Trinidad. • Pollution from *Sargassum* seaweed, likely due to climate change and eutrophication, is a major issue for many Caribbean (island) countries (CBC News, 2015; The Observers Direct, 2015; Mercopress, 2015).	Implementing the Maritime Emergency Response (RWS, 2013) for the Caribbean Netherlands, to clean up pollution as quickly as possible or to prevent it from reaching the beaches.

Data Quality and Completeness

There are hardly any measurements available for the physical parameters that determine beach quality in the Caribbean Netherlands. It is now important to implement a monitoring system for this purpose. On one hand, to establish long-term trends, and on the other hand, to evaluate the effects of any management measures. Monitoring of the size of beaches can be done using GIS tools.

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8 Conservation State of the Mangrove Forests of Bonaire

Van der Geest, M., Mücher, C. A., Gijsman, R. and Engel, S. 2025. From: State of Nature Report for the Caribbean Netherlands, 2024. WUR report C001/25.

International Protection Status

Within the EU, most coastal habitat types are protected under the European Habitat Directive, mainly due to their ecological importance for migratory shorebirds. The Caribbean Netherlands also contains various coastal habitats, including mangrove forests. An indication of the international protection status of mangrove forests within the Caribbean Netherlands can be derived from the WWF-classification for the Neotropical Ecoregion, which states: Critical/Endangered (WWF, 2017).

Within the Caribbean Netherlands, mangrove forests of considerable size occur only in the southeastern part of Bonaire, where they border the shores of Lac Bay. The entire Lac Bay lagoon and the mangrove forest it contains has been declared a protected Ramsar site, signifying its international importance in terms of natural characteristics (Debrot, Meesters and Slijkerman, 2010). The Pekelmeer in the south of Bonaire, which also contains some patches of mangrove forest, has also been declared a Ramsar site (Geelhoed et al., 2013). The mangrove forests of Bonaire are dominated by the red mangrove (*Rhizophora mangle*) and the black mangrove (*Avicennia germinans*), while the white mangrove (*Laguncularia racemosa*) and button mangrove (*Conocarpus erectus*) are also known to be occasionally present (Davaasuren and Meesters, 2012; Casal et al., 2024). These four species are listed in Annex III of the Specially Protected Areas and Wildlife (SPAW) protocol of the Cartagena Convention and are all protected under the Eilandsbesluit Natuurbeheer.

Characteristics

Description: Mangrove forests are evergreen forests that are formed by trees that have adapted to live in the warm intertidal areas of the world wherever waters are sufficiently calm and where there are sufficient sediments for them to take root (Leal and Spalding, 2022). These forests are found globally across the tropics and subtropics, where they grow in estuaries, deltas, lagoons and sheltered shores. Here, they provide important ecosystem services including enhanced biodiversity, coastal protection, mitigation of climate change through carbon sequestration, and provisioning of breeding and nursery grounds for many fisheries species (Nagelkerken et al., 2008; Leal and Spalding, 2022; Casal et al. 2024). Furthermore, they act as nutrient, sediment, and pollutant traps, thereby protecting adjacent habitats, such as seagrass beds and coral reefs, from these stressors.

The main global threat to mangrove forests is habitat loss due to aquaculture, agricultural production, coastal development for housing and tourism, and overexploitation (logging) (Polidoro et al., 2010). As a result, mangrove forests declined by 35 % worldwide between 1980 and 2000 (Millennium Ecosystem Assessment, 2005) and kept declining globally but at a slower rate of 3.4 % between 1996 and 2020 (Bunting et al., 2022). Compared to global trends, mangrove forests in the Caribbean have suffered even more damage over the last two decades, with a decline of 7.9 % between 1996 and 2020. This decline is mainly attributed to coastal development, demographic growth, and climate change (Bunting et al., 2022), and more recently also to the impact of massive influxes of holopelagic ("fully floating" lifestyle). *Sargassum* spp. brown algae (Chávez et al., 2020; Mücher and van der Geest, 2024). Work by Rull

(2023) suggests that if current rates of mangrove loss in the Caribbean continue, mangroves will go extinct from this region over the next three centuries. The effects of climate change on mangrove forests is less well understood. However, a recent review of the scientific literature on this topic by Trégarot et al. (2024) showed that altered rainfall regimes (i.e. reduced rainfall) and increased frequency and intensity of extreme weather events have the most negative impact on mangrove forest cover in the Caribbean.

In the Caribbean Netherlands, four species of mangroves are found: *R. mangle*, *A. germinans*, *L. racemosa* and *C. erectus*, of which the red mangrove *R. mangle* is the most common (Davaasuren and Meesters, 2012; Casal et al., 2024). Mangrove forests exhibit strong species-specific zonation patterns, which can be attributed to species-specific preferences regarding salinity, tidal flooding, and land elevation. These species-specific zonation patterns can be accurately distinguished and mapped using satellite spectral photography, as was shown for the mangrove forest in Lac Bay by Casal et al. (2024). *R. mangle* grows in frequently or permanently flooded areas near the seaward margins of the mangrove forest in Lac Bay, while *A. germinans* grows at higher elevation areas with low water content, as found in the northern backwaters of Lac Bay (Casal et al., 2024).

Relative importance within the Caribbean: In 2017, the total area of mangroves in the Lesser Antilles of the WWF Neotropical Ecoregion (ID: NT1416) was estimated at 20,636 hectares, spread over 263 different sites. The largest mangrove forests were found in Antigua and Barbuda, Guadeloupe, Martinique, and the US Virgin Islands (WWF, 2017). The total extent of mangrove forest on Bonaire is currently approximately 236 ha (Mücher and Verweij, 2020; Casal et al., 2024; Mücher and van der Geest, 2024), which is 1.1% of the total mangrove area in the Lesser Antilles that was estimated in 2017, thus making its relative importance within the Caribbean limited.

Ecological Aspects

Habitat: Mangrove habitat is characterized by a vegetation of evergreen mangrove trees that mainly occur in the intertidal zone, along sheltered and shallow-water coastlines in the (sub) tropics. Table 1 provides a more detailed overview of the environmental requirements for mangrove forests to persist.

Table 1. *Outline of the main abiotic conditions necessary for the development of mangrove forests where white, orange and green reflect unsuitable, slightly suitable, very suitable condition, respectively.*

Salinity	Very fresh	Moderately fresh	Slightly brackish	Moderately brackish	Strongly brackish	Saline (30-50 ppt)	Hypersaline (>50 ppt)

Temperature	<15 °C	15-20 °C	20-30 °C	30-37 °C	>37 °C

Water depth	Deep (>2 m)	Shallow (1-2 m)	Lower intertidal zone	Mean sea level	Higher intertidal zone	Terrestrial

Wave action	High	Intermediate	Low	None

The aerial roots of mangroves stabilise the seabed and provide a substratum on which many organisms depend. Above the water, the mangrove trees and canopy provide important habitat for a wide range of species, including birds, insects, reptiles and mammals (e.g. bats, primates). Below the water, the mangrove roots provide substrate for epibionts (e.g. tunicates, sponges, algae, bivalves), while the space between roots provides shelter and food for motile fauna such as prawns, crabs and fishes (Nagelkerken et al., 2008). Mangrove litter is transformed into detritus, which forms an important basis of the mangrove food web, in addition to plankton, epiphytic algae and microphytobenthos. Due to the high abundance of food and shelter, and low predation pressure, mangroves form an ideal breeding and nursery habitat for a variety of animal species, including (commercially important) crab, prawn and fish species (Nagelkerken et al. 2008). Mangrove forests are often also characterized by a dense concentration of mosquitoes, particularly the Crabhole Mosquito, *Deinocerites sp*. Other salt-tolerant and drought-resistant species found in mangrove areas include plants like *Sesuvium portulacastrum*, *Batis maritima*, and *Salicornia perennis* (Perennial Glasswort). Piscivores birds, such as herons, and insectivorous birds are often abundant, while mangrove forests are also known to provide important roosting sites for doves and parakeets (Harms and Eberhardt, 2003). Table 2 provides an overview of typical species that can be found in mangrove forests in the Caribbean Netherlands.

Table 2. *Typical species inhabiting mangrove forests in the Caribbean Netherlands.*

Common name	Scientific name	IUCN category	Taxa	Category[1]
Red mangrove	*Rhizophora mangle*	LC	Plants	Cab
Black mangrove	*Avicennia germinans*	LC	Plants	Cab
Yellow warbler	*Setophaga petechia*	LC	Birds	Cb
Green heron	*Butorides virescens*	LC	Birds	Cb
Grey snapper	*Lutjanus griseus*	DD	Fish	Cb
Rainbow parrotfish	*Scarus guacamaia*	NT	Fish	Cb

[1] Typical species categories are as follows: Ca = constant species indicating good abiotic conditions; Cb = constant species indicating good biotic structure; Cab = constant species indicating both good abiotic conditions and good biotic structure; K = characteristic species; E = exclusive species.

Ecosystem services provided by mangrove forest in the Caribbean Netherlands include climate regulation by carbon sequestration, provisioning of habitat and food for many species of fish, birds, marine mammals, and invertebrates, provisioning of honey, timber, fuel, and medical resources, nursery and breeding grounds for commercially fished species (e.g. grey snapper, great barracuda, and Caribbean spiny lobster), opportunities for ecotourism (e.g. guided canoe and/or bird watching tours), and trapping of land-based sediment, nutrients and pollutants, thereby protecting adjacent key habitats (i.e. seagrass beds, coral reefs), from these stressors.

Current Distribution and Reference Values

In the Caribbean Netherlands, mangrove forests are exclusively found on Bonaire, where they occur along the shores of Lac Bay, but occasionally also along the shores of Lagun, Pekelmeer and various salina’s. Based on the most recent literature, it is estimated that the total extent of mangrove forest on Bonaire is approximately 236 ha (value based on satellite image from 2014; Mücher and Verweij, 2020), of which 222.3 ha is located in Lac Bay (value based on satellite images from 2021 and 2022; Casal et al., 2024), 1.4 ha is located in Lagun (value based on satellite images from 2020; Mücher and van der Geest, 2024) and the remaining ~12.3 ha can be found along the shores of Pekelmeer and various salina’s in the south of Bonaire (Mücher and Verweij, 2020). Apart from the study by Mücher and Verweij (2020), there are no historical reference values for the total extent of mangrove forests on Bonaire, but they do exist for Lac Bay. Table 3 provides an overview of the extent of mangrove forest (ha) in Lac Bay reported for different years and shows that total mangrove cover in Lac Bay remained rather stable between 1961 and 1996 (range 238 - 239 ha) after which it has reduced from 239 ha in 1996 to 222.3 ha in 2021/2022.

Table 3. *Overview of the extent of mangrove forest (ha) in Lac Bay (Bonaire) reported for different years.*

Reference	Mapping year	Technique	Mangrove cover (ha) Lac Bay
Erdmann and Scheffers, 2006	1961	Aerial photo	239
Erdmann and Scheffers, 2006	1996	Aerial photo	238
Mücher and van der Geest, 2024	2014	Satellite (Pleiades)	221
Casal et al., 2024	2021/2022	Satellite (Sentinel-2)	222.3

Assessment of National Conservation State

Trends in the Caribbean Netherlands: Aerial and satellite maps of mangrove distribution in Lac Bay dating back to 1961, show largescale mangrove die-offs in the backwaters of Lac Bay. Simultaneously, the mangroves migrate seaward, and the lagoon becomes shallower due to both endogenous production and exogenous input of sediment (Wagenaar-Hummelinck and Roos, 1970; Erdmann and Scheffers, 2006; Debrot et al., 2019; Casal et al., 2024). Infilling is facilitated by mangroves that typically sequester most sediments on their landward margin and thereby are forced to migrate seawards. Over time, this process has resulted in isolated shallow hypersaline waters in the backlands of Lac Bay that used to be inhabited by mangroves, but are no longer suitable for mangrove growth and survival (Wagenaar-Hummelinck and Roos, 1970; Erdmann and Scheffers, 2006; Debrot et al., 2019; Mücher and van der Geest, 2024). Despite this ongoing process of seaward mangrove expansion and landward mangrove loss, the total extent of mangrove forest (ha) in Lac Bay remained rather stable between 1961 and 1996, ranging between 238 - 239 ha (Table 3).

Recent developments: In the past decades, the mangrove die-offs in the backwaters of Lac Bay outpaced the seaward expansion of mangroves, which has resulted in a 15.7 ha (6.6%) loss of mangrove habitat in Lac Bay between 1996 and 2021/2022 (Table 3). Part of this loss could also be attributed to the massive influxes of pelagic *Sargassum* seaweed that have intermittently invaded Lac Bay since 2018 (van der Geest et al., 2024). These influxes most likely caused mangrove die-offs at the seaward fringe of the forest, due to the anoxic conditions resulting from the large amount of accumulating and degrading *Sargassum* near the shore and may also have suppressed seaward expansion of the mangroves (van Tussenbroek et al., 2017; Mücher and van der Geest, 2024). In Lagun, 46.2% (1.2 ha) of the total area coverage in 2014 (i.e. 2.6 ha) was lost by 2020, which most likely could also be attributed to run-off related siltation of the backwaters of Lagun in combination with the direct impact of recent *Sargassum* influxes (Mücher and van der Geest, 2024).

To improve water circulation in the hypersaline backwaters of Lac Bay, where 15.4 ha of mangrove forest was lost between 2014 and 2020 (Mücher and van der Geest, 2024), a dedicated group of volunteers from Mangrove Maniacs started to restore historic tidal creeks (i.e., channels) and maintain existing tidal creeks, by removing excess sediment and mangrove regrowth. Recent field measurements show that the tidal connection between the lagoon and the backwaters of Lac Bay is still limited, but that the tidal creek restoration on average increased the tidal inflow volumes into the backwaters with about 12% (Gijsman et al., 2024). To reduce run-off related infilling of Lac Bay, budget has also been allocated to building water retention structures in the watershed of Lac Bay. In addition, Mangrove Maniacs has built mangrove nurseries, which have been used for small-scale mangrove outplant initiatives in degraded and coastal areas on Bonaire. Survival rates of planted mangrove seedlings varied between species and across sites but were overall low. Survival rates of red mangrove outplants were 27.5% and 29.6% in Lac Bay and the southwest coast of Bonaire respectively, while they were 8.4% for black mangrove outplants in the shore zones of Lagun (Haanskorf, 2024).

Assessment of distribution: unfavourable-inadequate
The total mangrove area in the Lesser Antilles in 2017 was estimated at 20,636 ha, spread over 263 different locations, which may seem favourable. However, in the Caribbean region, 7.9% of the mangrove forests have been lost between 1996 and 2020, which equals a loss rate of 0.32% per year (Bunting et al., 2022). Likewise, in Lac Bay, 6.6% (15.7 ha) of the of the total mangrove area coverage in 1996 (i.e. 238 ha) was lost by 2022, which equals a loss rate of 0.25% per year (Table 3). However, of the total mangrove area coverage in Lagun in 2014 (i.e. 2.6 ha), 46.2% (1.2 ha) was lost by 2020, which equals a very unfavourable loss rate of 6.6% per year (Mücher and van der Geest, 2024).

Assessment of surface area: unfavourable-inadequate
While the area coverage of mangroves in Lac Bay used to be rather constant at approximately 239 ha between 1961 and 1996, it is currently estimated to be approximately 222.3 ha, which is only 93% of its

former coverage (Table 3). Moreover, almost half of the mangrove area that was present in Lagun in 2014, was lost by 2020 (Mücher and van der Geest, 2024). These recent losses in mangrove cover can mainly be attributed to mangrove die-offs in the backwaters of Lac Bay and Lagun, due to ongoing siltation (Mücher and van der Geest, 2024). Although limited, *Sargassum*-induced mangrove die-offs at the seaward fringe of the forests most likely also played a role (Mücher and van der Geest, 2024).

Assessment of quality: unfavourable-inadequate
Abiotic Conditions: The relatively rapid siltation resulting from erosion and runoff of terrestrial sediment from the degraded and overgrazed hinterland is still the main cause of the loss of abiotic conditions suitable for mangrove growth and survival (Debrot et al., 2019; Mücher and van der Geest, 2024). In addition, the recent decomposition of large quantities of *Sargassum* at the seaward fringe of the mangrove forests also cause abiotic conditions that are detrimental to mangrove growth and survival (Mücher and van der Geest, 2024).

Typical species: Due to the siltation and influx of *Sargassum*, typical species for the mangrove forest are lost, such as the dominant red and black mangroves. The distribution of these species is thus an important indicator for the health of the forest. Using five Sentinel-2 images from 2021 and 2022, Casal et al. (2024) estimated the extent of mangrove forests in Lac Bay to be on average 222.3 ha, of which 136.0 ha were classified as red mangrove and 77.1 ha as black mangrove. The remaining unclassified mangrove area (~9 ha) most likely was dominated by white mangroves, although this needs validation in the field (Casal et al., 2024).

Productivity: Mean Net Primary Production (NPP) values in 2021/2022 were estimated to be 8.82 ± 1.46 ($g\ Cm^{-2}\ d^{-1}$), and showed a zonal distribution with highest values in the mid-West and East on the seaward side, and lowest values in the northern landward part of the mangrove forest of Lac Bay (Casal et al., 2024). Casal et al. (2024) also provided mean values for predicted Effective Leaf Area Index (LAI_e) in Lac Bay, which ranged from 3.37 to 3.85, with significantly higher values in the wet season (3.82 ± 0.57) compared to the dry season (3.40 ± 0.56). This suggest that a decrease in annual precipitation as predicted by Taylor et al. (2020), most likely will have a negative impact on mangrove productivity in Lac Bay.

Other characteristics: When investigating changes in carbon storage dynamics in gradually degrading mangrove forest of Lac Bay, Senger et al. (2021) calculated a loss of 1.51 $MgCO_2\ ha^{-1}\ yr^{-1}$ for degraded mangrove sites compared to intact mangrove sites. This illustrates that, as the mangrove forest area becomes silted up, it will lose all characteristic features of a mangrove forest, including its ecosystem services, and ecological values.

Assessment of future prospects: unfavourable-inadequate
The most substantial threat to the mangroves forest is siltation due to natural and human-induced erosion of terrestrial sediments, that in turn can cause hypersaline conditions in the backwaters of Lac Bay. The restoration of historic tidal creeks is a promising intervention to increase the tidal connection between the lagoon and the backwater of Lac Bay, and reduce the threat of hypersaline conditions, yet it does not reduce the erosion of terrestrial sediments (Gijsman et al., 2024). Additional threats to the quality of the mangroves are not only the recent coastal *Sargassum* influxes (van der Geest et al., 2024; Mücher and van der Geest, 2024), but also recreation (Debrot et al., 2012), pollution, eutrophication and litter (Slijkerman et al., 2011; Debrot et al., 2013). The effects of climate change on the mangrove forests are unclear. A decrease in annual precipitation, as predicted by Taylor et al. (2020), will most likely negatively impact mangrove productivity, while an increase in extreme events will most likely increase erosion of terrestrial sediments. Even though the decline of the mangrove forest continues steadily, the fact that restoration is technically possible and likely cost-effective provides perspective.

Table 4. *Overview of the status of mangrove forests of Bonaire for different ecological aspects.*

Aspect mangrove forest	**2024**
Distribution	Unfavourable-inadequate
Surface area	Unfavourable-inadequate
Habitat quality	Unfavourable-inadequate
Future prospects	Unfavourable-inadequate
Overall Assessment of Conservation State	**Unfavourable-inadequate**

Comparison to the 2018 State of Nature Report

Overall, the current state of the mangrove forest of Bonaire has slightly worsened compared to the 2018 assessment, especially due to siltation and reoccurring coastal *Sargassum* influxes.

Recommendations for National Conservation Objectives

Long Term Goals: The goal for a favourable CS is the preservation of the distribution and area, and the improvement of the quality of the mangrove forest in Lac Bay and Lagun.

Short-term (5 years) goals: Improving the quality mainly involves restoring water depth and circulation in the already filled-in parts of Lac Bay and Lagun, by removing accumulated sediment, by strategic placement of booms to prevent *Sargassum* influxes into the mangrove forest, and by opening existing and historic tidal creeks. In addition to a) carrying out pilot interventions in this regard and monitoring their effects, other priorities include: b) reducing overgrazing by free-roaming livestock; c) controlling human disturbance; d) reforestation of overgrazed hinterland with native trees, e) building and maintenance of water retention structures in the hinterland, and f) legislation to protect the hinterland drainage area of Lac Bay against urbanization, to limit the inflow of toxins, nutrients, and pathogens via surface and groundwater.

Key Threats and Management Implications

Table 5. *Overview of the main threats to the mangrove forests of Bonaire and implications for management.*

Key threat	**Consequence**	**Management intervention**
Infilling	Reduced water circulation resulting in hypoxia, hyper-salinity, and elevated temperatures during dry season	• Reduce overgrazing and actively manage livestock in hinterland. • Replant hinterland • Build and maintain water retention structures in hinterland • Reclaim eroded topsoil from silted mangrove areas to restore lost water areas. • Cut open and maintain channels
Sargassum influxes	Reduced water quality due to degrading *Sargassum* biomass causing hypoxia and sulfide levels that are toxic to all marine life.	• Strategic placement of oil booms to prevent *Sargassum* from invading the mangrove forest. • Timely clearing of *Sargassum* that accumulates behind oil booms during *Sargassum* influx event
Pollution	Soil and water pollution is washed away with surface and bottom water and accumulates in the mangrove areas	• Urban planning to prevent excessive development and industrial activity in upstream drainage areas • Remediation of landfill area upstream of Lagun
Disturbance	Disturbance of resting and foraging birds by uncontrolled recreational activity	• Zoning and improved visitor management • Surveillance and law enforcement

Data Quality and Completeness

There is sufficient knowledge on the distribution, coverage, species composition, ecological condition and stressors of the mangrove forests in the Caribbean Netherlands. This allows for testing and evaluating the effectiveness of mangrove conservation and restoration measures. Continued monitoring of these mangrove parameters at a 3-year interval, will provide sufficient insight into the trend, while it will also allow for the testing and evaluation of future mangrove conservation and restoration interventions.

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9 Conservation State of Seagrass and Macroalgal Fields of the Caribbean Netherlands

Van der Geest, M. and Engel, S. 2025. From: State of Nature Report for the Caribbean Netherlands, 2024. WUR report C001/25.

International Protection Status

A total of seven Caribbean seagrass species are listed in Annex III of the Specially Protected Areas and Wildlife (SPAW) protocol of the Cartagena Convention. This means that parties must adopt appropriate measures to ensure protection and recovery of these seagrass species. Yet, this is not the case for macroalgal species, of which none are listed in Annex III of the SPAW. Within the Caribbean Netherlands, the occurrence of 6 native seagrass species has been reported (i.e. *Thalassia testudinum*, *Syringodium filiforme*, *Halodule wrightii*, *Halophila decipiens*, *Halophila baillonii*, *Ruppia maritima*), five of which are listed as Least Concern (LC) on the IUCN Red List, and one, namely *H. bailloniii*, which is listed as vulnerable (VU) (Short et al., 2011; Willette et al., 2014). From these native species, *T. testudinum* and *S. filiforme* are the only two species that are also protected in Bonaire under the Eilandsbesluit Natuurbeheer Bonaire. Since 2010, the invasive species *Halophila stipulacea* has also been spreading in the Caribbean Netherlands.

Characteristics

Description: Seagrass beds are found in shallow, nutrient-poor coastal waters around the world, from the tropics to sub-polar areas. Evolutionarily, seagrasses are land plants that have adapted to underwater life. Many characteristics of land plants have been retained, including roots to extract nutrients from the soil, flowers, pollen, and seeds. Seagrass beds provide food and shelter for a diverse range of marine life, including invertebrates, molluscs, fish, reptiles, and mammals. In the Caribbean Netherlands, this is reflected by their importance as nursery and/or feeding grounds for various species of Snappers (*Lutjanidae*), Parrotfish (*Scaridae*), Surgeonfish (*Acanthuridae*), Sea breams (*Sparidae*) as well as the Queen Conch (*Lobatus gigas*), and Green sea turtle (*Chelonia mydas*). Seagrasses also provide coastal protection services, as their root systems stabilize sediments and their leaves attenuate wave energy (Orth et al., 2006; James et al., 2019). In addition, seagrass beds act as nutrient, sediment, and pollutant traps, thereby protecting adjacent habitats, such as coral reefs, from these stressors. Seagrass beds also play a vital role in carbon sequestration, as they store large amounts of carbon in their leaves and roots and in the underlying sediment (Duarte et al., 2005).

Despite their ecological and economic importance, seagrass beds are rapidly declining worldwide due to the immediate impacts of coastal development, demographic growth, and the impact of ecological degradation and climate change (Orth et al., 2006; Waycott et al., 2009). As such, there is a growing need to protect and conserve seagrass beds. This is particularly true for seagrass beds in the Caribbean, which have also been suffering from the impact of massive influxes of holopelagic (is fully planktonic) *Sargassum* spp., brown algae, since 2011 (Wang et al., 2019). Close to the shoreline, the decomposition of these algae produces leachates and organic particles, resulting in murky brown waters known as *Sargassum* Brown Tides (SBT). These tides lead to decreased light penetration, oxygen levels, pH, and overall water quality, which eventually results in seagrass mortality (van Tussenbroek et al., 2017). Regarding the impacts of climate change, a recent review of the scientific literature on this topic by Trégarot et al. (2024) showed that warming (and associated increased salinity) and increased frequency and intensity of extreme weather events (i.e. heat waves, storms, hurricanes) have the most negative impact on seagrass beds in the Caribbean.

Canopy-forming benthic macroalgae provide a productive and unique habitat for a diversity of organisms, and recognition for their role in the provision of food, shelter and nursery habitat for fishes and support of local fisheries has recently grown (Eggertsen et al., 2017; Tano et al., 2017; Fulton et al., 2020). Despite their ecological economic importance, macroalgal fields are threatened by the immediate impacts of coastal development, water quality degradation and global climate change (Walker and Kendrick, 1998; Fulton et al., 2019). Like seagrasses, macroalgae need sufficient light for photosynthesis, and therefore they are also restricted to shallow coastal waters.

In the Caribbean Netherlands, seagrass beds are often mixed with macroalgae, especially with green macroalgae of the genus *Halimeda* and *Caulerpa*. Macroalgae fields generally consist of brown algae of the genus *Sargassum* sometimes mixed with brown algae of the genus *Turbinaria*. These *Sargassum* fields are found along the entire east coast of Bonaire at depths between 5 and 20 m (Kemenes van Uden et al., 2024). Due to the strong water currents in this zone, corals struggle to settle and grow, resulting in the seabed being covered by dense mats, mainly of *Sargassum polyceratium* (Bak, 1975). This dominant species has been subject of many studies on productivity (Wanders, 1976a, b), biomass (de Ruyter van Steveninck and Breeman, 1981), population dynamics (Wanders, 1977; de Ruyter van Steveninck and Breeman, 1987a, b; Engelen et al., 2005a, b), and genetics (Engelen et al., 2001). These *Sargassum*-dominated fields appear to play a special role in the coral reef system as feeding and nursery areas for coral reef fish species (Chaves et al., 2013).

The Saba Bank also has extensive macroalgae fields, which have recently been estimated to range between approximately 656 and 807 km² depending on which technique was applied on the dataset (Meesters et al., 2024). These macroalgae fields have a different structure and composition than the *Sargassum*-dominated fields of Bonaire (Toller et al., 2010). For example, on the Saba Bank, there are green algae fields, brown algae fields (which include *Sargassum* and *Lobophora* fields), and red algae fields, all with a high species richness (Littler et al., 2010; Meesters et al., 2024).

Relative Importance Within the Caribbean

Seagrass beds limited, seaweed fields large.

The distribution of seagrasses on a global scale has been divided into six bioregions (Short et al., 2007). The Caribbean Sea, along with the Gulf of Mexico, Bermuda, and the two tropical coasts of the Atlantic Ocean, belongs to the Tropical Atlantic bioregion (Fig. 1). Within this region, 8 native species of seagrasses occur, namely *Thalassia testudinum*, *Syringodium filiforme*, *Halodule wrightii*, *Halophila baillonii*, *Halophila decipiens*, *Halophila engelmannii*, *Ruppia maritima*, and the endemic *Halodule bermudensis* (endemic to Bermuda), and 2 invasive exotic species, namely *Halophila ovalis* subs. *ovalis* and *Halophila stipulacea*.

All species are found in the Caribbean Netherlands, except for native *H*. *engelmannii*, endemic *H*. *bermudensis* and invasive *H*. *ovalis* subs. *ovalis*. The invasive *H. stipulacea* is encountered on both the Leeward and Windward Islands. The main seagrass beds in the Caribbean Netherlands are in Lac Bay on Bonaire and around St. Eustatius. Small seagrass beds of *H*. *decipiens* have been occasionally found on Saba (Debrot et al., 2018). Yet, seagrass beds have never been observed on the Saba Bank (Meesters et al., 2024). Given the limited extent of seagrass beds in the Caribbean Netherlands, their relative importance for the whole Caribbean is limited.

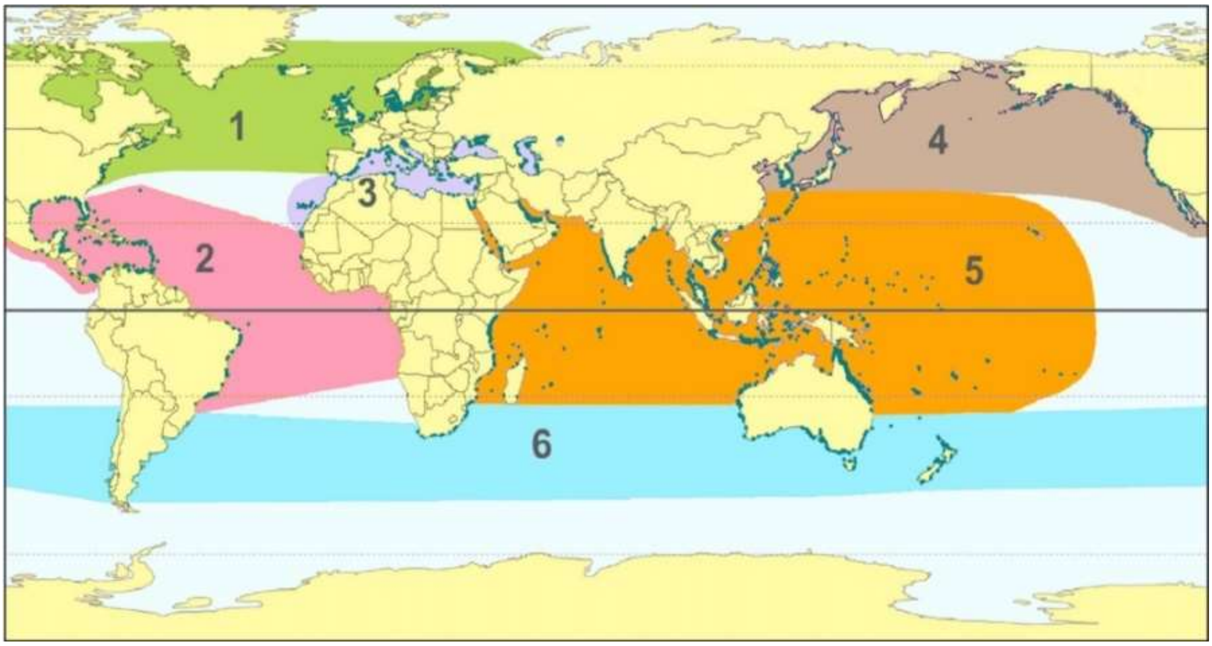

Figure 1. *Distribution of seagrasses on a global scale (blue dots and polygons; data from 2005 UNEP-WCMC) and geographic bioregions. 1. Temperate, 2. Tropical Atlantic, 3. Mediterranean, 4. Temperate North Pacific, 5. Tropical Indo-Pacific, 6. Temperate Southern Ocean. (Short et al., 2007).*

The algal fields on the Saba Bank, according to Littler et al. (2010), encompass "previously unknown unique algal communities." They consider the Saba Bank to be the most diverse area for seaweeds in the Caribbean and suggest that "Habitats on the Saba Bank far surpass species diversity per unit sampling effort". Due to the expanse of these species-rich algal fields on the Saba Bank, their relative importance within the Caribbean is high.

Ecological Aspects

Habitat: Within the Tropical Atlantic bioregion, seagrass beds are found in lagoons, shallow coastal zones, near coral reefs, and in deeper coastal zones down to 50 m and more (Fig. 2). Seagrass beds are dominated by three species: *Thalassia testudinum*, *Syringodium filiforme*, and *Halodule wrightii*. These can occur as monotypic vegetation but usually represent a successional stage with a diverse species composition (Creed et al., 2003). *T. testudinum* is the most common species in the region. *S. filiforme* has a similar distribution and usually grows together with *T. testudinum*, but can also form monospecific seagrass beds from the upper sublittoral to 20 m deep. *H. wrightii* can be found throughout the Caribbean on sandy and muddy bottoms from the intertidal zone to 3 m depth. *Ruppia maritima* can also be found throughout the Caribbean, where it inhabits shallow (mostly) brackish waters in bays, saliña and estuaries between 0 and 4 m deep. This opportunistic species is occasionally abundant in the backwaters of Lac Bay (Bonaire) and has also been observed in small patches in Saliña Matijs (Bonaire) (M. van der Geest, pers. obs. (2021). The seeds of *R. maritima* are an important food source for the Caribbean flamingo (*Phoenicopterus ruber*). The two native species belonging to the genus *Halophila* (*H. baillonii*, and *H. decipiens*) are relatively small and delicate. *H. decipiens* occurs in deep water, up to 85 m, while *H. baillonii* is only found down to 15 meters depth (see Table 1). *H. baillonii* is either very rare or possibly no longer present in the Caribbean Netherlands.

In various locations, including in the Caribbean Netherlands, the invasive *Halophila stipulacea* is becoming increasingly dominant (Debrot et al., 2014; Willette et al., 2014; Christianen et al., 2019; Engel, 2024). This species can occur up to a depth of 65 m.

Table 1. *Potential and observed seagrass species in the Caribbean Netherlands (Hoeksema (2016) and pers. comm. S. Engel and A.O. Debrot).*

Common name	Scientific name	Observed in the Caribbean Netherlands			IUCN category	Max. depth (m)
		Bonaire	St. Eustatius	Saba		
Widgeon Grass	*Ruppia maritima*	X			LC	4
Shoal Grass	*Halodule wrightii*	X			LC	3
Star Grass	*Halophila engelmanni*				VU	60
Turtle Grass	*Thalassia testudinum*	X			LC	30
Manatee Grass	*Syringodium filiforme*	X	X	X	LC	20
Clover Grass	*Halophila baillonii*				VU	15
Paddle Grass	*Halophila decipiens*		X	X	LC	85
-	*Halophila stipulacea*	X	X	X	LC	65

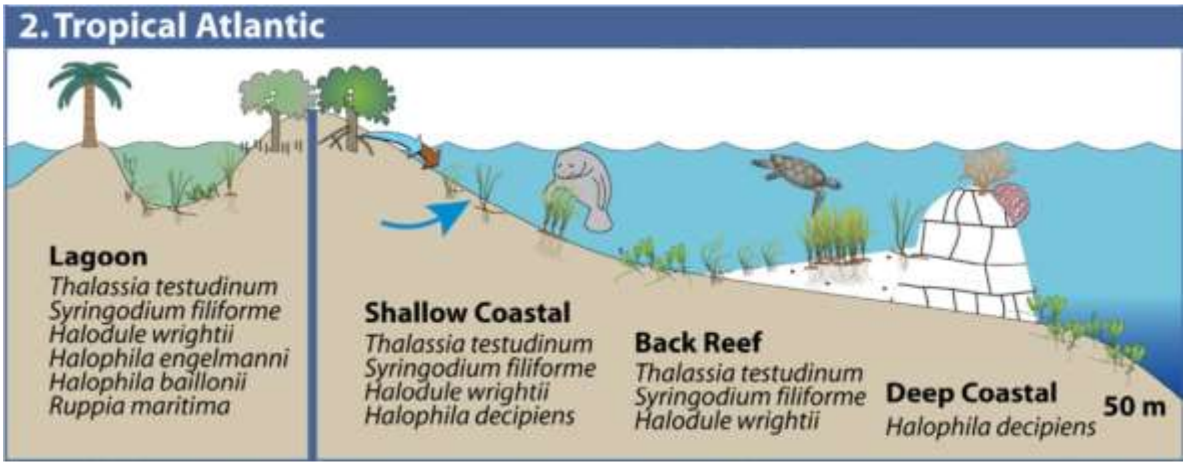

Figure 2. *Seagrass habitat diagram for the Tropical Atlantic bioregion. The main species are ranked according to dominance within the respective habitat (Short et al., 2007).*

The macroalgal fields on the Saba Bank are characterized by a high diversity. Based on an initial exploration in 2006, it is estimated that there are 150-200 algal species present on the Saba Bank, of which 98 were found in an initial brief survey, with possibly a dozen brown algal species that are new to science (Littler et al., 2010). The different communities are characterized by a very rich species composition and dominance of a specific seaweed group. There are brown algal communities dominated by large brown algae (Phaeophyceae, 26 species found), including *Dictyopteris* and a variety of *Sargassum*, *Lobophora* and *Dictyota* species. There are also green algal communities with many green algae (Chlorophyta, 26 species found), including many "rooting" Bryopsidales species characteristic of healthy seagrass beds in sedimentary habitats (*Penicillus, Udotea, Codium,* and *Caulerpa* species). However, seagrass is absent on the Saba Bank. Additionally, red algal communities have been found with a wide variety of spectacular fleshy species (Rhodophyta, 43 species found, including various *Dasya*, *Gracillaria*, and *Laurencia* species) (Littler et al., 2010).

Environmental requirements: Tables 2 and 3 provide a more detailed overview of the environmental requirements for the development of seagrass beds and macroalgal fields, respectively.

Table 2. *Outline of the main abiotic conditions necessary for the development of seagrass beds where white, orange and green reflect unsuitable, slightly suitable, very suitable condition, respectively.*

Salinity	Fresh	Slightly brackish	Brackish	Strongly brackish	Saline (33-38 ppt)	Highly saline (38-45 ppt)	Hypersaline (>45 ppt)
Water temperature		<15 °C	15-23 °C	23-31 °C		31-37 °C	>37 °C
Nutrient richness		Oligotrophic		Mesotrophic		Eutrophic	Hypereutrophic
Water clarity		Very turbid	Turbid	Slightly turbid		Clear	Very clear
Wave action		Low		Intermediate			High

Table 3. *Outline of the main abiotic conditions necessary for the development of macroalgal fields where white, orange and green reflect unsuitable, slightly suitable, very suitable condition, respectively.*

Salinity	Fresh	Slightly brackish	Brackish	Strongly brackish	Saline (33-38 ppt)	Highly saline (38-45 ppt)	Hypersaline (>45 ppt)
Water temperature		<15 °C	15-23 °C	23-31 °C		31-37 °C	>37 °C
Nutrient richness		Oligotrophic	Mesotrophic		Eutrophic	Hypereutrophic	
Water clarity		Very turbid	Turbid	Slightly turbid		Clear	Very clear
Wave action		Low	Intermediate			High	

Typical species

Table 4 provides an overview of typical species that can be found in healthy seagrass beds in the Caribbean Netherlands. Vegetation structure, complexity and low epiphyte cover also serve as good indicators of a healthy seagrass bed.

Table 4. *Typical species inhabiting healthy seagrass beds in the Caribbean Netherlands.*

Common name	**Scientific name**	**IUCN category**	**Taxa**	**Category[1]**
Antillean manatee	*Trichechus manatus manatus*	EN	Marine mammals	E
Green sea turtle	*Chelonia mydas*	EN	Sea turtles	K
Queen conch	*Lobatus gigas*	-	Gastropods	K
Caribbean spiny lobster	*Panulirus argus*	-	Crustaceans	Cb
Turtle Grass	*Thalassia testudinum*	-	Plants	Cab
Green sea urchin	*Lytechinus variegatus*	-	Sea urchin	Cab
Red cushion sea star	*Oreaster reticulatus*	-	Sea star	Cab

[1] Typical species categories are as follows: Ca = constant species indicating good abiotic conditions; Cb = constant species indicating good biotic structure; Cab = constant species indicating both good abiotic conditions and good biotic structure; K = characteristic species; E = exclusive species.

Being a climax species, high coverage of native *T. testudinum*, is also an important indicator for resilient seagrass beds with good structure and function. Such healthy and resilient seagrass beds provide important ecosystem services, such as:

- Stabilization of sediment and prevention of coastal erosion (James et al., 2019).
- Trapping of land-based sediment, nutrients and pollutants, thereby protecting adjacent coral reefs from these stressors.
- Habitat and nursery grounds for many
- Sequestration of carbon in soil and plants.
- Production of biocides, filtration of pathogens (Lamb et al., 2017).

Low hydrodynamics and clear nutrient-poor waters are important conditions for the development of healthy and resilient seagrass beds. This means that coastal erosion-related runoff of sediment and nutrients must be prevented, in addition to massive influxes of *Sargassum* spp. brown algae. Van Tussenbroek et al. (2016) found that the invasive species *H. stipulacea* forms dense "mats" under eutrophic conditions that prevents other species propagating themselves. As ecosystem services like fish nursery function, coastal protection, and carbon sequestration provided by small-leaved and shallow-rooting *H. stipulacea*-dominated seagrass beds are less than those provided by native large-leaved and deep-rooting *T. testudinum*-dominated seagrass beds (Becking et al., 2014; van Tussenbroek et al., 2016; Smulders et al., 2017; James et al., 2020), it seems even more important to limit coastal eutrophication. Physical damage from activities such as recreation should also be avoided to maintain optimal seagrass coverage.

Regarding the macroalgal fields on the Saba Bank, there is still insufficient knowledge, although there are indications of a positive relationship with nutrient-poor, undisturbed conditions.

Current Distribution and Reference Values

Seagrass beds: On Bonaire, the most extensive seagrass beds are found in Lac Bay, in addition to some small seagrass patches in the saltpan area South of Lac Bay. While seagrass beds used to be present in Lagun before the first massive *Sargassum* influx in 2015 (S. Engel, pers. comm.), in 2021 only some small patches of surviving *T. testudinum* and *H. wrightii* were observed nearshore at depths < 0.5 m, which in total covered no more than 0.1 ha (Fig. 3, M. van der Geest, pers. obs. (2021)). Apart from suffocation due to recent influxes of *Sargassum*, this decline in seagrass beds in Lagun could also be attributed to chemical pollution of the sediment and seawater due to spill-over effects from the nearby upstream landfill (Dogruer et al., 2024). This is also reflected by *T. testudinum* leaf tissues containing significantly higher concentrations of heavy metals (i.e. Cd, Co, Mn, Ni, Zn, As) in Lagun than in other bays on Bonaire (Ouwersloot, 2022).

In March-May 2022, seagrass cover was determined in Lac Bay by visual inspection of species-specific seagrass cover inside 0.5 m^2 quadrats that were placed at 49 different locations that covered the whole open water area of Lac Bay (see Fig. 3). This resulted in a mean total seagrass cover of 59.8% of invasive *H. stipulacea* and for 24.0% (SD = 35.4), 1.9 % and 1.7% of native *T. testudinum, H. wrightii* and *S. filiforme*, respectively. Based on the observed 59.8% seagrass cover, and the knowledge that the open water area in Lac Bay covers 355 ha, it is estimated that Lac Bay contained 213 ha of seagrass beds in 2022 (see Fig. 4). Assuming an additional 2 ha of seagrass beds outside of Lac Bay, the total area of seagrass beds on Bonaire is estimated to be 215 ha.

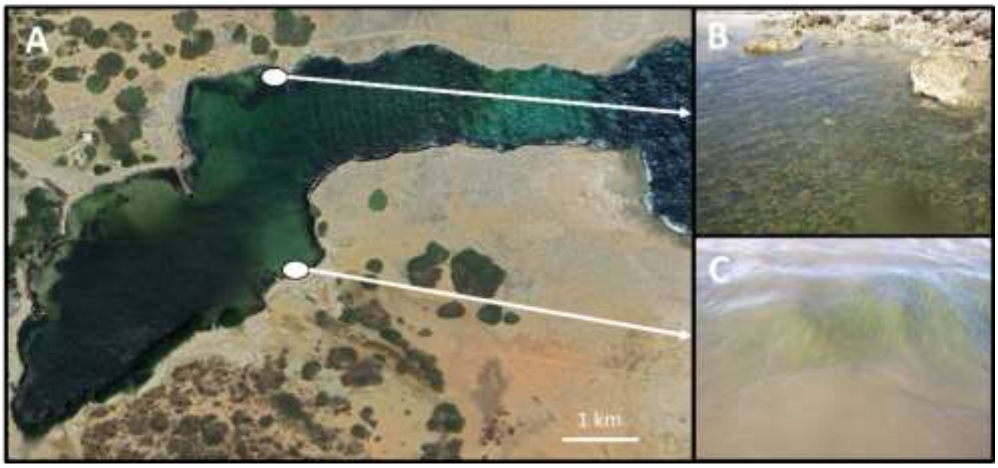

Figure 3. *Overview of Lagun area (A), with the location of a seagrass patch of Thalassia testudinum (B) and Halodule wrightii (C) (photos: M. van der Geest, March 2021).*

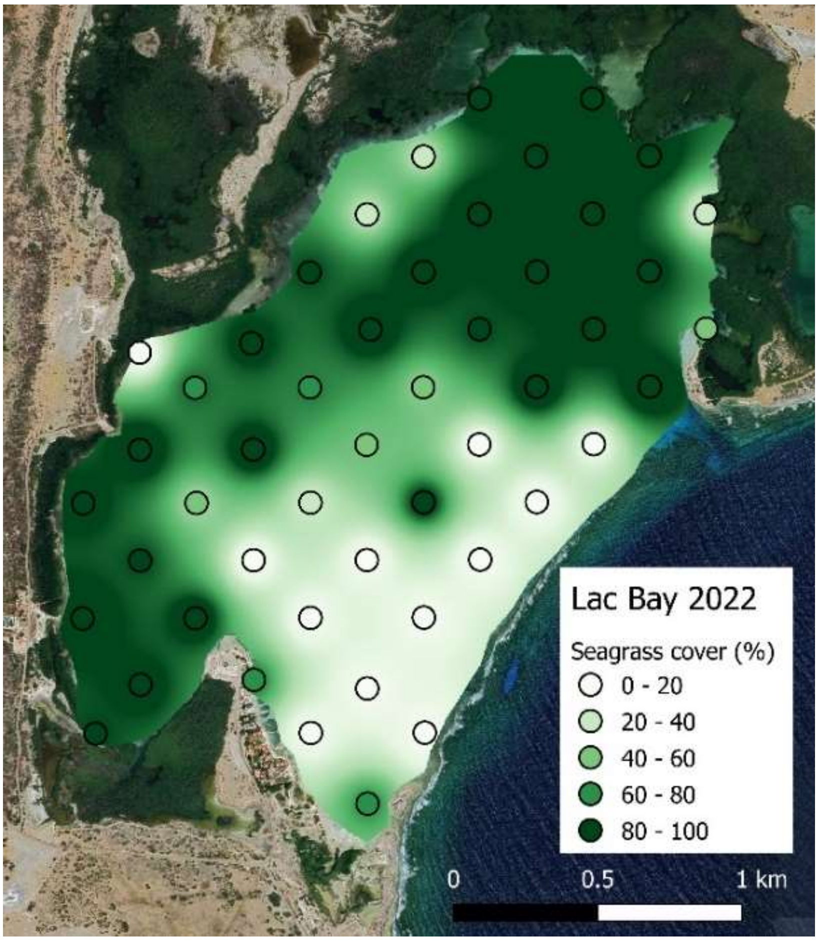

Figure 4. *Total seagrass cover (%) at 49 sites in Lac Bay in March-May 2022 and predicted seagrass cover based on the data collected at the 49 sites (image was made using QGIS 3.16 'Hannover').*

On St. Eustatius, seagrass beds are mainly found in the northern coastal waters at a mean depth of 24.2 m, where they cover an area of 124 ha (Debrot et al., 2014). Here, two different seagrass beds were distinguished by Debrot et al. (2014): dense seagrass beds dominated by the invasive *Halophila stipulacea* (between 45-95% cover), and sparse seagrass beds dominated by the native *H. decipiens* (between 8-25% cover). A third seagrass species was *Syringodium filiforme,* which was only found at densities of 2% or less (Debrot et al. 2014). Seagrass beds of native *Thalassia testudinum*, reported as being important in St. Eustatius by MacRae and Esteban (2007), no longer existed in 2012 and 2013, according to Debrot et al. (2014).

Saba lacks *T. testudinum* seagrass beds, because of its exposed coasts (Buchan, 1998). However, patches of seagrass are sporadically found in the coastal waters of Saba, where they cover up to 20 ha (Kuramee and van Rouendaal, 2013; Debrot et al., 2018). These patches mainly consist of *S. filiforme* (Buchan, 1998) and since 2019 also of invasive *H. stipulacea* (DCNA, 2019a). Seagrass beds have never been reported for the Saba Bank. A recent benthic survey by Meesters et al. (2024) also confirmed the absence of seagrass beds on the Saba Bank.

Macroalgal fields: On the East coast of Bonaire there are extensive macroalgal fields dominated by *Sargassum* spp. or by a mixture of *Sargassum* spp. and *Turbinaria* spp. (Kemenes van Uden et al., 2024). The area of seaweed fields on the east coast of Bonaire has previously been estimated to be 475 ha (Debrot et al., 2018), but this value still needs to be validated. On St. Eustatius, the macroalgal fields have been estimated to cover 578 ha (Debrot et al., 2014). However, by far the largest seaweed fields are found on the Saba Bank. The area of macroalgal fields at the Saba Bank has recently been estimated to range between 656 and 804 km², depending on which modelling technique is used (Meesters et al., 2024). Apart from an initial exploration by Littler et al. (2010) and brief descriptions by Toller et al. (2010), these macroalgal fields have not yet been described or mapped. Table 5 provides an overview of the area coverage of seagrass beds and macroalgal fields in the Caribbean Netherlands. This table shows that almost 60% of the seagrass beds in the Caribbean Netherlands are located on Bonaire, 34.5% on St. Eustatius and 5.6% on Saba, while over 98% of the macroalgal fields are located on the Saba Bank.

Table 5. *Overview of estimated area coverage (ha) of seagrass beds and macroalgal fields in the Caribbean Netherlands.*

	Bonaire	*St. Eustatius*	*Saba*	*Saba Bank*
Seagrass cover (ha)	215	124	20	0
Macroalgal cover (ha)	475	578	22	65,600 – 80,400

Assessment of National Conservation State

Trends in the Caribbean Netherlands

Seagrass beds: Long-term seagrass monitoring data over multiple years are limited, making it difficult to assess trends in the composition and extent of seagrass beds in the Caribbean Netherlands. However, there are clear indications of a negative trend in the coverage of native seagrass species at the expense of invasive *Halophila stipulacea*. For example, MacRae and Esteban (2007) reported that seagrass beds of St. Eustatius were dominated by native *Thalassia testudinum* and *Syringodium filiforme*. However, only a few years later these seagrass beds were dominated by dense beds of the invasive seagrass *H. stipulacea*, while *T. testidinum* was no longer found (Debrot et al., 2014). Moreover, when plotting the raw data on seagrass occurrence at 49 fixed locations in Lac Bay (Bonaire) between 2011 and 2024 as collected by by Engel (2024), we see a strong increase in occurrence of the invasive *H. stipulacea* from 6.0% *T. testudinum* occurrence decreased from 48.8% to 19.0% over the same period (Fig. 5). Note that the occurrence of native *S. filiforme* did stay rather constant with 3.9% in 2011 and 7.3% in 2024 (Fig. 5). Moreover, when plotting the change in species-specific seagrass occurrence per monitoring site in Lac Bay between 2011 and 2024, it becomes clear that at most sites where *T. testudinum* occurrence decreased from 2011 to 2024, it has been replaced by *H. stipulacea* (Fig. 6).

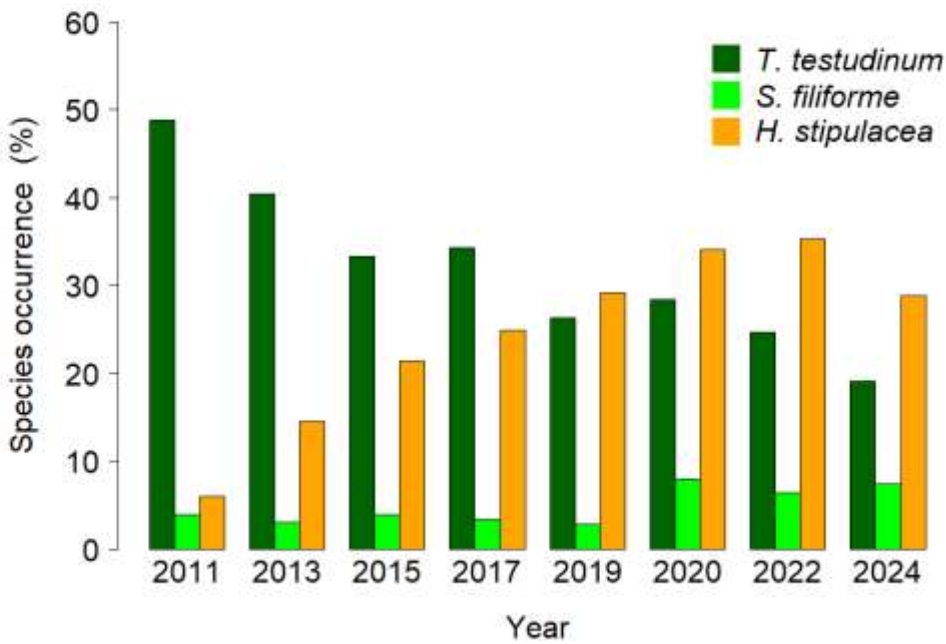

Figure 5. *Occurrence of native Thalassia testudinum and Syringodium filiforme and invasive Halophila stipulacea in Lac Bay, Bonaire between 2011 and 2024. Figure is based on data reported in Engel (2024).*

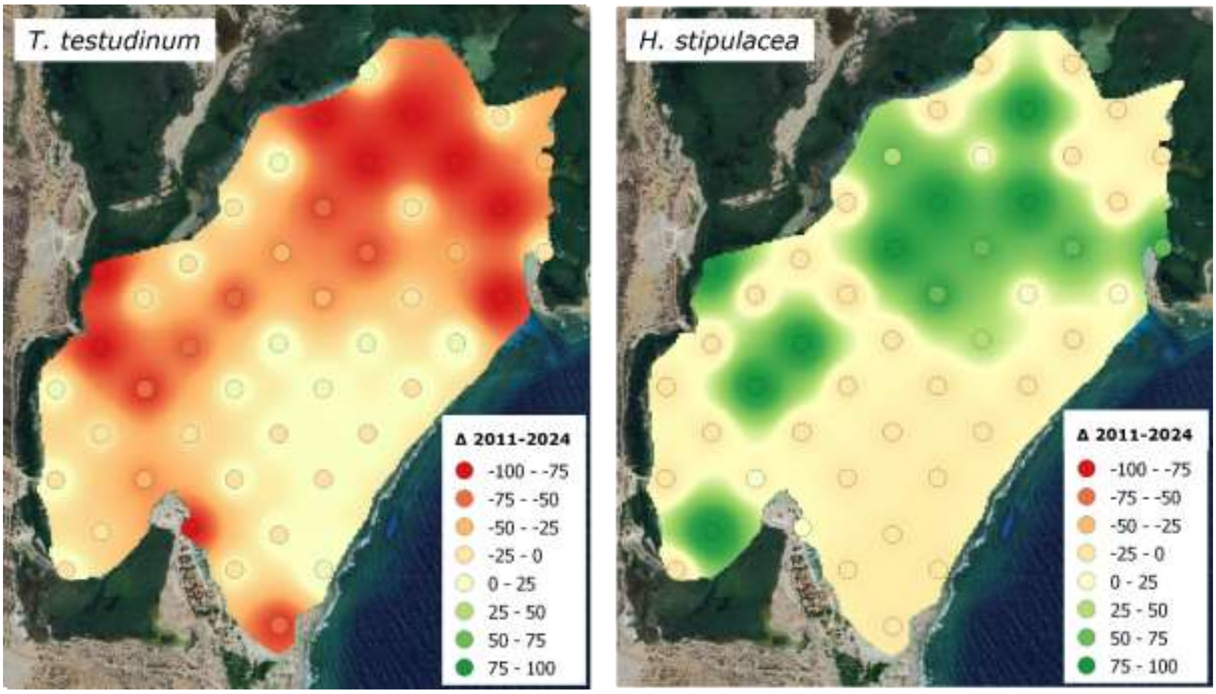

Figure 6. *Predicted change in seagrass occurrence (%) in Lac Bay between 2011 and 2024 for native T. testudinum (left panel) and invasive H. stipulacea (right panel). Decreased, increased or no change in seagrass species-specific occurrence over time is reflected in red, green and yellow, respectively. Dots reflect grid locations (N=49) where seagrass occurrence was visually assessed. Figure 6 is based on data reported in Engel (2024) and was made using QGIS 3.16 'Hannover'.*

Macroalgal fields: Due to the lack of long-term monitoring data on the composition and extent of macroalgal fields in the Caribbean Netherlands, no assessments can be made on possible trends.

Recent developments

Since 2015, the Caribbean Netherlands has also been hit hard by *Sargassum* influx events, especially on the East coast of Bonaire where *Sargassum* rafts washed up on beaches and in coastal bays (i.e., Lagun and Lac Bay) (DCNA, 2019b; van der Geest et al., 2024). In Lac Bay and Lagun, sudden die-offs of mangroves and seagrass beds have been observed at sites where *Sargassum* accumulated (Hanssen et al., 2024), and the extent of *Sargassum*-influx related mangrove die-offs have recently also been quantified by Mücher and van der Geest (2024). However, the negative impact of these recent *Sargassum* influx events on seagrass beds and macroalgal fields in the Caribbean Netherlands has not yet been quantified. As these massive *Sargassum* influx events are likely to become the new norm in the Caribbean (Wang et al., 2019), it is expected that they will continue to harm seagrass and macroalgal communities in the Caribbean Netherlands.

Assessment of distribution: favourable / favourable

The 6 native seagrass species that are found in the Caribbean Netherlands are widely distributed within the Tropical Atlantic bioregion, even so *H. baillonii* (VU) is rare listed on the IUCN Red List. These species can reproduce both vegetatively and via seeds. The natural distribution range of seagrass beds is assessed as favourable.

The natural distribution range of macroalgal fields is also considered favourable. The benthic *Sargassum* spp. Fields on the east coast of Bonaire and the more diverse and extensive macroalgal fields on the Saba Bank grow just above the coral reef zones in those locations (Toller et al., 2010), which makes them closely connected to the reef system elsewhere around Bonaire and/or on the Saba Bank.

Assessment of surface area: unfavourable-inadequate / favourable
Covering a total of 359 ha (see Table 5), the seagrass beds in the Caribbean Netherlands are relatively small in surface area. Moreover, monitoring data indicate a decline in seagrass coverage. However, the extent of this decline is not well quantified as historical reference values are lacking. For Lac Bay, a semi-enclosed bay on Bonaire, it has been estimated that the bay area suitable for seagrass beds has been decreasing by approximately 2.34 ha per year due to mangrove expansion into the bay (Erdmann and Scheffers, 2006; Hylkema et al., 2015). Given the relatively small surface area and the observed decline in seagrass coverage on both Bonaire and St. Eustatius, especially of native species, the surface area is assessed as unfavourable-inadequate.
The area of benthic *Sargassum*-dominated fields on the east coast of Bonaire, and the more diverse macroalgal fields of the Saba Bank have likely not decreased over the years (A.O. Debrot, pers. Comm.). Therefore, the surface area of seaweed fields is assessed as favourable.

Assessment of quality: unfavourable-inadequate / unknown
Seagrass bed species composition: Seagrass monitoring data indicate an ongoing shift from native to invasive *H. stipulacea*-dominated seagrass meadows, which may compromise seagrass ecosystem functioning (Smulders et al., 2017). For example, seagrasses with more opportunistic life strategies like *H. stipulacea* allocate less energy into the development of their below-ground biomass, and are therefore, more vulnerable to uprooting in storms (James et al., 2020). This susceptibility to uprooting reduces the overall storm resilience of the seagrass ecosystem and potentially accelerates the spread of *H. stipulacea* by dispersing vegetative propagules (Smulders et al., 2017). Moreover, grazing by green turtles (*Chelonia mydas*) has been shown to facilitate the rate and spatial extent of this invasive species' expansion, due to their preference for native seagrass species, and by increasing space for settlement (Christianen et al., 2019). Likewise, *Sargassum*-influx induced die-off events of native seagrass may also facilitate the spread of opportunistic *H. stipulacea* by increasing space for settlement. Moreover, Becking et al. (2014) found that the number of fish in *H. stipulacea*-dominated seagrass fields was only half of that in 'native' seagrass fields. More specifically, on transects in *H. stipulacea*-dominated seagrass fields, damselfish (Pomacentridae), goatfish (Mullidae), and barracudas (Sphyraenidae) were absent.

Based on the shift from native to invasive *H. stipulacea*-dominated seagrass beds, the current quality of the seagrass beds is assessed as unfavourable-inadequate. For the macroalgal fields, the quality is unknown.

Assessment of future perspective: unfavourable-bad / unfavourable-inadequate
Seagrass beds: The invasive *H. stipulacea* is rapidly replacing native seagrass beds, both in Bonaire and St. Eustatius, with negative effects on seagrass ecosystem functioning and resilience. Another significant threat is water quality degradation due to land-based inputs of sediment, nutrients and pollutants (Slijkerman et al., 2011; Debrot et al., 2010, 2012). Moreover, physical damage from tourist activities such as wading, surfing, or boating (propeller action) threatens shallow seagrass meadows (Debrot et al., 2012). Additionally, there are expected consequences of climate change, such as higher sea water temperatures, and more frequent and intense storms and hurricanes, which will negatively affect seagrass beds (Trégarot et al., 2024). Seagrass beds are also threatened by suffocation due to massive reoccurring influxes of floating *Sargassum* brown algae that are likely to persist in the future (van der Geest et al., 2024). Therefore, the future perspective for seagrass beds is assessed as unfavourable-bad (Table 6).

Table 6. *Overview of the status of seagrass beds of the Caribbean Netherlands for different aspects.*

Aspect Seagrass beds	2024
Distribution	Favourable
Surface area	Unfavourable-inadequate
Habitat quality	Unfavourable-inadequate

Future prospects	Unfavourable-bad
Overall Assessment of Conservation State	**Unfavourable-bad**

Macroalgal fields: Little is known about the extent by which macroalgal fields in the Caribbean Netherlands are affected by local stressors and climate change. However, as almost all macroalgal fields in the Caribbean Netherlands are found on the remote Saba Bank in relatively deep and well-flushed waters, they are likely to receive limited impact from local anthropogenic stressors. In contrast, these macroalgal fields seem sensitive to climate change, especially to increases in maximum seawater temperatures, which could depress biomass production and/or drive phenological shifts in canopy formation that could affect their capacity to support higher trophic levels (Fulton et al., 2019). In addition, benthic macroalgal fields are expected to be negatively impacted by the recent massive influxes of floating *Sargassum* that are likely to persist in the future (van der Geest et al., 2024). Therefore, the future perspective of macroalgal fields is assessed as unfavourable-inadequate (Table 7).

Table 7. *Overview of the status of macroalgal fields of the Caribbean Netherlands for different ecological aspects.*

Aspect macroalgal fields	2024
Distribution	Favourable
Surface area	Favourable
Habitat quality	Unknown
Future prospects	Unfavourable-inadequate
Overall Assessment of Conservation State	**Unfavourable-inadequate**

Comparison to the 2018 State of Nature Report

Overall, the current state of the seagrass beds of the Caribbean Netherlands has measurably worsened compared to the 2018 assessment, especially due to the rapid expansion of invasive *Halophila stipulacea* at the expense of native seagrass species. For the current state of the macroalgal fields of the Caribbean, no major changes can be meaningfully identified.

Recommendations for National Conservation Objectives

National long-term goals
The target for achieving a favourable Conservation State is an increase in the current distribution of native seagrass beds, an increase in their area coverage, and an improvement of their quality. Preservation of the distribution, area coverage, and improvements in the quality of macroalgal fields along the east coast of Bonaire and on the Saba Bank.

National short-term (5-year) goals
Improvement of the quality and resilience of seagrass beds by addressing numerous local threats such as terrestrial sediment run-off, eutrophication, pollution, massive influxes of floating *Sargassum*, and damage from tourist activities.

Key Threats and Management Implications

Table 8. *Overview of the main threats to the seagrass beds of the Caribbean Netherlands and implications for management.*

Main threats		**Management actions**
Climate change	Increase in frequency and intensity of storms, hurricanes and heat waves, rising sea surface temperatures.	• Reduce the impact of local stressors, so that seagrass ecosystems are more resilient to the impacts of climate change.
Eutrophication	Eutrophication leads to unfavourable light conditions for seagrass, as nutrient over-enrichment stimulates algal overgrowth as epiphytes and macroalgae, and phytoplankton blooms in the water column.	• Reduce coastal eutrophication, preferably to zero, but at least aim for standards as stringent as those in Europe. • Reduce *Sargassum*-induced organic loading of coastal bays. Reduce land-based nutrient run-off into the sea, by improved watershed management.
Invasive species	Invasive *H. stipulacea* is outcompeting native seagrass species, resulting in a loss of ecosystem services like coastal protection, carbon storage and enhanced biodiversity.	• Reduce coastal eutrophication, as eutrophication is suggested to stimulate the expansion of *H. stipulacea*. • Reduce Sargassum influx-induced die-offs of native seagrass, as species with more opportunistic life strategies like *H. stipulacea,* will quickly colonize the bare sediment.
Pollution	Waste is not processed but dumped on land, much of which eventually ends up in the sea.	• Implementation of a waste management system according to Dutch standards.
Coastal erosion	Land degradation, particularly due to overgrazing, leads to erosion and terrestrial sediment input into the sea.	• Reduce overgrazing and implement active management of livestock. • Reduce coastal construction and industrial activity in upstream drainage areas of coastal bays and saliñas.
Damage from tourism-related activities	Damage or loss of seagrass beds due to wading, boating, anchoring, or coastal development.	• Zoning and improved visitor management • Supervision and law enforcement • Ecological impact studies and implementation of mitigating and compensatory measures when implementing construction projects.
Massive *Sargassum* influxes	Since 2015, seagrass beds in the Caribbean Netherlands are threatened by suffocation due to massive reoccurring influxes of holopelagic *Sargassum* brown algae that are likely to persist in the future	• Implementation of an early warning system • Strategic placement of oil booms to prevent *Sargassum* from smothering (native) seagrass beds • Timely removal of *Sargassum* that accumulates behind the oil booms • Development of a *Sargassum*-based valorization chain to offset *Sargassum* cleaning costs.

Data Quality and Completeness

In recent years, numerous studies have been conducted that have provided a reliable overview of the current Conservation State and threats to seagrass beds on Bonaire. However, the seagrass beds of St. Eustatius and Saba have received less attention. Therefore, it is now important to implement a monitoring system that addresses the coverage and functioning of all the seagrass beds in the Caribbean Netherlands. This is necessary both to establish long-term trends and to evaluate the effects of management measures taken. In comparison, research on Conservation State and threats of macroalgal fields is still in its early stages.

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10 Conservation State of Coral Reefs and Communities of the Caribbean Netherlands

Meesters[1], E. H., van der Geest[1], M., Kemenes van Uden[1], T., Boman[2], E., Butler[2], E., Hylkema[3,4], A., Lehwald[3], M., Wulf[5], K., Eckrich[6], C. and Francisca[6], R. 2025. From: State of Nature Report for the Caribbean Netherlands, 2024. WUR report C001/25.

Status

The coral habitats in the Caribbean Netherlands can be found in the marine environments of Bonaire, Saba, and St. Eustatius, and the Saba Bank. When viewed from a historical perspective the current status of all these areas should generally be viewed as extremely unfavourable, though there are local areas that present hopeful exceptions. Caribbean wide coral cover has decreased steadily over the last 50 years (Jackson et al., 2014), caused by anthropogenic pressures such as overfishing, pollution, diseases, and, more recently, by climate change induced events like bleaching.

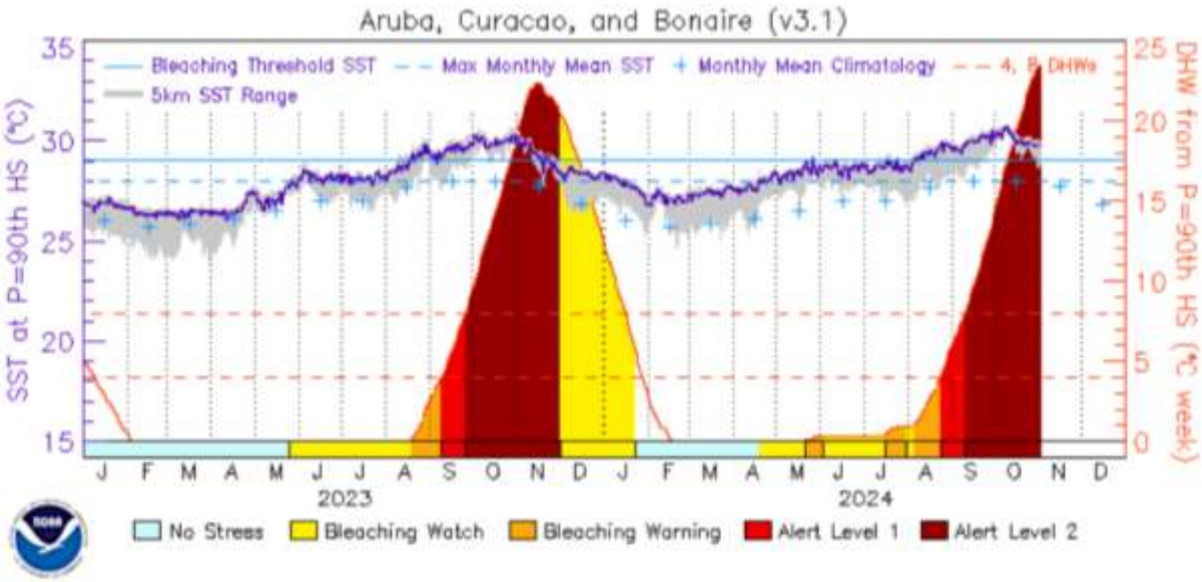
Conservation and protection of corals is a goal of many international treaties, such as the Convention of Biological Diversity (CBD, entered into force 1993), the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES, 1975), the Ramsar Convention on Wetlands (1975), the United Nations Convention on the Law of the Sea (UNCLOS, 1994), and UNESCO's World Heritage Convention (WHC, 1975). Important for the Caribbean are also regional agreements such as The Protocol Concerning Specially Protected Areas and Wildlife (SPAW), as part of the Cartagena Convention (1986). Notwithstanding these agreements, little progress has been made in turning the tide for coral reefs. Organizations that play an important role in the protection of coral reefs are the United Nations Environment Programme (UNEP), the International Coral Reef Initiative (ICRI), the Global Coral Reef Monitoring Network (GCRMN), the World Wildlife Fund (WWF), and The Nature Conservancy (TNC). Within the Kingdom of the Netherlands the Caribbean Netherlands' coral reefs are protected through established Nature Parks, national laws, and local regulations.

Management and protection of the marine resources of the Caribbean Netherlands is carried out under supervision of the Ministry of Agriculture, Fisheries, Food Security and Nature (LVVN), the Ministry of the Interior and Kingdom Relations (BZK), and the Ministry of Infrastructure and Water Management (I&W), with delegated responsibilities to the island governments of the public entities Saba, St. Eustatius, and Bonaire. Day to day management of the marine parks is carried out by mandated non-governmental organizations, being Stichting Nationale Parken (STINAPA) on Bonaire, St. Eustatius National Parks (STENAPA) on St. Eustatius, and Saba Conservation Foundation (SCF) on Saba. Monitoring of the status of the coral communities of the Caribbean Netherlands is carried out in collaboration with the local NGOs.

Characteristics

Description
An evaluation of the Conservation State of corals in the Caribbean Netherlands requires a separation between the southern and the northern part. These two areas are more than 600km apart and climatologically very different. In the south the environment is characterized by semi-arid conditions receiving much less rain than the northern area. A very important consideration for reef development is

the difference in hurricane return times between the northern and the southern Caribbean. Historical patterns show that around Aruba, Curaçao, and Bonaire hurricane frequency is much less than around Saba, St. Eustatius, and St. Maarten. This is most likely the reason why reef development on the Saba Bank occurs deeper than on Bonaire and fringing reefs have not developed around Saba, St. Maarten, and St. Eustatius. Local threats vary between the relatively small and sparsely populated Saba and St. Eustatius, and densely populated Bonaire. The main local threats are overfishing (often artisanal), eutrophication, pollution, and erosion. More regional threats include diseases and the consequences of climate change. Diseases spread through the whole Caribbean without much distinction between islands and appear to occur more frequently in recent years. In 2023 the Stony Coral Tissue Loss Disease (SCTLD) has wreaked havoc on Bonaire and on Saba, and a (possibly recurring) disease among black sea urchins has decimated these important herbivores once again (Hylkema et al., 2023). Bleaching of corals through extensive periods of increased sea surface temperatures also occurs almost yearly now and has been extreme in 2023 (Figure 1) and 2024 is currently developing into one of the warmest years on record with bleaching likely causing severe stress again later in the year (Figure 2). Observations in October 2024 suggest that the combination of bleaching and disease have had a devastating effect on the reef, where certain coral species having been removed completely and coral cover decreased



generally everywhere.

Figure 1. *Sea surface water temperature around the A, B, C islands in 2023 and 2024. The dark blue line depicts the temperature (left y-axis), the grey area indicates temperature values during previous years till 1985. The coloured area displays the Degree Heating Weeks (right y-axis), a measure for the amount of stress corals are exposed to as a consequence of heating. Data courtesy of NOAA.*

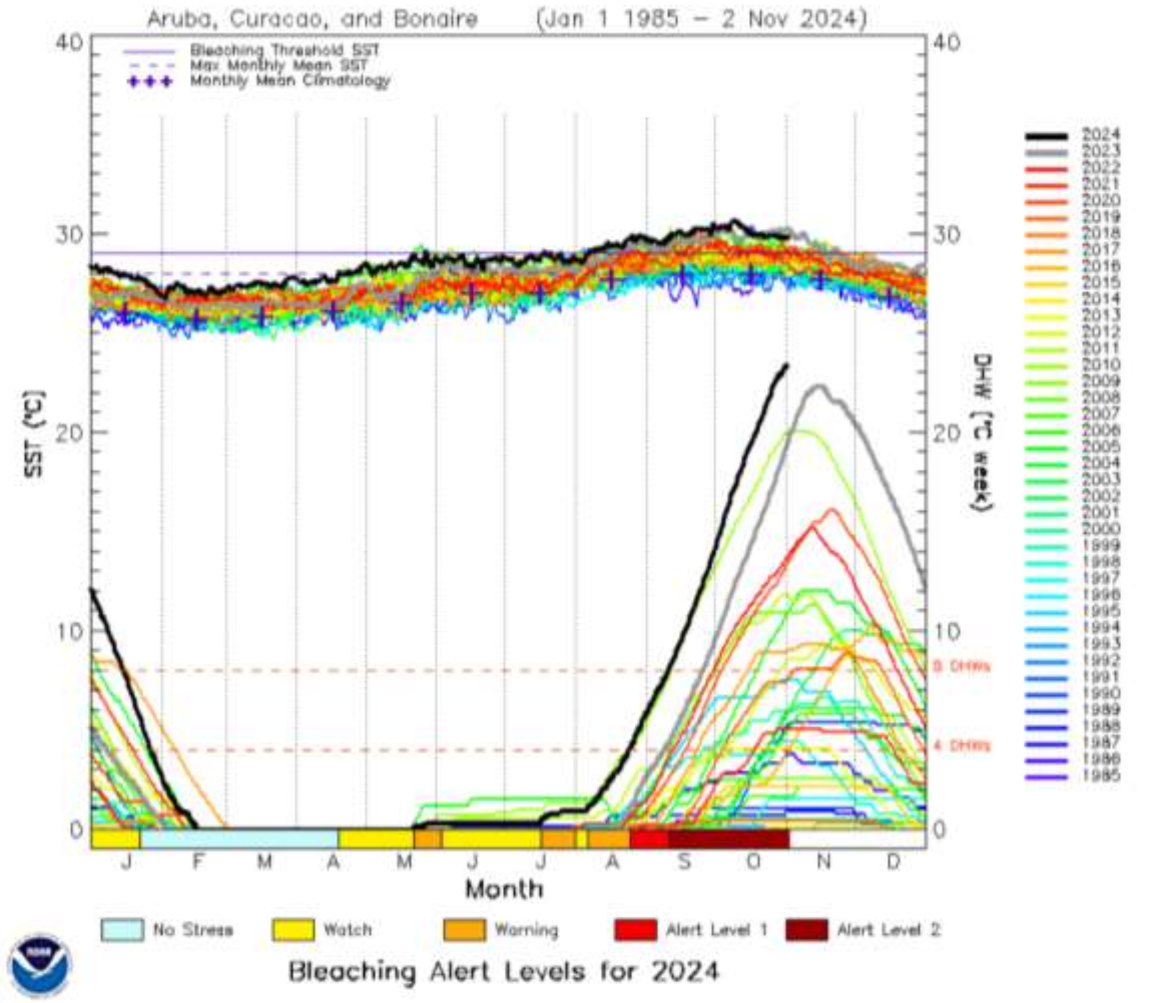


Figure 2. *Multiple years graph of temperature and Degree Heating Weeks (DHW) for A, B, C islands. Courtesy NOAA. DHW is a metric for the amount of stress corals experience due to warmer sea surface temperatures.*

Relative Importance within the Caribbean

Coral reefs are the most diverse ecosystem in the world. They are biodiversity hotspots, providing food and protection to numerous species, and a plethora of ecosystem services to human societies. For the islands of the Caribbean Netherlands, underwater nature has always been one of the pillars of the local economy which depends heavily on tourism. The reef of Bonaire reportedly is one of the best reefs in the Caribbean (Jackson et al., 2014), however since the start of monitoring in the Dutch Caribbean this reef has suffered under many different pressures (Bak and Luckhurst, 1980; De Bakker et al., 2016; de Bakker et al., 2017; de Bakker et al., 2019).

Ecological aspects

Habitat

There is a very clear distinction between the reefs in the south, the so-called leeward islands, and the reefs in the north, also called the windward islands (Bak, 1975). Reef development is much higher in the south along the leeward sides of the islands of Bonaire, Curaçao and Aruba. The reefs in the south are quite similar with a gradually sloping terrace to a drop-off at 7-12m depth after which the slope increased to 45 degrees to vertical (Bak, 1977; Duyl, 1985). A second terrace at 50-60m may be present locally and a second drop-off beyond that. Depending on depth, different coral species can be found at any locality, for example, *Acropora palmata*, or elkhorn coral, generally occurs between 1-4m depth in a zone with strong water movement. *A. cervicornis*, or staghorn coral, generally occurs somewhat deeper, between 4 and 8m depth. Around 10-12m is generally the area with the highest coral growth and diversity. Over the drop-off, coral cover and diversity remain high to approximately 35-40m after which they rapidly decline. As light decreases exponentially with depth, coral colonies become flatter and coral growth decreases.

In the northern area of the Caribbean Netherlands, a large reef complex occurs along the eastern and southern rim of the Saba Bank. Around Saba and Statia corals can be found in varying densities on mostly volcanic underground.

Survival
Corals, having evolved in nutrient-poor waters, depend for their survival on clear and clean water. Living in symbiosis with zooxanthellae, unicellular algae, they are to a large degree dependent on sunlight for their energy. Main threats are therefore often related to water clarity, such as eutrophication and sediment run-off. Other important factors for coral survival are temperature and diseases. Diseases that have had a large impact on coral reefs can be diseases that target corals directly, such as the White Band Disease (Bak and Criens, 1981; Gladfelter, 1982; Duyl, 1985) or the Stony Coral Tissue Loss Disease (Alvarez-Filip et al., 2019), but also diseases that targeted important functional species such as the black sea urchin *Diadema antillarum* (Lessios et al., 1984) which is a crucial grazers of turf and macro-algae. Marine heat waves cause coral bleaching, can have devastating impacts, and are increasing in frequency and severity (Bove et al., 2022). Climate change induced sea level rise is also a threat as reefs may not be able to keep up with rising waters (Perry et al., 2013; de Bakker et al., 2019) and corals end up in deeper water where their growth is even slower.

Coral reefs perform many functions for resident flora and fauna, but also to human societies, where they are called ecosystem services such as the provisioning of food in the form of fish and shellfish, shoreline protection, and opportunities for tourism and recreation. With the degradation of coral reefs, these services are also under threat.

Minimum viable population size
Globally, and even more so in the Caribbean, the survival of coral reefs is in grave peril (Hoegh-Guldberg et al. 2023) and the warming of the oceans appears to be speeding up. Crossing the 1.5 degrees threshold will have devasting effects on ecosystems all around the globe, but particularly for shallow water tropical coral reefs that have evolved in waters with very limited variation in temperature. The IPCC Climate Change 2023 report predicts 70-90% loss of warm water coral reefs at 1.5 degrees heating and more than 99% at 2 degrees heating with high confidence (Core Writing Team, 2023).

Present distribution and Reference Values

Of the 65 coral species in the IUCN Red List database 19 are critically endangered (Table 1), 11 near threatened, 44 least concern, and 2 data deficient (IUCN database accessed 27/06/2024).

Table 1. *Critically endangered coral species according to IUCN.*

Common name	*Species name*
Staghorn Coral	*Acropora cervicornis*
Pillar Coral	*Dendrogyra cylindrus*
Smooth Flower Coral	*Eusmilia fastigiata*
Lowridge Cactus Coral	*Mycetophyllia danaana*
Atlantic Mushroom Coral	*Scolymia lacera*
Maze Coral	*Meandrina meandrites*
Rough Cactus Coral	*Mycetophyllia ferox*
Grooved Brain Coral	*Diploria labyrinthiformis*
Ten-ray Star Coral	*Madracis decactis*
Closed-valley Brain Coral	*Colpophyllia breviserialis*
Elkhorn Coral	*Acropora palmata*
Lamarck's Sheet Coral	*Agaricia lamarcki*
Lowrelief Lettuce Coral	*Agaricia humilis*
Jackson maze coral	*Meandrina jacksoni*
Artichoke Coral	*Scolymia cubensis*
Symmetrical Brain Coral	*Pseudodiploria strigosa*
Massive Starlet Coral	*Siderastrea siderea*
Thin Leaf Lettuce Coral	*Agaricia tenuifolia*
Sunray Lettuce Coral	*Helioseris cucullata*

Next to the Critically Endangered species according to the IUCN, in 2023 the Stony Coral Tissue Loss Disease (SCTLD) has severely impacted the following species: Maze coral (*Meandrina meandrites*), flower coral (*Eusmilia Fastigiata*), great star coral (*Montastraea cavernosa*) and the brain corals (*Pseudodiploria strigosa*, *Diploria labyrinthiformis* and *Colpophyllia natans*) as well as pillar coral (*Dendrogyra cylindrus*) and star coral (*Dichocoenia stokesi*) on Bonaire. It must be feared that all these species are now as well critically endangered and close to extinction (Pepe et al., 2025).

Description of the distribution and community composition of the coral reefs of the Dutch Caribbean (all 6 islands) go back to the early seventies of the previous century (Roos, 1971; Bak, 1975; Bak, 1977; Van der Land, 1977; Duyl 1985). Only the islands in the southern Caribbean and the Saba Bank harbour fringing coral reefs in the sense that the areas around the islands consist of calcium carbonate bottoms built by corals over thousands of years. The islands of St. Maarten, Saba, and St. Eustatius are characterized by the presence of coral communities that grow on lava outcrops and where calcium carbonate bottom formation is either absent or at best very rudimentary.

Assessment of National Conservation State

The status of the coral reefs of the Caribbean Netherlands should be described as extremely unfavorable.

Trends in the Caribbean Netherlands
Trends of varying length can be constructed with data from different sources for coral cover for each of the three islands. Historical data for coral reefs are generally limited in duration, but for Bonaire (and Curaçao) we have access to data that form the longest time-series in the world. The data collection was initiated by Prof. Dr. R.P.M. Bak from the Netherlands Institute of Sea Research and the University of Amsterdam in 1973 (1974 for Bonaire) and is nowadays continued by researchers from Wageningen Marine Research with support from the Ministry of Agriculture, Fisheries, Food Security and Nature. The data are based on images of approximately 9 square meters of reef at 10, 20, 30, and 40m depth at multiple locations on Bonaire and Curaçao. On Bonaire the data are mainly from Karpata, one of the oldest diving locations in Bonaire, and by many divers still considered as one of the best locations on the leeward side of Bonaire.

Bonaire: long-term prospects

The four long-term trends in Figure 3 indicate that the reef used to have much higher coral cover in the nineteen seventies. From 30m upward cover by corals was higher than 60%. At 10 and 20m depths, coral degradation started already during the eighties with some recovery at 10m depth during the late nineties but then continued further downward from 2000 to 2010. At 10m some recovery occurred between 2015 and 2022, but in 2023 coral cover again appears to decrease. At 20m depth coral cover decreased more rapidly, bottoming around the end of the previous century, not showing any signs of recovery till this day. At 30 and 40meters depth coral cover decreased almost continuously during the last 50 years.

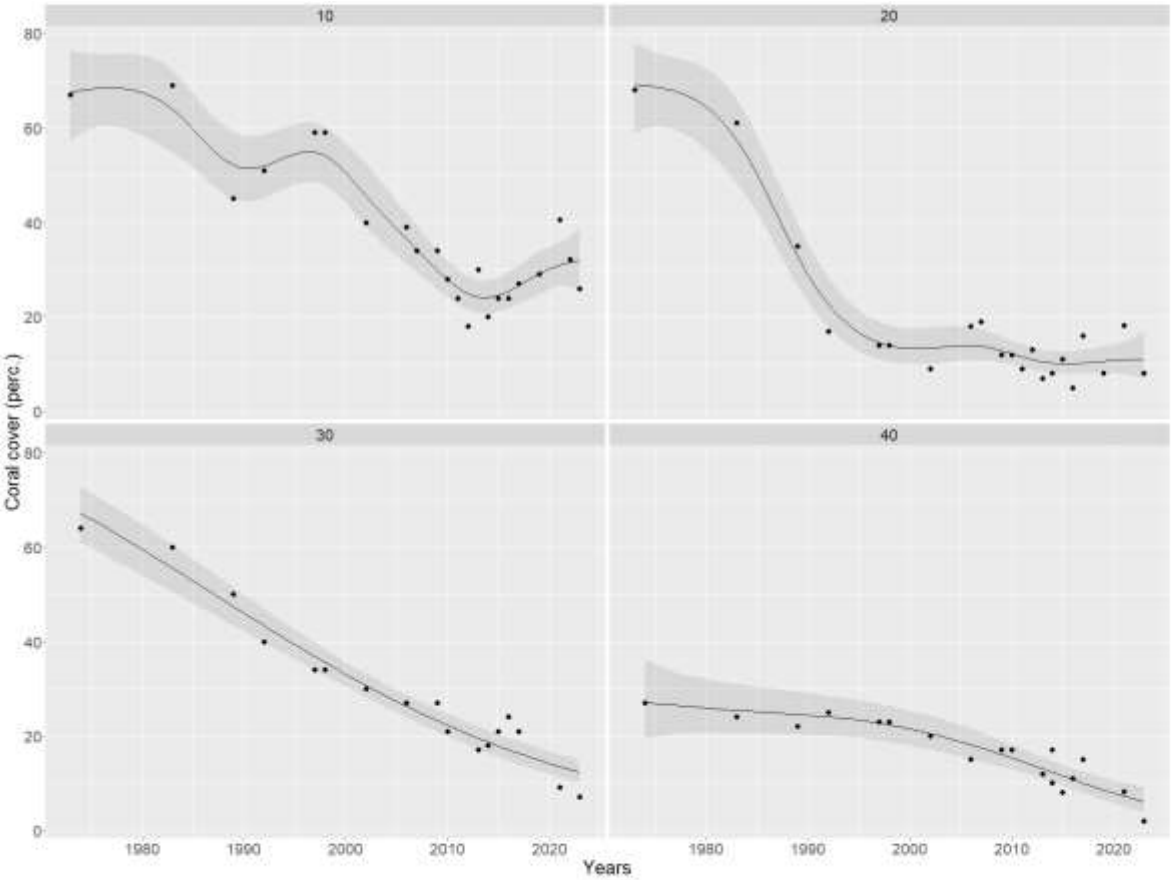


Figure 3. Long-term trend for Karpata, Bonaire, at 10, 20, 30, and 40m depth. The black line denotes the estimated mean value and the shaded area indicates the 95% confidence interval. Note that the trends do not yet include the effects of 2024.

Bonaire: area wide prospects.

In 2014, the first reef inventory of the entire leeward coast of Bonaire was conducted. Fish and coral communities were assessed at 115 sites at 5 and 10m depth. This survey has now been repeated in 2017, 2020, and 2023 and for 10m depth the data are used to calculate the so-called Reef Health Index (RHI[2]). This index uses 4 important variables and is being consistently used in the meso-American reef area. For each location, a score from 1 to 5 (Table 2) is calculated for the amount of coral cover, cover by macro-algae, biomass of herbivorous fish (mainly parrotfish), and the biomass of commercial fish (mainly groupers and snappers) based on pre-set criteria (Table 2). Next, the average value per location is calculated to arrive at the RHI value. The average value is assessed as follows: critical, 1-1.8; poor, >1.8-2.6; fair/ok, >2.6-3.4; good, >3.4-4.2; very good >4.2-5. Thus, an RHI value is obtained for each of the 115 locations and an overall mean index is calculated for the leeward reef as a whole (Figure 4). Figure 4 shows the mean RHI value based on 115 sites for the years 2014, 2017, 2020, and 2023 only

[2] www.healthyreefs.org

for sites at 10m depth. This is depth that this index is generally meant for as it is the zone where most research in the past has concentrated upon. Overall, the reef of Bonaire at 10m depth appears relatively stable and is judged as still doing ok, but the 2023 value is significantly lower than 2020 and close to becoming bad at the next survey round. The data from 2023 also do not yet include the consequences of the bleaching and disease which were both most pronounced at the end of 2024.

Table 2. *Reef Health Index categories. Very good scores get 5 points, good 4 points, fair 3, etc. The final RHI value is calculated by averaging the values of the 4 variables.*

RHI variables	Very good (5)	Good (4)	OK/Fair (3)	Bad/Poor (2)	Critical (1)
Coral cover (%)	≥ 40	20.0-39.9	10.0-19.9	5.0-9.9	<5
Macro-algal cover (%)	0-0.9	1.0-5.0	5.1-12.0	12.1-25	>25
Biomass herbivorous fish (g/100m2)	≥3480	2880-3479	1920-2879	960-1919	<960
Biomass commercial fish (g/100m2)	≥1680	1260-1679	840-1259	420-839	<420

Table 3. *Mean values (between brackets the limits of the 95% confidence interval) for fish biomass of herbivorous and commercial fish, and cover of corals and macro-algae from 2014 to 2023 for the whole leeward side of Bonaire at two depths. RHI categories for 10m as in Table 2.*

	2014	2017	2020	2023	Depth (m)
Mean biomass commercial fish ($g/100m^2$)	54 (34; 83)	65 (41; 100)	137 (95; 192)	45 (27; 70)	5
	325 (238; 434)	466 (400; 539)	832 (714; 964)	422 (338; 521)	10
Mean biomass herbivorous fish ($g/100m^2$)	1547 (1282; 1852)	1448 (1204; 1726)	1492 (1189; 1849)	775 (592; 997)	5
	1769 (1553; 2008)	2332 (2156; 2520)	2712 (2428; 3021)	1841 (1648; 2051)	10
Mean coral cover (percentage)	6.2 (4.6; 8.2)	3.6 (2.4; 5.1)	5.2 (3.8; 6.9)	4.4 (3.1;6.0)	5
	18.7 (16.9; 20.7)	17.3 (14.6; 20.4)	19.6 (16.8; 22.7)	15.4 (13.2;17.9)	10
Mean cover macro-algae (percentage)	0.1 (0.1; 0.3)	1.1 (0.5; 2.0)	0.9 (0.5; 1.6)	0.6 (0.3; 1.1)	5
	1.7 (1.1; 2.5)	4.8 (3.4; 6.6)	4.2 (3.1; 5.5)	4.6 (3.2; 6.4)	10

Looking at the individual variables for 5 and 10m (Table 3) coral cover has decreased from 18.7 to 15.4 over the course of 10 years. At 5m depth coral cover also decreased, from 6.2 to 4.4 percent. Cover by macro-algae is relatively low, but fish biomass is generally in a bad condition. For the commercial fish especially, the values are almost critical. This seems to indicate that the larger predatory fish, groupers and snappers, are under too much pressure, most likely from illegal fishing. Also, all values at 5m are lower than at 10m. This is a common feature for the reef, but this means that on average, values for the whole reef terrace are lower than those found at 10m. Around 10m depth coral cover is generally highest but a value of 15.4% is, from an historical perspective (Figure 3), very low and based on scientific research not enough to keep up with sea level rise (de Bakker et al., 2019).

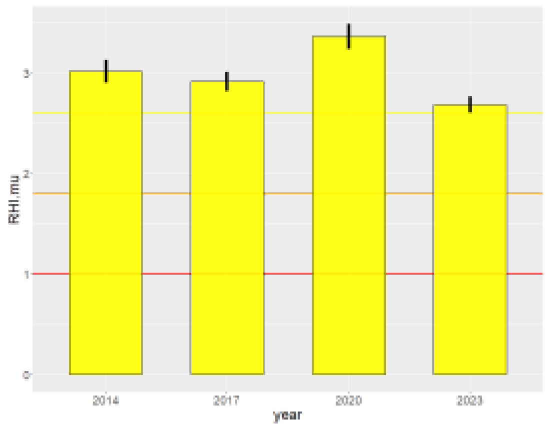


Figure 4. *The average RHI index from 2014 till 2023 for Bonaire at 10m depth. Horizontal lines present the thresholds below which a coral reef is judged as ok, bad, or critical. Each bar is based on measurements from 115 sites on the leeward side of the island.*

Figure 5 shows the average index and the distribution of the 115 sites over the different reef health categories from critical to very good. The overall index has decreased in 2023 after being relatively stable from 2014 to 2020 and the number of good and very good sites is decreasing, while the number of sites that qualify as bad and critical has increased.

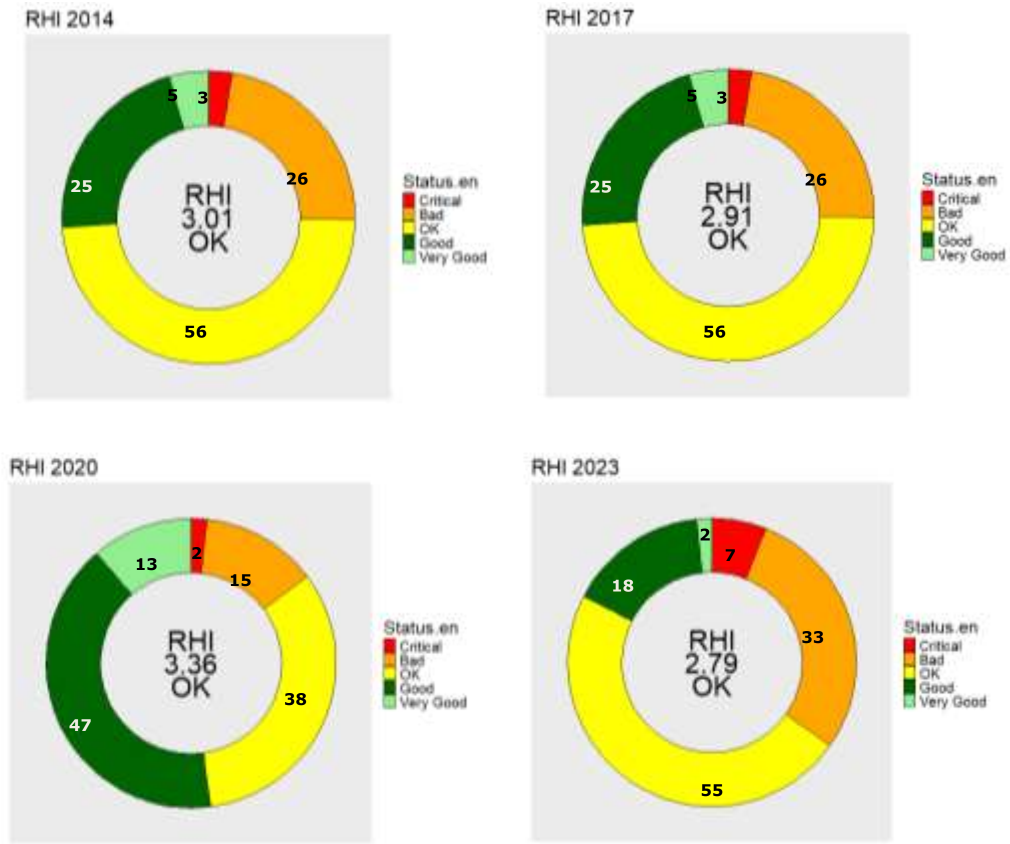


Figure 5. *The data of Table 3 at 10m depth visualized. Numbers in the ring refer to the number of sites in the corresponding category.*

Saba

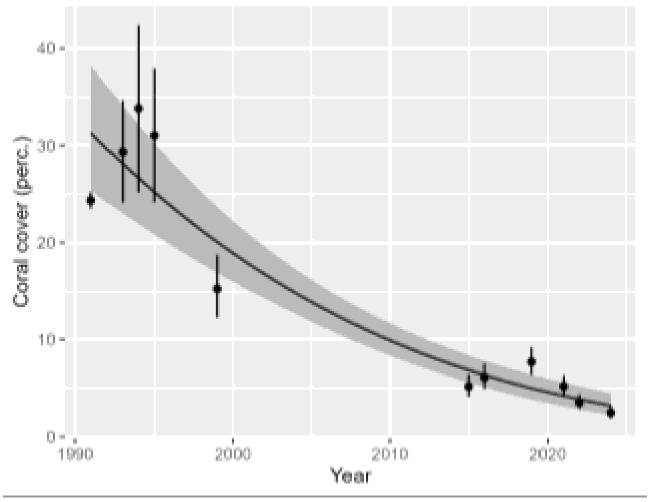


Figure 6. *Average coral cover around the island Saba (left) and a map (©OpenStreetMap) of the different monitoring locations around Saba (right).*

Around Saba coral cover has decreased since the early nineties (Figure 6). The amount of cover by algae (turf and macro-algae, see Table 4) is now more than 60%. The most likely causes for this degradation are increased run-off through erosion, bleaching mortality caused by climate change, coral diseases, and eutrophication of the coastal zone potentially aggravated by incidental sewage dumps by passing cruise ships. Passing cruise ships can have 4 times as many passengers as Saba has inhabitants. Under the MARPOL treaty cruise ships allowed to dump organic waste like food and sewage (lightly treated) already 3 nautical miles from land. This may create clouds of nutrient-rich water large enough to bath the whole island.

Table 4. *Cover of main benthic categories on Saba in 2024.*

Benthic category (>1%)	Mean percentage cover (95% confidence limits)
Corals	2.15 (1.63,2.79)
Crustose coralline algae	4.82 (3.31,6.79)
Cyanobacteria	1.01 (0.61,1.6)
Macro algae	33.65 (26.64,41.96)
Sponges	6 (3.94,8.77)
Turf algae	27.94 (22.48,34.33)

The status of the coral communities around Saba should be evaluated as extremely unfavourable.

St. Eustatius

Since the previous State of Nature coral cover in St. Eustatius has not improved. If anything, it has further deteriorated (Figure 7). The number of monitoring locations has increased, and monitoring is now yearly conducted by St. Eustatius National Parks (STENAPA). Average coral cover in 2023 is less than 1%, and with 1.5%, somewhat higher in the reserves. Estimated cover for the other important benthic categories is given in the table below. This indicates that more than 50% of the living bottom cover nowadays consists of algae. A notable difference between St. Eustatius and Saba is the cover by turf algae and cyanobacteria which is much higher than on Saba. Cover by macro-algae on the other hand appear to be much lower. Possibly, this is an effect caused by difference in the time of sampling, as cyanobacteria are most abundant in the warmer periods.

Table 5. Cover of main benthic categories on St. Eustatius in 2023.

Benthic category (>1%)	Mean percentage cover (95% confidence limits)
Corals	0.89 (0.5, 1.4)
Crustose coralline algae	0.04 (0.01,0.15)
Cyanobacteria	14.2 (8.8, 21.6)
Macro-algae	16.4 (12.1, 21.7)
Sponges	7.5 (5.58, 9.86)
Turf algae	39.3 (32.1, 47.6)

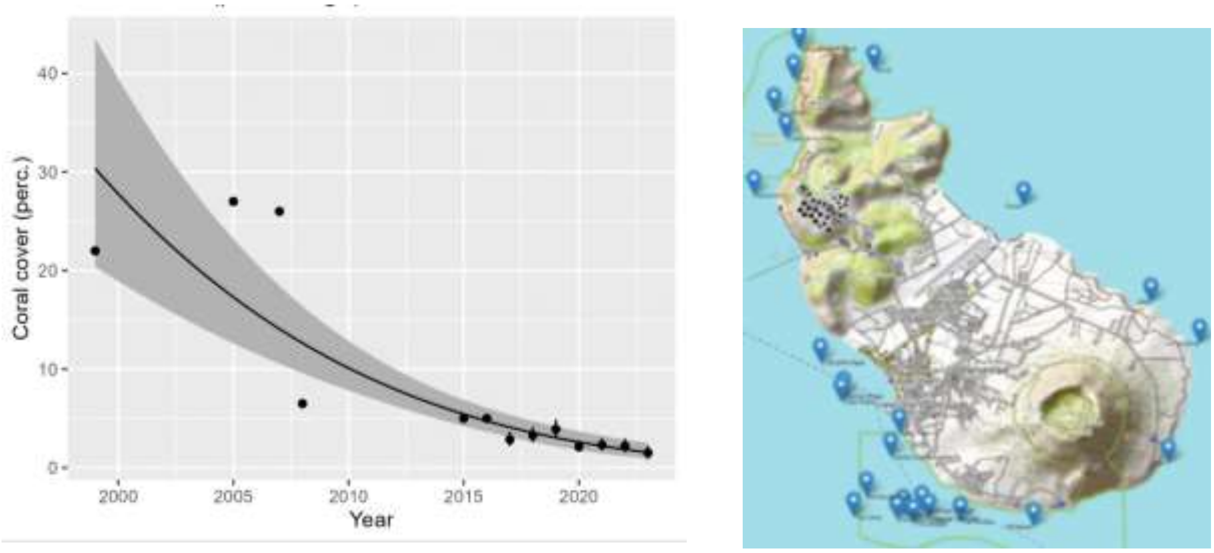


Figure 7. Average coral cover around St. Eustatius (left) and a map (©OpenStreetMap) of the different monitoring locations around the island (right) and locations of the two marine reserves (green lines). Data used for this graph include only the 15 sites in the northern and southern reserves so the data could be combined with data collected before 2017. The line is the best non-linear fit to the data with the grey band indicating the 95% confidence interval.

Saba Bank

The Saba Bank has been extensively described in the previous State of Nature (2017). A first expedition happened in 1972 by the Dutch Navy with van der Land (1977). The Saba Bank covers some 2500 km², but the areas of coral reef habitat are approximately 255 km² according to van der Land (1977). In terms of depth there are four main areas that can be distinguished on the bank (Figure 8). The eastern half ranges between a minimum depth of 12m and approximately 35m. The western half between 35 and 60m and a northern part that lies below 60m depth. The last part is known as the Luymes bank in the north which may have a very different origin than the rest of the bank.

Figure 8. *Bathymetric map of the Saba bank. Background from ESRI ocean. The islands of Saba and St. Eustatius can also be seen in the map.*

The Saba Bank has been visited occasionally by researchers and given the size of the bank, there is only limited quantitative data. Since 2010, there have been expeditions to the bank to investigate the status of the corals and fish at 10 locations. An overview of the coral cover from the reports and scientific literature is given in Figure 8. This figure also indicates events that likely led to the large decrease in coral cover. Especially, hurricane Lenny and two periods of very warm sea water are probably the main causes of the observed decline.

Although the Saba Bank may hardly be affected by pollution from land due to its location, the data also show a drastic decline in coral cover. Most likely, periods of extremely warm seawater are the main reason for this decline on the Saba Bank. New data are being collected in 2024 and will be available in 2025, however, given the fate of corals on Saba and St. Eustatius it must be feared that coral cover on the Saba Bank has also further declined.

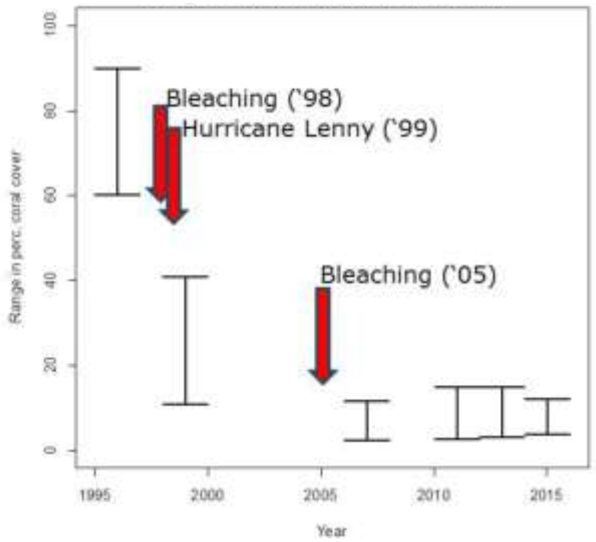


Figure 9. Reported minimum and maximum values of coral cover on the Saba Bank. (Data 1996, (Meesters et al., 1996); 1999, (Klomp and Kooistra 2003), 2007; (Toller et al., 2010); 2011 en 13, (Beek and Meesters, 2013)).

An extensive study on the different habitats on the Saba Bank (Meesters et al., 2024) has resulted in a map of the most likely distribution of habitats over the bank (Figure 9) based on machine-learning techniques. From this analysis it becomes clear that the coral reef area of the Saba Bank constitutes by far the largest continuous reef area of the Dutch Caribbean (Table 6). Therefore, its preservation and management are of international importance.

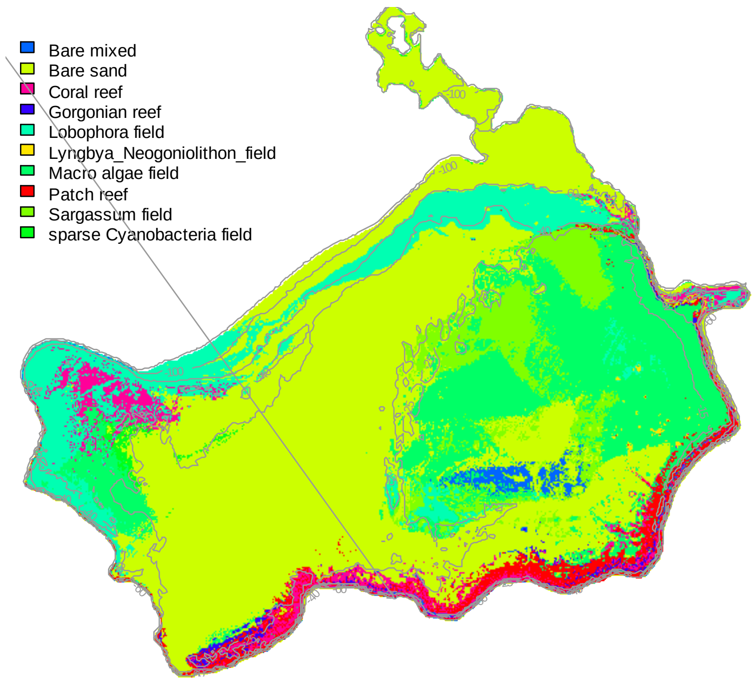


Figure 10. *Habitat prediction of the Saba Bank based on the weighted K Nearest Neighbour analysis using data extracted more than 2000 georeferenced images from the bank.*

Table 6. *Approximate areas of the different habitats on the Saba Bank.*

Habitat	Estimated area (km^2) based on Weighted K Nearest Neighbor analysis
Bare sand with mixed algae	26.65
Bare sand	1108.19
Total sand	**1135**
Coral reef	84.89
Gorgonian reef	14.79
Patch reef	86.18
Total reef	**186**
Lobophora fields	240.94
Macro algae fields	412.51
Sargassum fields	150.77
Total algae	**804**
Neogoniolithon-Lyngbya habitat	3.17
Cyanobacteria fields	13.31

Assessment of Future Prospects

The outlook for coral reefs around the world is looking grim (Core Writing Team, 2023). Under current emission trajectories exceeding 1.5 °C in temperature increase is more and more becoming a realistic possibility (Möller et al., 2024) increasing the probability of triggering climate tipping points (Armstrong McKay et al., 2022). For the Caribbean coral reefs the situation may even be worse as there is limited functional redundancy since there are much less coral species in the Caribbean than for example in the Indo-Pacific (McWilliam et al., 2018). Furthermore, the Caribbean Sea is surrounded by densely populated countries and islands with limited possibilities to reduce their impact on the marine environment. At the same time global pressures are increasing as a result of climate change. Marine heatwaves in the Caribbean are predicted to increase in duration and intensity to such an extent that by 2100, heat wave conditions may have become very common (Bustos Usta et al., 2024). This is likely to also increase hurricane intensity and frequency. Increasing ocean acidification is likely to negatively influence coral functioning (Williams et al., 2024), but exact effects remain unclear and are probably species specific. Many of these effects may be worsened by weakening of the Atlantic Meridional Overturning Circulation (AMOC) which may also lead to changes in rainfall patterns, accelerated sea level rise, and disrupted ocean currents (Pontes and Menviel, 2024). Collapse of the AMOC has recently been predicted mid-century under the current scenario of future emissions (Ditlevsen and Ditlevsen, 2023).

The year 2023 was the warmest on record, and 2024 is headed to surpass 2023 (Figure 1 and 2). During 2024 sea surface temperatures remained much warmer than average and bleaching of corals (Figure 11) is predicted to continue till the end of 2024 which will cause additional coral mortality. In September 2023 the world global temperature reached 1.5 degrees Celsius-increase above pre-industrial levels (1850-1900) (Copernicus, 2023) and there are no signs that our CO_2 emissions are decreasing. For Bonaire specifically the local threats appear to be mounting as the population, now around 24 thousand, is estimated to grow to 30 to 50 thousand by 2050 (CBS, 2023) and 80% of sewage produced on the island is estimated to enter the coastal zone through non-working septic tanks and cesspits and not through the sewage treatment plant (Haskoning, 2023). Even at the governmental level the environmental risks of further population growth is not acknowledged, as the governments of Bonaire and the Netherlands have agreed to facilitate further growth of the population (Rijksoverheid, 2024). Population growth has always been one of the strongest indicators of reef degradation, mostly linked to a decrease in water quality (Cramer et al., 2020). Together with rising sea levels, increasing sea water temperature and acidification, water quality is likely to seal the fate of Bonaire’s reef. Only a policy directed at improving water quality and limiting terrestrial runoff together with strict enforcement and combined with active restoration of corals and herbivores may be able to improve the condition of Bonaire’s reef (Van der Geest et al., 2020). But even then, the outcome is unsure in the face of failing international attempts to halt climate change. However, not to act should not be an option, given our responsibility towards future generations. The NEPP for the Caribbean Netherlands assigns a high priority to addressing water quality issues (both in terms of eutrophication and sediment runoff due to erosion) (Min. LNV et al., 2020).

Comparison to the 2018 State of Nature Report

Overall, the CS of the coral reef habitats of the Caribbean Netherlands has measurably worsened compared to the 2018 assessment, especially due to climate change effects and the effects of the Stony Coral Tissue Loss Disease (Table 7).

Table 7. *Overview of the status of the coral reefs of the Caribbean Netherlands with respect to different ecological aspects.*

Aspect coral reefs	**2024**
Distribution	Favourable
Surface area	Favourable
Quality	Unfavourable-inadequate
Future prospects	Unfavourable-bad
Overall Assessment of Conservation State	**Unfavourable-bad**

Figure 11. *Bleaching in November 2024. Image by T. Kemenes van Uden.*

Recommendations for National Conservation Objectives

National long-term goals
The target for achieving a favourable Conservation State for corals is an increase in the cover of scleractinian corals, mainly to be achieved by creating local environmental conditions that support growth and survival of corals and possibly supported by restoration from coral nurseries. At this moment it is unclear if these will be sufficient in view of the wider regional and global effects on the state of the Caribbean Sea, however, by improving local conditions it is hoped that coral reefs will start recovering.

National short-term (5-year) goals
Improvement of the quality and resilience of corals by addressing numerous local threats such as terrestrial sediment run-off, eutrophication, pollution, and overfishing of groupers and snappers.

Key Threats and Management Implications

Table 8. *Overview of the main threats to coral reefs of the Caribbean Netherlands and implications for management.*

Main threats		**Management actions**
Climate change	Increase in frequency and intensity of storms, hurricanes and heat waves, rising sea surface temperatures.	• Reduce the impact of local stressors, so that coral reefs are more resilient to the impacts of climate change.
Eutrophication	Eutrophication, nutrient enrichment, from runoff and sewage leakage, causes diseases, bacterial and algal overgrowth of corals.	• Prevent runoff and eutrophication as much as possible by creating an integral water and spatial management plan which should include vegetated buffer zones. • Phase out cesspits and control leakage from septic tanks. • Limit population growth.
Coastal erosion	Land degradation (loss of vegetation), particularly due to overgrazing by goats and donkeys, leads to erosion and terrestrial sediment input (runoff) into the sea.	• Reduce overgrazing and implement active management of livestock. • Reduce coastal construction and industrial activity in upstream drainage areas of coastal bays and saliñas. • Create and manage vegetated buffer zones that catch runoff and prevent sediments from reaching the sea.
Pollution	Waste is not processed but dumped on land, much of which eventually ends up in the sea. This includes chemicals and metals that negatively impact corals and other marine life.	• Implementation of a waste management system according to Dutch standards.
Overfishing	Fish populations of groupers and snappers are still overfished. This decreases the health status of the coral reefs.	• More strict supervision and law enforcement

Data Quality and Completeness

The status of the coral reefs on the leeward side of Bonaire to a depth of approximately 10m is relatively well known. Much less is known about deeper areas of the reef and the windward side of the island. There is no structured monitoring system (e.g. Statutory Research Tasks) in place. Coral reef monitoring on all islands can also be improved by structural funding.

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11 Conservation State of the Open Sea and Deep Sea of the Dutch Caribbean

Debrot, A. O. 2025. From: State of Nature Report for the Caribbean Netherlands, 2024. WUR report C001/25.

Status

The open sea and deep sea of the Caribbean Netherlands encompass three categories of waters, as defined by the UNCLOS (United Nations Convention on the Law of the Sea):

- The waters within the 12-nautical mile zone of "territorial waters," for which primary responsibility belongs to the respective island. Ultimately, the Kingdom of the Netherlands also carries final responsibility for the territorial waters.
- The "contiguous zone," from 12 to 24 nautical miles which is part of the EEZ.
- The waters from 24 to a maximum of 200 nautical miles, referred to fully as the "Exclusive Economic Zone" (Meesters et al., 2010).

The last two zones fall directly under jurisdiction of The Netherlands with no role for the islands. All three zones are subject to various international treaties, to which the Netherlands is bound, but the the national fishing legislation only applies to the EEZ and not to the territorial waters. Since 2015, the sea areas surrounding Bonaire, Saba, and the Saba Bank have been designated as marine mammal and shark sanctuaries, known as the Yarari - Marine Mammal and Shark Sanctuary. The name Yarari comes from the Taíno language spoken by pre-Columbian indigenous peoples and means "a fine place." Legislation has been developed to implement this status effectively (Overheid.nl., 2023).

Characteristics

In the literature, the open sea and deep sea are divided into five depth zones, primarily based on the amount of light. These are:

- Epipelagic zone (also known as the photic zone) where enough light exists for active photosynthesis. This zone extends to 200 meters in depth, which corresponds to the formal depth of the "continental shelf."
- Mesopelagic zone, between 200 and 1,000 meters, where the light is insufficient for photosynthesis.
- Bathypelagic zone, from 1,000 to 4,000 meters in depth, where temperatures drop to 4°C, there is no sunlight, and energy comes from material falling from above.
- Abyssopelagic zone, between 4,000 and 6,000 meters deep, where there is no light, almost no food sources, water remains at 4°C, and water pressure is extreme.
- Hadalpelagic zone, deeper than 6,000 meters, also known as the "trench zone." These depths often suffer from a lack of oxygen, and biodiversity is very low.

Within the Caribbean Netherlands, only the last zone may not be present. The Gebco (General Bathymetric Chart of the Oceans) underwater map of the Caribbean Sea shows depths of over 5,500 meters in the Venezuela Basin north of Bonaire, so it cannot be ruled out that depths greater than 6,000 meters may eventually be documented for the Dutch Caribbean EEZ (https://www.gebco.net/).

In this overview, no distinction is made between the different deep-sea zones. All waters deeper than 100 meters are referred to here as "deep sea" or "open sea."

Primary productivity through photosynthesis (mainly by the blue-green algae *Trichodesmium*; Castro and Huber, 2010) is crucial for biodiversity and biomass. The Caribbean region is considered a Class II moderately productive system (150-300 g C m-2 yr-1, Heileman and Mahon, 2009), but productivity varies greatly in time and space. The highest productivity is found off the coast of Venezuela, east of Bonaire, where it is around 500 g C m-2 yr-1 (Couper, 1983; Richardson and Young, 1987; Tyler, 2003, in Couperus et al., 2014). This is primarily due to wind-driven upwelling of deeper nutrient-rich waters from January to May (Rueda-Roa and Muller-Karger, 2013), a system known as the southern Caribbean upwelling system. Thanks to the location of this upwelling region, the southern Caribbean region generally experiences lower sea surface temperatures throughout the year, which seems to protect coral reefs in this part of the Caribbean (including Bonaire, Curaçao, and Aruba) from the global phenomenon of coral bleaching caused by rising sea surface temperatures (Eakin et al., 2010).

North of Bonaire and extending to the Saba Bank lies the Venezuela Basin, which often receives cold deep water from the more western Colombia Basin via the "Aruba Gap," which, at a depth of 4,078 meters, forms the deepest entrance to the Venezuela Basin. The southern margin of the Venezuela Basin, north of Bonaire, is formed by the Curaçao Ridge, an active subduction zone where the Caribbean plate is being pushed under the South American plate (Matthews and Holcombe, 1985).

The mesophotic coral reefs found at depths between 30 and 150 meters are highly valuable. Up to 60 meters, these communities share many species with shallower coral reefs. Below 60 meters, the communities are dominated by sponges, some of which grow very slowly and can live for 500-1,000 years, horn corals, and algae that are not typically seen on shallow reefs, along with specialized fish fauna (Slattery et al., 2011).

On these deep reefs, the primary reef builders are calcifying encrusting red algae sometimes found as "rhodoliths" (Becking and Meesters, 2014), as well as small coral species (Vermeij et al., 2003). These reefs may serve as refuges for many fish species that may move back and forth between depths but even for coral or sponge populations that are normally found along a wider depth range (Lesser et al., 2009). Many species regularly move between shallow and deep reefs daily or at different stages of their life cycle (Slattery et al., 2011). Since little is known about them, reef communities deeper than 60 meters are considered part of the "deep sea."

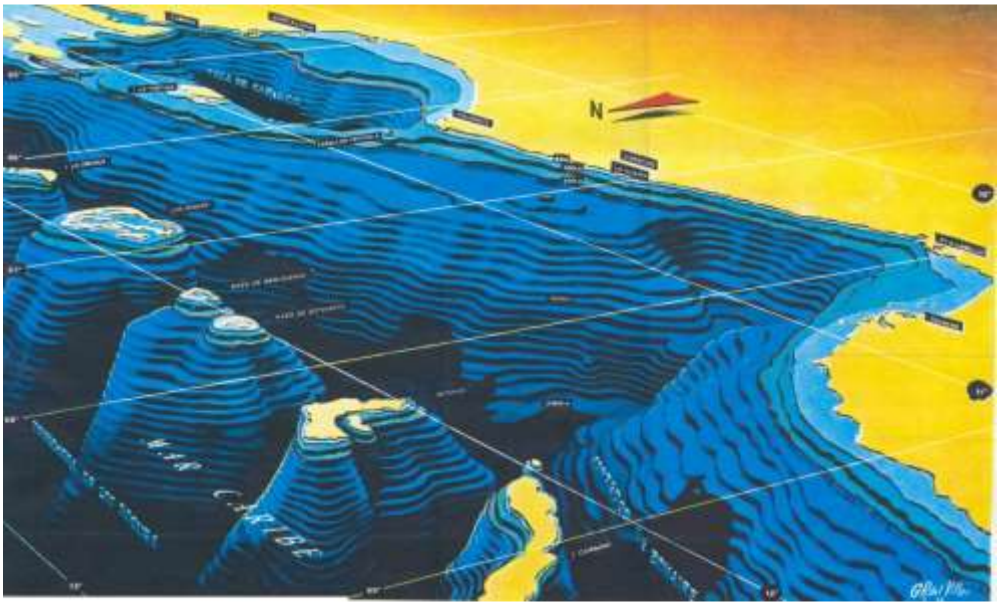

Figure 1. *Deep-sea profile around Bonaire and the island chain off the coast of Venezuela (Smith et al., 2002).*

Relative Importance Within the Caribbean Region

The “open sea and deep sea” area in the Caribbean Netherlands is only a small part of this habitat within the Caribbean region. However, an analysis of the distribution of known fish and fisheries-relevant invertebrates across the entire Caribbean reveals that the habitat around Bonaire is part of the second-richest Caribbean hotspot of marine biodiversity, with high species richness and high endemism. This hotspot includes the sea area around the island chain north of Venezuela—Aruba, Curaçao, Bonaire—and the northern coast of Venezuela and Colombia (Fig. 2; Smith et al., 2002). Since little is known about this biodiversity, particularly in deep waters, it is expected that further research will reveal many new species, including many endemic ones.
This expectation is supported by an exploratory submersible survey of mesophotic reefs down to 300 meters off Bonaire, which found at least 15 species (shrimp, sponges, and fish) that were previously undescribed (Becking and Meesters, 2014).

Deep waters around the islands of Saba and St. Eustatius and the Saba Bank also host much unique, undescribed biodiversity. In April 2017, at least eight previously undescribed fish species were collected in five deep-sea dives around St. Eustatius, including two gobies that seem unique to St. Eustatius and/or nearby islands (Bert Hoeksema, pers. comm.). Additionally, 38 new fish records were found for St. Eustatius.

The pelagic and deep-sea habitat is vast compared to human-induced disruptive and polluting factors, mostly originating from land. This provides the system with some resilience. However, it remains vulnerable to climate change, which can affect the fundamental processes of temperature and current patterns (both horizontal and vertical) and other global influences, such as increased nutrient concentrations, reduced oxygen concentrations and ocean acidification, which can have significant effects on ecosystem values and services. For instance, changing climatological conditions will considerably increase the risk of hypoxia due to a combination of increased stratification, reduced solubility of oxygen, and enhanced metabolic rates (Meire et al., 2013), and worldwide oxygen concentrations of surface waters are steadily declining (Li et al., 2020). This can potentially have a great impact on larval fish, plankton and other organisms of the surface waters. Ditlevsen and Ditlevsen (2023) even see the collapse of the Atlantic meridional overturing circulation as unavoidable which will cause unprecedented climatic disruption with very serious ecological consequences. In addition, Johns et al. (2014) have found that the Caribbean sometimes experiences large plumes of fresh(er) Amazon River water and that this is associated with lower larval densities of coral reef fishes. Hence, this factor may also prove to be a critical factor possibly affecting coral reef fish recruitment but also offshore ecology in the region and is deserving of further study.

Definition of habitat
For this overview, no distinction is made between the different depth zones of the deep sea. All waters deeper than 100 meters are classified here as "open sea and deep sea."
Quality requirements
Tables 1 and 2 provide insight into the abiotic conditions and typical species of pelagic (open sea and deep sea) areas.

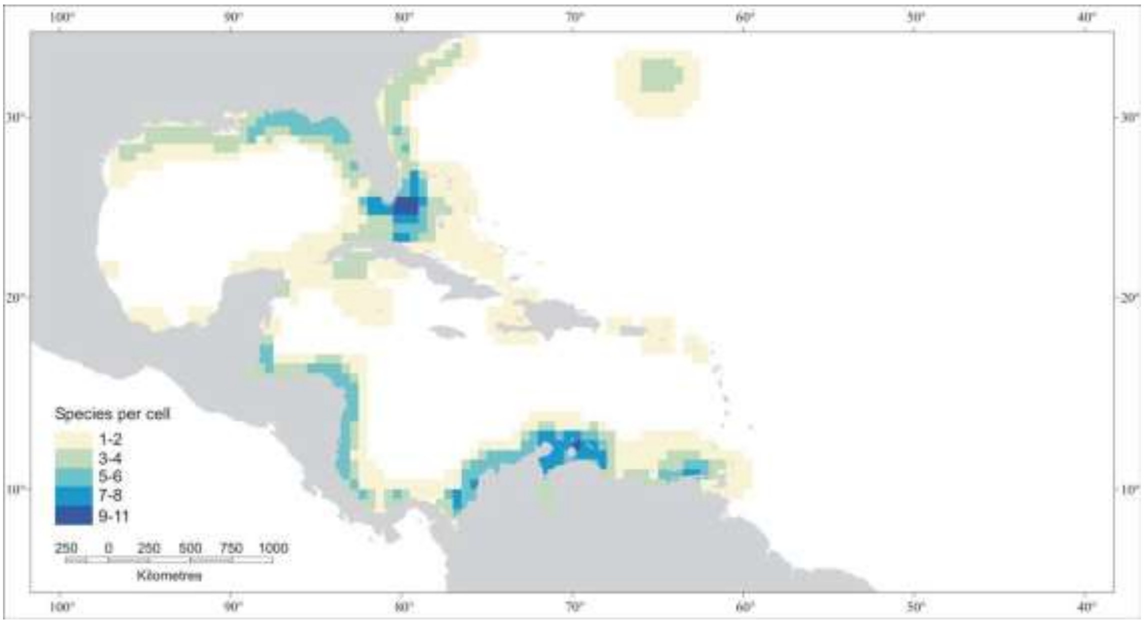

Figure 2. *Combined distributions of 75 endemic fish species (green arrow pointing to Bonaire and the concentration area in the southern Caribbean) (Smith et al., 2002).*

Abiotic parameters

Table 1. *Overview of principal abiotic environmental conditions in the open sea and deep sea of the Dutch Caribbean.*

Depth	Hadalpelagic 4,000-6,000	Bathypelagic 1,000-4,000	Mesopelagic 200-100 m	Epipelagic surface to 200m
Light	Hadalpelagic: no light	Bathypelagic: No light	Mesopelagic: Insufficient light	Epipelagic: Sufficient light
Temperature	Hadalpelagic Cold; 4 °C	Bathypelagic Cold; 4 °C	Mesopelagic Cool; Thermocline 13-4 °C	Epipelagic Warm; 28-13 °C
Waves	Hadalpelagc: None	Bathypelagic: None	Mesopelagic: None	Epipelagic: High mixing to 100 m
Salinity	Hadalpelagic 35 ppt	Bathypelagic 35 ppt	Mesopelagic 35 ppt	Epipelagic 36 ppt

Typical species

Table 2. *Typical species for the open and deep sea of the Dutch Caribbean.*

English Name	Scientific name	IUCN category	Depth zone	Species group
Spinner Dolphin	*Stenella longirostris*	LC	epi- mesopelagic	marine mammals
Bryde's Whale	*Balaenoptera edeni*	LC	epipelagic	marine mammals
Humpback Whale	*Megaptera novaeangliae*	LC	epipelagic	marine mammals
Sperm whale	*Physeter macrocephalus*	VU	bathypelagic	marine-mammals
Pilot Whale	*Globicephala marcrorhynchus*	LC	epi- mesopelagic	marine-mammals
Leach's Storm-petrel	*Oceanodroma leucorhoa*	VU	epipelagic	birds
Black-capped petrel	*Pterodroma hasitata*	EN	epipelagic	birds
Brown Booby	*Sula leucogaster*	LC	epipelagic	birds
Audubon's Shearwater	*Puffinus lherminieri*	LC	epipelagic	birds
Red-billed Tropicbird	*Phaeton aethereus*	DD	epipelagic	birds
Oceanic White-tip	*Carcharhinus longimanus*	VU	epipelagic	sharks
Tiger Shark	*Galeocerdus cuvieri*	NT	meso-epipelagic	sharks
Cuban Dogfish	*Squalus cubensis*	LC	mesopelagic	sharks
Bluntnose Sixgill shark	*Hexanchus griseus*	NT	meso-bathypelagic	sharks
Cookie-cutter Shark	*Isistius brasiliensis*	LC	bathypelagic	sharks
Whale Shark	*Rhincodon typus*	EN	epipelagic	sharks
Blue Marlin	*Makaira nigricans*	VU	epipelagic	fish
Dorado	*Coryphaena hippurus*	LC	epipelagic	fish
Blackfin tuna	*Thunnus atlanticus*	LC	epipelagic	fish
Big-eye Tuna	*Thunnus obesus*	VU	epipelagic	fish
Wahoo	*Acanthocybium solandri*	LC	epipelagic	fish
Rainbow Runner	*Elagatis bipinnulata*	LC	epipelagic	fish
Oilfish	*Ruvettus pretiosus*	LC	mesopelagic	fish
Fourwing Flyingfish	*Hyrundichthys affinis*	DD	epipelagic	fish
Vermillion Snapper	*Rhomboplites aurorubens*	VU	meso- epipelagic	fish
Diamondback Squid	*Thysanoteuthis rhombus*	LC	meso-epipelagic	molluscs
Sargassum	*Sargassum fluitans*	LC	epipelagic	seaweed
Trichodesmium	*Trichodesmium sp.*	DD	epipelagic	cyanobacteria

Other Characteristics of Good Structure and Function

Healthy and resilient open sea and deep-sea areas provide important ecosystem services, such as:

A stable and favourable climate;

- Suitable nursery areas for the larval stages of most coral reef species (corals, fish, molluscs, crustaceans, sponges, algae) (e.g., Wells and Rooker, 2004; Witherington et al., 2012). For example, the floating *Sargassum* seaweed, which is crucial as a nursery habitat for many pelagic species;
- Healthy fish stocks for commercial exploitation (Couperus et al., 2014);
- Foraging areas for endangered seabirds such as the endangered Black-capped Petrel (*Pterodroma hasitata*, EN), and the Leach's Storm-petrel (*Oceanodroma leucorhoa*, VU) (Prins et al., 2009; Poppe, 1974; Debrot et al., 2020); migration and living areas for endangered marine mammals (Debrot et al., 2011) and sharks (van Beek et al., 2014).
- Environmental Quality Requirements

For a healthy open sea and deep-sea environment, it is necessary to limit pollution and reduce noise pollution from shipping and geological exploration.

Current Presence and Reference Values

Most of the Caribbean Netherlands and other Exclusive Economic Zone (EEZ) areas (of Aruba, Curaçao, and St. Maarten) consist of open sea and deep-sea habitats (Table 2). Within the historical period, no changes have occurred in the area or coverage of this habitat. However, it is increasingly subject to disturbance, pollution, and climate change. These processes can significantly and permanently affect the quality of this habitat.

Assessment of National Conservation State

Trends and Recent Developments:

- A clear increasing trend of warmer surface waters (Eakin et al., 2010);
- Dramatically increasing pollution of the seabed (Debrot et al., 2014) and sea surface (Law et al., 2010);
- Regionally higher surface water temperatures leading to massive coral bleaching (including reefs deeper than 60 meters) (Eakin et al., 2010);
- Increasing regional outbreaks of massive *Sargassum* blooms (especially the fully pelagic species *S. fluitans* and *S. natans*). This phenomenon began around 2011 and seems to be worsening. Possible causes may include higher sea temperatures, changing ocean currents, and/or nutrient enrichment of surface waters from major rivers (Mississippi, Amazon, Orinoco), coastal agriculture (fertilizer), and wastewater from increasing tourism and coastal development, or possibly even dust particles carried from Africa. However, the impacts of the pelagic blooms on open-ocean pelagic nutrient dynamics, species aggregations that use *Sargassum* and input of nutrients to the deep sea are unknown.

Distribution Assessment: Favourable
The "open sea and deep sea" habitat is the most common marine habitat in the Caribbean Netherlands and surrounds the three islands and the Saba Bank at short distances. Due to the steep bathymetry of the islands, this habitat is located just a few hundred meters offshore.

Area Assessment: Favourable
The current area of "open sea and deep sea" within the Caribbean Netherlands is significant (approximately 22,404 km²). The Kingdom is also fully responsible for the EEZ areas associated with Curaçao, Aruba, and St. Maarten (i.e., the entire EEZ area in the Dutch Caribbean islands). In terms of area, it is the largest habitat within the Caribbean part of the Kingdom, covering approximately 81,000 km².

Quality Assessment: Unfavourable-inadequate

Abiotic Conditions:
Trends or developments in the abiotic conditions of the open sea and deep sea, and how these will affect species and ecological processes, are largely unknown. Acidification and warming of surface waters are likely to have major consequences in the coming years.

Typical Species:
Effects on typical species are insufficiently known. Little research has been conducted or knowledge gathered about the ecology of tropical pelagic and deep-water species.

Other Characteristics:
The "open sea and deep sea" is affected by pollution from litter, warming of surface layers, and anthropogenic noise from shipping and exploration. Many commercially important fish stocks of large migratory predators are overfished. There is little knowledge of many species that have not yet been overfished or may still be unaffected.

Assessment of future prospects: Unfavourable-inadequate
A recent literature review shows that little concrete information is known about the "open sea and deep sea" of the Caribbean Netherlands (van Beek, 2016). Some new work for the Caribbean Netherlands has been done on oceanic eddy formation (Van der Boog et al., 2019) and on the offshore distribution of seabirds (Debrot et al., 2020). Limited deep-sea dives and collections from past expeditions indicate that it houses much undescribed and possibly unique biodiversity. The area is also known as the habitat of many commercially important fish species and fish stocks that have not yet been fished, representing a potential source for future economic exploitation. However, too little is known about the deep sea and associated pelagic fish stocks to make any concrete statements about the future prospects. Overfishing is already a concern for several fish stocks and is a looming threat for newly targeted fish stocks. The exploitation of new species should be preceded by a thorough ecological assessment. For example, the ecological role of the Diamondback Squid as food for sperm whales must be carefully considered before this species is commercially exploited. The deep sea will not escape the large-scale effects of climate change. The warming of surface waters due to climate change is ongoing, and this is likely to have significant negative consequences for ecosystem values and functions.

Table 3. *Summary overview of the status of the open sea and deep-water habitat of the Dutch Caribbean in terms of different conservation aspects.*

Aspect open sea and deep sea	2024
Distribution	Favourable
Surface area	Favourable
Quality	Unfavourable-inadequate
Future prospects	Unfavourable-inadequate
Overall Assessment of Conservation State	**Unfavourable-inadequate**

Comparison to the 2018 State of Nature Report

Overall, in comparison to the 2018 assessment, no major changes can be meaningfully identified for the CS of the deep and open sea habitats of the Caribbean Netherlands. However, the unquestionable ecological impacts of measurable ocean warming remain largely unassessed.

Recommendation for National Conservation Objectives

Long-term goals
The vision for a favourable Conservation State is the preservation and safeguarding of functioning deep-sea ecosystem values for the islands. A conservation goal is proposed as a long-term objective.

Short-term (5-year) goals

- Conduct research to fill the knowledge gap on the functioning of the deep sea to enable scientifically based management,
- Conduct research to map economically promising, previously unexploited fish stocks, and

- Conduct research to identify and map the rich biodiversity of the deep sea, particularly in the regional hotspot area of Bonaire, but also around the Saba Bank (e.g., van Soest et al., 2014).

Table 4. *Overview of main threats to the open sea and deep-sea habitats of the Dutch Caribbean and implications for management.*

Core threats		**Management interventions**
Pollution	Soil and water pollution, as well as litter, are carried from the coast to deeper habitats and the deep sea. Existing oil industry installations cause both chronic and periodic large-scale oil spills.	• Implement measures to control terrestrial pollution sources and litter.
Overfishing	Many commercially important deep-sea fish species that primarily inhabit the upper 200 meters are highly migratory and are overfished on a regional scale. Fish stocks in the deeper layers of the open sea grow slowly, making them particularly vulnerable to fishing.	• Participate in regional initiatives for research and management. • Conduct basic research and apply precautionary management for the exploitation of deep-sea fish.
Disturbance and noise contamination	Disturbance and noise pollution are caused by the rapidly increasing maritime traffic, recreational boating, and seismic surveys for oil exploration or geological studies.	• Research is needed to assess the current levels and effects of maritime disturbance. Based on the insights gained from this: • Zoning • Threshold values • Monitoring and law enforcement
Climate change	Potential loss of plankton and juvenile stages of coral reef organisms (due to acidification and warming), shifting migration patterns of fish and other animals, and potential changes in upwelling patterns.	• Actively participate in forums and research related to climate issues.

Data Quality and Completeness

Very little research has been conducted on the deep sea and deep coastal waters, and almost nothing is known about the functioning of the deep-sea system. All indications suggest that deep-sea habitats are home to many unique and yet-to-be-described species and that the deep habitats are closely intertwined with the much better-known shallow marine habitats near the coast.

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Part 2: Species and Species Groups

The Caribbean Netherlands is part of the habitat of at least 143 species of international policy relevance (Annex 1). These include species listed on the IUCN Red List, SPAW, CMS, and CITES (see Annex I). Additionally, there are many species locally threatened and/or of local ecological and economic importance, for which periodic insight into their status is required to adequately inform nature policy plans. Examples of these are two rare endemic plants of St. Eustatius (the rediscovered St. Eustatius Morning Glory, Howard and McDonald 1995, and the newly discovered *Gonolobius aloensis*, Krings and Axelrod, 2013), the newly described endemic palm of Bonaire, *Sabal looghidiana* (Griffith et al., 2019) and the list of protected plants of Bonaire (Annex II). The Caribbean Netherlands is home to approximately 130 endemic species. Unfortunately, due to a structural knowledge and monitoring deficit, there is insufficient information or data available for most important species to conduct in-depth analyses of their state of conservation or their future prospects.

In this chapter, and for the purposes of this second report on the state of nature in the Caribbean Netherlands, several groups and/or species groups are highlighted where sufficient information is available to make a substantiated assessment. There are certainly more species and/or species groups for which enough is known to allow for meaningful reporting, even if only in summary form (birds of prey and rare breeding birds at the group level). Our selection is purely pragmatic, based on the quality and availability of data and on available funding.

This report focuses on 14 species and species groups (specifically, the orchids of Saba and St. Eustatius, endangered trees of Bonaire, the land snails of the Caribbean Netherlands, the butterflies of the Caribbean Netherlands the Lesser Antillean Iguana, the Saba Green Iguana, the Red-billed Tropicbird, the Bridled Quail-dove, the nesting terns, bats, sea turtles, elasmobranchs, deep-sea fish fauna and fish stocks of the islands. All these species and/or species groups are key monitoring priority species (Verweij et al., 2015) and are among those for which the most pressing research questions are posed and/or for which there are biodiversity obligations (Jongman et al., 2009).

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12 Conservation State of the Orchids of Saba and St. Eustatius

Debrot, A. O., Boeken, M. and van der Wal, J. T. 2025. From: State of Nature Report for the Caribbean Netherlands, 2024. WUR report C001/25.

Status

Even though orchids are extremely diverse and widespread, so far only about 1,000 species have been assessed for the IUCN Global Red List (IUCN, 2017). Of these, 56.5% fall into one of the categories of serious concern (critically endangered, endangered and vulnerable). Orchids are also a conspicuous component of the biodiversity of the Dutch Caribbean islands of Saba and St. Eustatius. Bonaire was not included as it hardly has any orchids. Early floral assessments of the group were given by Boldingh (1909), Stoffers (1962) and Garay & Sweet (1974). More recently Feldmann (2012) published an orchid checklist for the Lesser Antilles. However, aside from some relatively recent floristic accounts (e.g., Boeken, 2014) and one quantitative yet short-term assessment of population trends in a single species (Ackerman et al., 2020), until now, practically nothing has been known about the status of this important and vulnerable component of the native biodiversity of the Dutch Caribbean. Of the orchids of Saba and St. Eustatius, only one invasive species is presently listed as being of "Least Concern" and one island endemic is listed as "Near Threatened". For none of the other Lesser Antillean range restricted species has the global Conservation State been assessed and no information is available on the status of any of those species on any of their other islands of occurrence. The Conservation State of plants in general in the Caribbean is very poorly known (Torres-Santana et al., 2010), and this is no different for the Dutch Caribbean.

Table 1. *Occurrence, relative abundance and protection status of the 38 orchid species (Orchidaceae) documented from the wild on Saba and St. Eustatius, Caribbean Netherlands. Status: N = native, R = restricted range (i.e., endemic), I = invasive, - = absent. Island occurrence: A = abundant, C = common, P = patchy, R = rare, V = very rare. Substrate: T = terrestrial, A =arboreal (live), D = arboreal (dead), L = lithophilic. IUCN status: LC = least concern, NT = near threatened, blank = data deficient. CITES II regulated: X Legal status: S = protected in Saba, E = protected in St. Eustatius. Occurrence by veg. zone: L = lowlands, E= evergreen forest, R = rainforest, H = "fog" forest a.k.a. elfin forest.*

Species	Synonym; name in literature	Status	Occurrence SAB	Occurrence EUX	Substrate	IUCN status	CITES II	Island legal status	Vegetation zone	Reference Saba
Brassavola cucullata (L.) R. Brown 1813		N	C	P	A,D,L		X	S, E	L, E	1
Cranichis muscosa Swartz 1788		N	R	-	T		X	S	H	1
Cyclopogon cranichoides (Grisebach) Schlechter 1920	*Spiranthes cranichoides*	N	V	-	L		X	S	E	1
Cyclopogon elatus Sw. (Schltr.) 1919	*Spiranthes elata*	N	-	V	T		X	E	R	

Species	Synonym; name in literature	Status	Occurrence SAB	Occurrence EUX	Substrate	IUCN status	CITES II	Island legal status	Vegetation zone	Reference Saba
Epidendrum anceps Jacquin 1763		N	C	P	A,D,L		X	S, E	E, R	1
Epidendrum antillanum Ackerman & Hágsater 1992		N	V	-	A		X	S	R, H	1,5
Epidendrum ciliare L. 1759	*Coilostylis ciliaris*	N	A	C	A,D,L		X	S, E	L, E	1
Epidendrum difforme Jacquin 1760		R	R	P	A		X	E	R, H	1
Epidendrum mutelianum Cogn. 1910	name in lit: *Epidendrum sp.*	N	R	-	A		X		R, H	1
Epidendrum nocturnum Jacquin 1760		N	V	-	A,D,L		X		E, R	1,3
Epidendrum patens Swartz 1806		R	V	-	D		X	S	H	1
Epidendrum strobiliferum Reichenbach f. 1859		N	V	-	A		X	S	R	1,5,7
Habenaria monorrhiza (Swartz) Reichenback 1885		N	V	-	T		X	S	H	1
Jacquiniella globosa (Jacquin) Schlechter 1920		N	V	V	L		X	S, E	R	1,7,8
Liparis nervosa (Thunb.) Lindl. 1830		N	-	V	T		X	E	R	
Malaxis massonii (Ridl.) Kuntze 1891	*Microstylus massonii*	N	V	-	T		X		H	4
Malaxis spicata Swartz 1788	*Microstylus spicata*	N	V	-	T		X		H	1
Mesadenus lucayanus (Britton) Schlechter 1920	*Spiranthes lucayana*	N	R	V	T		X	S, E	E	1
Microchilus familiaris Ormerod 2009		N	R	V	T		X		R, H	1
Microchilus hirtellus (Swartz) D. Dietrich 1852	*Erythrodes hirtella*	N	R	V	T		X	S, E	E	1
Microchilus plantagineus L. D. Dietrich 1852	*Ertyhrodes plantaginea*	N	P	V	T		X	E	R, H	1
Octomeria graminifolia (L.) R. Brown 1813		N	V	V	A		X		E	1
Oeceoclades maculata (Lindley) Lindley 1833		I	C	R	T	LC	X	S	E	1
Ornithidium coccineum (Jacquin) Salisbury ex R. Brown 1813	*Maxillaria coccinea*	N	R	-	A,(L)		X	S	R, H	1
Ornithidium inflexum (Lindley) Reichenbach f. 1864	*Maxillaria inflexa*	R	V	-	A		X	S	H	1
Polystachia concreta (Jacquin) Garay & Sweet 1974	name in lit: *P. foliosa*	N	V	R	A,D		X	S, E	L, E	1
Ponthieva petiolata Lindley 1824		R	P	-	T		X	S	H	1
Ponthieva racemosa (Walter) C. Mohr 1901		N	V	-	T		X		H	1

Species	Synonym; name in literature	Status	Occurrence SAB	Occurrence EUX	Substrate	IUCN status	CITES II	Island legal status	Vegetation zone	Reference Saba
Prescottia oligantha (Sw.) Lindley 1840		N	-	R	T		X		R	
Prescottia stachyodes (Sw.) Lindl. 1836		N	-	V	T		X		R	
Psilochilus macrophyllus (Lindley) Ames 1922		N	V	-	T		X	S	H	1
Psychilis correllii Sauleda 1988	*Epidendrum kraenzlinii*	R	C	C	L		X	S, E	L	1
Sacoila lanceolata (Aublet) Garay 1982	*Spiranthes lanceolata*	N	P	V	T		X	S	L, E	1
Spathoglottis plicata Blume 1815		I	R	-	T		X	S	H	1
Tetramicra elegans (Hamilton) Cogniaux 1910	name in lit: *T. canaliculata*	N	V	C	T		X	S, E	E	1
Tolumnia prionochila (Kraenzlin) Braem 1986	*Oncidium prionochilum*	N	V	V	L	N T	X	S	E	1
Tolumnia urophylla (Lodd. Ex Linl.) Braem 1986	name in lit: *Oncidium. variegatum*	R	V	R	T		X	S, E	E	1,3
Triphora surinamensis (Lindlley Ex Benth.) Britton 1924		R	V	-	T		X	S	R	1

Characteristics

Description:
Orchids are one of the earliest higher plant families to adopt a unique and complex yet successful life cycle with intimate ties to a combination of fungi, trees, and insects. They are one of the most widespread plant families (Givnish et al., 2016; Charitonidou et al., 2021) and amount to more than 28,000 species or about 8% of the total angiosperm species diversity (Willis, 2017). Thanks to their amazing diversity in form, colour, and life cycle they have always attracted special interest from science and floriculture. Due to their life cycle complexities and great dependencies on other organisms during their various life phases, they are believed to be especially vulnerable to climate change (Swarts and Dixon, 2009, Seaton et al., 2013; Fay, 2018). As such, they can also be uniquely suitable as early indicators of ecosystem change (Akhalkatsi et al., 2014; Newman et al., 2015).

Relative Importance within Caribbean:
There are approximately 700 orchids that are native to the Antilles (Trejo-Torres et al., 2003) of which some 130 species are found in the Lesser Antilles (Feldmann, 2012). Partly because of their tendency towards wind-dispersal by means of tiny "dust" seeds the Lesser Antilles only has five species that are fully endemic to a single (large) island. Even so, 27% of the Lesser Antillean native orchids are endemic to the Caribbean and 16% are endemic to the Lesser Antilles (Feldmann, 2012). A total of 38 orchids are found in the wild on Saba and St. Eustatius, of which two are invasive, 36 are native, and six of these additionally are "range restricted" (i.e., endemic to a small group of surrounding islands).

Ecological Aspects

Orchids have two kinds of roots; "normal" roots and aerial roots. Because of the latter, they often can grow om structures that do not accumulate soil (like rocks or branches of trees). In this, certain orchids adopt a typical terrestrial mode of life while others a more lithophilic or arboreal mode of life. On Saba and St. Eustatius all three life-history modes can be found (Table 1). In addition, (almost) all are further wind-dispersed by means of dust-seeds and can appear in unexpected places when their fine seed is transported over long distances by the wind. This means that orchids may suddenly appear in unexpected places or islands and run the risk of being erroneously labelled as non-native. Because many orchids look a great deal alike (especially when not flowering) and because many may live in a dormant phase (underground) for long periods, identifying and quantitatively surveying orchids can be extremely challenging. This has also frequently led to the misnaming or misidentifying of orchids (Boeken, 2014; Feldmann, 2012). When this is combined with numerous revisions of the nomenclature of various groups, it means that species lists need to be reviewed and updated frequently in order to purge such errors or multiple counting of the same species under different names. After correcting and "cleaning" the orchid lists of misnomers and misidentifications for Saba and St. Eustatius (Debrot et al, in prep.), the total corrected list of species amounts to 38 species for both islands combined.

The compiled list is based on a review of literature and field surveys done in 2022. In 2022, 19 native or naturalized orchid species were found on Saba while 15 other species confirmed previously were not encountered. For St. Eustatius, only seven native or naturalized species were confirmed in 2022, while 14 other species confirmed previously, were not encountered. The reason many known or previously-known species could not be found during our 2022 visits may be several among which: a) their general rarity, b) disappearance from the island or c) due to their differing phenology — which may include a subterraneous or otherwise inconspicuous vegetative phase. Hence, the total orchid flora for Saba can be considered 34 species of which 32 are native species and two are naturalized invasive species (Table 1). For St. Eustatius, the total valid known flora of 21 species was composed of 20 native species and one naturalized invasive orchid (Table 1).

There is considerable overlap between the two islands in terms of the most prevalent species. Five species are shared between islands within the six most prevailing species of each island. These are in declining order *Epidendrum ciliare, Brassavola cucullata, E. anceps,* the endemic *Psychilis correlli* and the naturalized and invasive *Oecoclades maculata*. On St. Eustatius the endemic *E. difforme* also is among the six most prevailing species. The great majority of the orchid species represented on both islands are either rare (R) or very rare to (already) possibly locally extinct (V) (Table 1, Debrot et al. in prep.).

Habitat:
Aside from many different vegetation types, four main "climatic" habitats can be distinguished on these islands which are not equal to but closely linked to the various vegetation types that have been described before (de Freitas et al., 2014, 2016). These can be labelled: <u>a) Elfin forest</u>; the summits of Mt. Scenery on Saba and The Quill on St. Eustatius (above appr. 800 m of Saba and at 600 m on St. Eustatius); <u>b) Rain forest</u>: the upper slopes of Mt. Scenery below the elfin forest down to about 500—600 m and on St. Eustatius exclusively inside the Quill crater; <u>c) Evergreen forest</u>: occurring between approximately 250 and 600 m on both islands; <u>d) Lowlands:</u> all areas below 250 m on both islands. There are large differences in orchid species composition and species richness between these overall habitats which differ importantly in such aspects as humidity, rainfall and cloud cover.

Orchid species distribution within and between these (climatic) habitats showed that the habitat with the most species on these islands was the elfin "fog" forest zone (18 species) followed by the evergreen forest zone (14 species), the rain forest zone had 13 species and the lowlands zone had 5 species. In terms of species composition, the elfin forest zone has 12 species limited to that zone but shares 6 species with and that extend down into the rainforest zone. The rainforest had only 5 species limited to that zone but aside from the species shared with the elfin woodlands, shared only two species with the evergreen zone. The evergreen zone in turn has eight species limited to that zone and shares four species with the lowland zone which only had 1 unique species limited to that zone.

This means that the orchid flora principally is composed of two floras. One of the Elfin woodland extending down to the rainforest zone, and one of the evergreen zone extending down into the lowland zone (Debrot et al. in prep.).

Minimum viable population size: a minimum viable population (MVP) means a 5% extinction risk within 100 years. The MVP for plants are unknown. However, three things can be said with a fair degree of certainty:

There should be little local concern at present for the most common, widespread and abundant orchid species. For Saba these would be: *Epidendrum anceps, E. ciliare, Brassavola cucullata*, and the endemic *Psychilis correlli*. For St. Eustatius these would be: *Epidendrum anceps, E. ciliare, E. difforme, Brassavola cucullata*, the endemic *Psychilis correlli* and *Tetramicra elegans*.
Most other orchids for these islands have only been documented a handful of times (or less) and can confidently be deemed rare and very vulnerable at the local level. However, the "global" Conservation State of even the regionally more-widespread species remains unknown due to the lack of quantitative assessments in their range states, such that conservation assessments also remain dearly lacking (Table 1).

Finally, for those locally rare species that are also regional endemics (range restricted) it is highly likely that they are locally well-below MVP population sizes. This category concerns the following species: *Epidendrum difforme, E. patens, Ornithidium inflexum, Pontieva petiolata, Tolumnia prionochila, T. urophylla and Triphora surinamensis*. These species may unknowingly also be rare and vulnerable on the other Caribbean islands from which they are known. Hence, quantitative or semi-quantitative assessments for other range islands of these species are dearly needed in order to obtain a better assessment of their global Conservation State.

Present Distribution and Reference Values

The only significant semi-quantitative population and distribution assessment and mapping currently available is based on a quite limited coverage of the total available habitat (Debrot et al., in prep.). This means that, ultimately, little remains known about the full extent of distribution of the species on the two islands and much further work is needed. What can be stated at present with certainty is that even though most species show a very patchy distribution there are clear and large differences in the altitudinal ("habitat") distribution of individual species and overall orchid diversity (Table 1).

Assessment of National Conservation State

Trends in the Caribbean Netherlands
Nothing is known about trends as no quantitative or semi-quantitative studies have been available prior to this assessment.

Reference values for population size and distribution on St. Eustatius: unknown

Recent developments:
Until the 1950s, small-scale agriculture was widespread on these islands. Therefore, at those times, the problem of roaming feral livestock was highly controlled throughout the Caribbean Netherlands even by law (e.g., Debrot, 2016). However, since the 1950s and the demise of small-scale local agriculture, there has been little perceived need to limit livestock roaming and in recent decades this has become a major problem on these islands (Debrot et al., 2018). Such uncontrolled and high roaming livestock densities has certainly had a major harmful effect on orchid populations, including also the endemic species. Even

though several recent efforts have been undertaken to address the roaming livestock problem, any success has so far only been variable and temporary.

Another fairly recent development pertaining to Saba has been the almost total loss of lowland forests (De Freitas et al., 2016). The loss was due to a mid-1990s outbreak of a plant pest but since then recovery of forestation has been negligible due to roaming grazers (goats) which prefer the lowland areas.

Finally, climate change is unstoppably for these islands (Debrot and Bugter, 2010; IPCC, 2022) and will have major consequences in the coming decades. At this stage, we can only speculate about the consequences due to lack of baseline data and quantitative monitoring. However, in general the trend is a gradual loss of the richest elfin woodland and rainforest orchids and uphill migration of the orchids of the dry evergreen woodlands in the coming decades. An additional effect of climate change will be increased susceptibility to invasion by exotic, non-native orchids, the first of which has already arrived and is now well established and spreading (*Oecoclades maculata*).

Assessment of distribution: Favourable
On neither Saba nor St Eustatius is lack of habitat currently a major limitation to the orchid flora in general. Large parts of both islands are well forested and relatively safe from deforestation associated with urbanisation. The only major exceptions are the upper montane vegetation of the "fog" (elfin forest) and rainforest zones. The fog forest orchid flora is the most endangered, as the habitat on which it depends has almost disappeared from St. Eustatius and is very limited on Saba (de Freitas et al., 2014, 2016).

One current threat to St Eustatius that needs to be kept in check is the recent big push to urbanise the slopes of the Quill volcano whereby much (dry-evergreen) orchid habitat is being destroyed. Fortunately, so far these initiatives have not threatened all lower Quill slope habitat and there still remains sufficient scope for controlling and limiting the loss of orchid habitat through greater attention to land-use planning.

Assessment of population: Unfavourable-bad (for almost all species)
This assessment varies from species to species. While a few species can be classified as either abundant, common or patchy and fairly secure in terms of population size and distribution (Table 1), most of the orchid flora is either rare and/or very rare and therefore in poor condition.

Assessment of habitat: Favourable
In general, habitat quality of those areas with orchids is adequate (de Freitas et al. 2014, 2016). The only major persistent negative pressure is uncontrolled and excessive grazing by roaming livestock (e.g., Debrot et al., 2015, Madden, 2020).

Assessment aspect future prospects: Unfavourable-bad
Given the seemingly intractable nature of the migratory livestock problem, the drive to urbanise much of the critical orchid habitat on the Quill slope, and the inexorable long-term effects of climate change, the future prospects for the native orchid flora and endemic orchid species appear bleak.

Table 2. *Overview of key threats to the orchids of Saba and St. Eustatius and implications for management.*

Aspect (for most species)	**2024**
Distribution	Favourable
Population	Unfavourable-bad
Habitat	Favourable
Future prospects	Unfavourable-bad
Overall Assessment of Conservation State	**Unfavourable-bad**

Comparison to the 2018 State of Nature Report

This is the first CS assessment made for orchids in the Caribbean Netherlands and hence no comparison can be made to any earlier assessments.

Recommendations for National Conservation Objectives

a) Reduce uncontrolled livestock grazing pressure and erosion to improve forestation to benefit all orchids.
b) Locate and safeguard population sizes of the most endangered Lesser Antillean endemic orchids while having local orchid enthusiasts help to artificially propagate and plant out the most endangered endemic orchid species.
c) Develop island land-use plans to safeguard sufficient habitat and vegetation in natural state as habitat for orchids.
d) Conduct island-wide surveys to locate and better protect and study the rare endemic orchid species.

Key Threats and Management Implications

The major threats to orchids are threefold.

a) The first and most immediate is grazing pressure by goats. This factor may be expected to be of greatest impact to the terrestrial orchids. However, in the aftermath of hurricanes and storms which continue to increase in frequency and severity, grazing by goats on orchids broken off to the ground is also a major threat.
b) The second is the pressure of increasing human land-use whereby large swaths of natural habitat are removed. So far, this threat principally plays a role on St. Eustatius in which major building projects are concentrating in the dry-evergreen slopes of the Quill.
c) The third threat that will unfold more gradually in time is climate change whereby the expected warming and drying trend in the Caribbean will reduce and ultimately eliminate the rainforest and remnant elfin woodlands on the highest zones of these islands and endanger the richest montane orchid floras. At the same time, ultimately the dry-evergreen orchids can be expected to expand upward as the climatic conditions they require will migrate uphill.

Data Quality and Completeness

Current data quality and completeness are sufficient to document the likely perilous Conservation State of most Lesser Antillean endemic orchids. However, due to the inherent difficulty of quantitative assessments for orchid populations against the great arrears in vegetation conservation science in the Caribbean in general (Torres-Santana, 2010), data availability and quality remain inherently very poor.

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13 Conservation State of Rare and Protected Trees of Bonaire

Debrot, A. O., de Freitas, J. A. and van der Wal, J. T. 2025. From: State of Nature Report for the Caribbean Netherlands, 2024. WUR report C001/25.

Status

This section describes the Conservation State of 38 rare and/or protected tree species of Bonaire. Saba and St. Eustatius also have protected species, such as the endemic Statia Morning Glory (*Ipomoea sphenophylla*), one of the rarest plant species in the Kingdom of the Netherlands.

Based on the Island Decree Nature Management Bonaire (A.B. 2008, no. 23), various local plant and animal species are designated as protected species (see Appendix II). Regarding plants, this list includes a total of 26 woody tree species, one tree semi-parasite, one halophytic herb, one halophytic shrub, two bromeliad species, all ferns and orchids, and finally, one bulb cactus and the seven ferns mentioned by Freitas et al. (2008), as well as four cacti that (like all cacti and orchids), enjoy protection as CITES Appendix II species (i.e., a total of 39 plant species).

Table 1. *Woody tree species of the Caribbean Netherlands with an internationally recognized IUNC Red List threatened status.*

Name				IUCN categorie	SPAW Anne	CITES Appendix
Latin	English common name	Local	Dutch			
Cedrela odorata	Spanish cedar	Stinky cedar		VU	-	-
Guaiacum officinale	Lignum-vitae	Wayaka	Pokhout	EN	III	II
G. sanctum	Holywood lignum-vitae	Wayaka shimaron	Pokhout	EN	III	II
Zanthoxylum flavum	West-Indian satinwood	Kalabari	Geelhout	VU	-	-

Characteristics

Description:
The list of protected plant species on Bonaire includes tree species, cactus species, two bromeliad species, orchids, and ferns. It primarily comprises (very) rare and endangered hardwood species, as well as ferns and orchids that survive as epiphytes high in trees or on steep rock faces that are difficult for roaming livestock to reach. The list also includes the three mangrove species that play a crucial role locally as nursery areas for many coral reef fish species.

Additionally, there are three columnar cactus species on the list of protected species. These species are listed not because of their rarity on the island but because of their key role in the terrestrial ecosystem. Columnar cacti bloom and bear fruit in the dry season when deciduous trees are mostly bare. Therefore, they form an essential food source for the fauna during the dry season (Petit, 1997). There is a strong interdependence between the columnar cactus species and (locally endangered) bat species. The list excludes several species that are very rare, some of which have only recently been discovered or rediscovered on the island. It also excludes some that are so rare and isolated that they were unlikely to be impacted by direct human interventions and so not really be able to benefit from legal protection. Examples include the rare endemic *Myrcia curassavica*, the very rare tree species *Eugenia procera* (de Freitas, 2008), the newly discovered very rare tree *Cyrtocarpa velutinifolia*, and the epiphytic bromeliad *Tillandsia balbisiana* (Freitas and de Lannoy, 2013). *Ficus brittonii* has been found in only a few specimens, and the presence of *Maclura tinctoria* on Bonaire is still unclear. Only one specimen of *Monilcarpa tenuisiliqua* has yet been found (Freitas 2008).

Relative Importance within the Caribbean: Limited
The list of locally rare protected species mostly includes species with a wider distribution within the region. There are a few exceptions. In addition to the three internationally important species mentioned above and that are found on Bonaire (*Guaiacum officinale*, *G. sanctum* and *Zanthoxylum flavum*), there are four tree species with a very limited worldwide distribution. These are the endemic trees *Myrcia curassavica, Maytenus versluysii*, and the recently described *Sabal lougheediana* (Griffith et al. 2019), restricted to the ABC-islands, as well as *Condalia henriquezii* which has a very limited distribution outside the ABC islands. Aside from trees there are also several other endemic plants, but here we limit our discussion to trees.

Ecological Aspects

Habitat:
The list of protected plants includes species found across the full range of soil types and landscape types of Bonaire (de Freitas et al., 2005). It largely concerns species that are strongly tied to rare moist microhabitats. Many currently rare plants were at some point in the past (prior to deforestation for wood harvest and prior to chronic overgrazing) more abundant and an important component of plant diversity. Plant diversity is key to ecosystem resilience in the light of climate change but also key to faunal diversity. Each plant has their own season for carrying leaves, fruits and flowers (Restrepo et al., 2022) which serve as food to the many native and endemic animal species (Bos et al., 2018). Hence, preserving and restoring plant diversity is essential for maintaining a healthy and resilient island ecology.

Minimum Size of Sustainable Population:
Values for the minimum size of a sustainable population are unknown. The species list mostly includes species that may be represented by only a few or just a handful to a few dozen mature specimens (e.g., the Sabal palm, or the tree *Clusia rosea*). In all cases, as with many trees not on the protected species list, these are species that show little or no regenerative growth (e.g., de Freitas, 2008; de Freitas and de Lannoy, 2013). Without successful reproduction, this means that these species will disappear from the island in the coming decades after the still-living mature specimens die. The cause of the overall alarming state of much of the island's flora is historical logging combined with the still ongoing high grazing pressure and resulting erosion from uncontrolled roaming livestock (Lagerveld et al., 2015).

Present Distribution and Reference Values

Rare and endangered plants are distributed across Bonaire and Klein Bonaire. Preliminary research has, however, been able to define a number of concentration areas (Smith et al., 2012). Important concentration areas for the occurrence of rare and endangered plant species include the hills of and around Mount Brandaris in the Washington-Slagbaai Park (Lo fo Wong and de Jong, 1994; de Freitas, 2008), the limestone area of Lima (de Freitas, 2011a), and the terrace landscape of Central Bonaire (de Freitas and de Lannoy, 2013).

With the data available at the prior assessment, no conclusions could be drawn on the short- or long-term trends for these protected species on Bonaire (Debrot et al., 2018). However, it was known that a) these species predominantly survive in limited numbers and delimited areas b) as a rule there was little to no rejuvenation of the populations, c) the primary threat was overgrazing by roaming livestock. However, at present, with new data collected, the key areas of distribution and concentration can be mapped (Debrot et al, in prep.) and much more can be said about surviving population size, longer-term population trends and minimal interventions needed to safeguard 38 legally protected and unprotected species for future generations. The maps and full report will be published elsewhere.

Assessment of National Conservation State

Trends: continued decline and local extinction impending

Many native plant species on Bonaire are likely already extirpated (e.g., *Abrus precatorius, Bromelia humilis, Clusia rosea* and *Phoradendron trinervium*), and many others will undoubtedly follow in the coming decades if measures are not taken (Lo Fo Wong and de Jongh, 1994; van Proosdij, 2012; de Freitas et al., 2005).

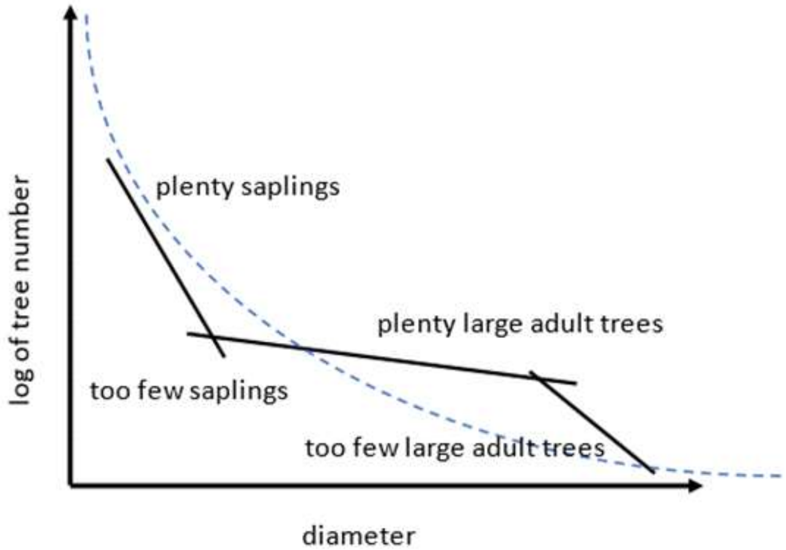

Figure 1. *Hypothetical tree population size structure for a long-lived showing three different growth and mortality phases and four diagnostic deviations from the expected equilibrium population structure (adapted from Westphal et al. 2006).*

Even though longitudinal time trends following specific long-lived tree cohorts is yet totally absent in the Caribbean Netherlands, basically similar insights can be extracted from cross-sectional tree population size structure data, if sufficient trees are present to sample. In this approach tree species population size structures are examined looking deviations from normal exponential decline in tree abundance in different size-classes. The following situations can then be identified:

(H): a healthy (H) population structure with good representation of young recruits and adult trees. No major intervention would seem needed;

(R): a population structure indicative of a recent trend towards recovery (R) with a high preponderance of saplings and young trees, even though older trees may be almost or totally absent (for instance due to overharvest of wood in earlier times). No major field intervention would seem needed assuming all pressures remain the same;

(P): a population structure with low representation of young trees but with a preponderance of large trees. This is a clear indication of potential (P) for recovery due the presence of active seed production notwithstanding the lack of recruits. The minimum intervention needed in this situation would be to reduce grazing pressure. Only in the worst case that grazer reduction is not enough to restart regeneration then additional propagation may be needed.

(D) a population structure with under-representation of both young and old mature growth. In the case of Bonaire such population structures can be interpreted as symptomatic for the combined effects of (past) wood cutting and (lasting) overgrazing and is a sure sign of a long-term declining trend (D). This is a quite worrisome population structure. If the reduction of grazer pressure is not enough to restart regeneration, then additional propagation may (M) quickly be needed.

(U): a population so threatened and small, and from which so few trees can be measured that population structure can give few or no insights into current or past pressures on the population and these remain unknown (U). Unless it is a question of finding more trees somewhere, when so few trees remain it is indicative that artificial propagation may be the last resort to reestablish these species before all individuals are lost. Artificial propagation and out-planting must always be done in grazer-protected areas.

In 2021 and 2022 we visited rare tree hotspot areas for Bonaire (Smith et al., 2012) and mapped and measured the more than 2,095 individual trees of 34 different species. Table 2 shows which species those were, the number of trees assessed, and the diagnosis obtained from the population size-structure data collected. An additional 4 species are included for which no data was collected during our 2021-2022 fieldwork, but which was obtained from other studies (de Freitas and de Lannoy, 2013; de Freitas and Rojer, 2013; de Freitas et al., 2019).

Figure 2 summarizes the results in terms of currently known Conservation State based on documented trends as extracted from population structure assessment. Only two species showed populations structures indicative of long-term health and eight species showed good indications of recovery based on a preponderance of saplings and young trees entering the populations. Twenty-three showed population structures indicative of either long-term decline or were so low in numbers that the only conclusion possible was that the species were in grave danger of disappearing from the flora in the near future even though population structure data too little to reveal anything about temporal trends. There were also four species for which a preponderance of mature trees signified a potential for recovery based on active seed production. These were *Crateva tapia*, the endemic palm *Sabal lougheediana, Sideroxylon obovatum* and *Tabebuia bilbergii.*

Assessment of distribution: Unfavourable-bad
Most species are sparsely distributed or occur in small local clusters of a few individuals in suitable microhabitats. If a species is only found in one or a few small areas, it becomes very vulnerable to chance accidents or may become stranded in suboptimal habitat if local conditions change for the worse. The sparse distribution of many of these species also likely cause their reproductive and genetic viability to be seriously impaired.

Assessment of population: Unfavourable-bad
Some systematic counts have now been conducted for these species allowing more exact insight into the Conservation State of the different species and their prospects for recovery. Based on years of field work, expert knowledge and recent extensive field surveys in hotspot areas, it is clear that many species have no more than a handful of individuals (Table 2) that may be closely related and genetically depauperate.

Assessment of habitat: Unfavourable-bad
The current area and available habitat types for the conservation of the native flora and the protected and endangered species therein are sufficient. However, in many cases the quality of the habitats is severely degraded. The main cause is the heavy overgrazing that generally occurs, leading to severe erosion and reduced water and nutrient retention (Vergeer, 2017). Additionally, a large area in southern Bonaire, which hosts many rare species (de Freitas et al., 2011a; Smith et al., 2012) and important

evergreen vegetation, has been severely damaged by the expanse of salt production areas and groundwater salinization (de Freitas et al., 2005). Furthermore, the openness of the vegetation and the lack of undergrowth result in additional drying during dry periods.

Table 2. *Summary of documented wild population size and population structure diagnoses for 38 rare and/or protected tree species of Bonaire. NNN = no native name. H = healthy, R = recovering, P = potential, D = declining, ? = unknown, tree sample size is too low (<25). Four endangered trees were included from other surveys but not documented by us. These were the extremely rare C. rosea, E. cotinifolia, M. tenuisliqua, and L. lougheediana, some of which (like C. rosea and E. cotinifolia) may no longer be present.*

Nr	Latin name	Local name(s) on Bonaire	Number of trees measured	Diag. Popul.Struc.Type	DBH max (cm) Bonaire #1	DBH max (cm) Curaçao #2	DBH MAX (cm) #3	Legal Protection A.B. 2008 Y/N
1	*Amyris ignea*	NNN	5	U	16.0	13.6	20	Y
2	*Bursera simaruba*	Pal'I sia kòrá	117	R	68.0	19.7	80	N
3	*Celtis iguanaea*	Bèshi di yuana; Pal'I djuku	19	U	19.0	-	25	Y
4	*Clusia rosea*	Kuchiu; Kuchua	0	U	-	32	60	N
5	*Crateva tapia*	Ishiri	54	P	61.3	-	40	Y
6	*Croton niveus*	Bara blanku	91	D	11.5	9.1	15	N
7	*Cynophalla hastata*	Pal'I lora; Pal'I tambú	106	D	53.0	-	60	N
8	*Cynophalla linearis*	NNN	1	U	9.0	13.7	20	N
9	*Cyrtocarpa velutinofolia*	NNN	10	U	28.6	-	40	N
10	*Eugenia procera*	NNN	2	U	17.0	18.5	20	N
11	*Euphorbia cotinifolia*	Manzaliña bobo	0	U	-	-	-	Y
12	*Ficus brittonii*	Palu di mahawa	2	U	30.0	-	20	Y
13	*Geoffroea spinosa*	Pal'I taki	45	D	38.0	-	60	Y
14	*Guaiacum officinale*	Wayaká	112	D	120.0	-	100	Y
15	*Guaiacum sanctum*	Bera; Burobari	188	D	48.0	-	60	Y
16	*Guapira fragrans*	NNN	43	D	68.0	24.5	70	Y
17	*Guapira pacurero*	Mafobari; Mushibari	98	R	40.0	15.8	25	Y
18	*Guettarda roupalifolia*	NNN	10	U	8.3	-	25	N
19	*Jacquinia arborea*	Huku	136	D	29.6	24.9	30	N
20	*Krugioodendron ferreum*	Koubati	50	D	15.0	16	50	Y
21	*Malpighia glabra*	Shimaruku machu	1	?	16.2	-	20	N
22	*Manihot carthagenensis*	Marihuri	69	R	13.0	-	10	Y
23	*Maytenus tetragona*	Dakawa	96	D	46.0	7.1	15	Y
24	*Maytenus versluysii*	NNN	76	R	31.4	12.7	15	Y
25	*Melicoccus bijugatus*	Kenepa	6	U	53.0	-	67	N
26	*Monilcarpa tenuisiliqua*	NNN	0	U	-	10.4	20	Y
27	*Myrcia curassavica*	NNN	47	R	18.9	6.6	15	N
28	*Ouratea guildingi*	NNN	25	R	12.0	-	15	N

Nr	Latin name	Local name(s) on Bonaire	Number of trees measured	Diag. Popul.Struc.Type	DBH max (cm) Bonaire #1	DBH max (cm) Curaçao #2	DBH MAX (cm) #3	Legal Protection A.B. 2008 Y/N
29	*Psidium sartorianum*	Guyaba bè	10	U	26.3	-	20	N
30	*Quadrella indica*	Oliba machu	88	D	35.0	26.5	40	N
31	*Sabal lougheediana*	Cabana	0	P	-	-	37	Y
32	*Schoepfia schreberi*	NNN	105	D	29.4	-	25	Y
33	*Sideroxylon obovatum*	Plaka chikí; Rambèshi	106	P	59.4	26.6	35	N
34	*Spondias mombin*	Oba	4	U	61.2	-	60	Y
35	*Handroanthus billbergii*	Kibrahacha	102	P	54.2	41.8	50	N
36	*Ximenia americana*	Kashu di mondi	50	R	14.4	-	10	Y
37	*Zanthoxylum flavum*	Kalab(a)ri	138	H	56.3	-	40-60	Y
38	*Zanthoxylum monophyllum*	Bosua	83	H	41.7	31	50	Y

#1: DBH max (cm) obs. this study Bonaire
#2: DBH max (cm) Debrot, unpubl. Curaçao
#3: DBH MAX (cm) expert knowledge elsewhere or published

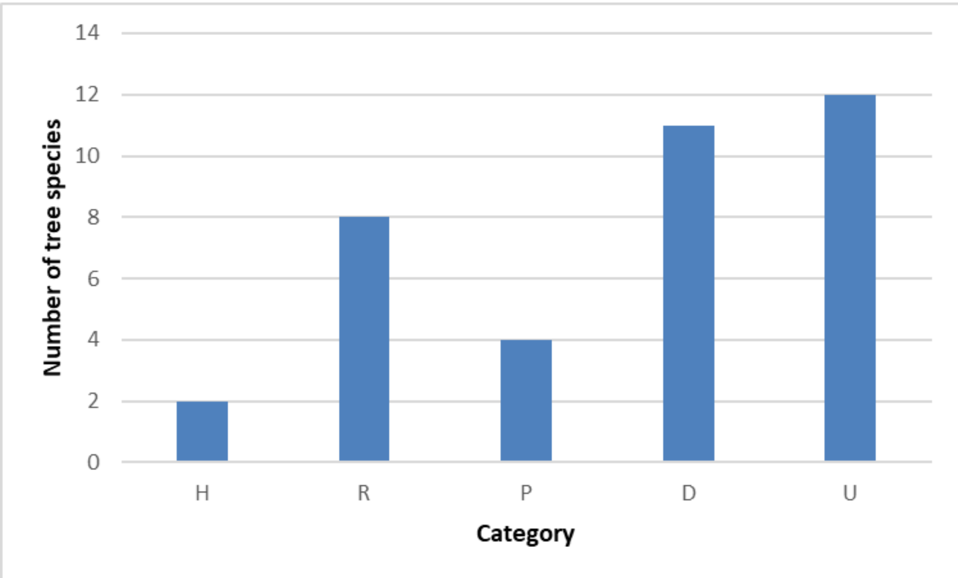

Figure 2. *The number of trees in each of five conservation situations as inferred from the combination of counts and population size-structure data. H = "Healthy": full range of sizes and dominance of young trees; R= "recovering": small trees well represented but large size-classes absent; P = "Potential": mid- to large size-classes present which provide a seed rain for potential recovery; D = "Declining": trees with only largest size-classes present and few if any saplings to mid-sized trees; U = "Unknown": Too low sample size to assess.*

Assessment of future prospects: Unfavourable-bad

On Bonaire, the most arid of the three islands of the Caribbean Netherlands, the issue of roaming livestock is most acute, resulting in many tree species being unable to rejuvenate because seedlings do not survive the grazing pressure and resulting in impoverished (micro)habitats.

While the problem of overgrazing has long been recognized (e.g., Duclos, 1954; Coblentz, 1980), few measures have been taken so far, or initiatives have been launched, but no progress has been made due to limited stakeholder cooperation. Of particular concern is the way in which roaming goats and donkeys strip the bark from columnar cacti, leading to the death of these 'keystone' trees (Anonymous, 2009). There is currently no solution in sight for the problem of roaming livestock. Additionally, climate change and sea-level rise may place a heavy burden on these and other vulnerable species in the future. According to Harter et al. (2015), climate change will take the greatest toll on the flora of small and low islands with homogeneous topography.

Table 3. *Summary overview of the status of rare trees of Bonaire in terms of different conservations aspects.*

Aspect trees Bonaire	**2024**
Distribution	Unfavourable- bad
Population size	Unfavourable - bad
Habitat	Unfavourable - bad
Future prospects	Unfavourable - bad
Overall Assessment of Conservation State	**Unfavourable - bad**

Comparison to the 2018 State of Nature Report

Overall, the CS of the rare and endangered trees of Bonaire has not measurably worsened compared to the 2018 assessment. This second assessment only makes the assessment much more detailed and precise.

Recommendations for National Conservation Objectives

Reducing and controlling roaming livestock (as the primary threat) is the most important goal to improve habitat quality, which will support the expansion and restoration of populations of endangered and protected tree and plant species.

Further identification of the most rare and endangered plant species and the restoration of the recruitment and population sizes of these species (once it is clear which species these are).

Conservation Sub-goals:

Expand the list of protected tree species (Table 5) with the following 17 species that are not yet protected on Bonaire: The very or extremely rare and threatened species: *Clusia rosea, Cynophalla linearis, Cyrtocarpa velutinifolia, Eugenia procera, Euphorbia cotinifolia, Ficus brittonii, Guettarda roupalifolia, Malpighia glabra, M. bijugatus* and *Psidium sartorianum*; the declining species, *Croton niveus, Cynophalla hastata* and *Quadarella indica*; the species with potential to recover: *Sideroxylon obovatum* and *Handroanthus billbergii*; the recovering but still rare species *Bursera simaruba* and *Ouratea guildingi*.

Eradicate and keep important nature areas free of goats. This measure should be sufficient to speed up recovery of the following seven species that are already experiencing influx of young plants (R) under current conditions: *Bursera simaruba, Guapira pacurero, Manihot carthaginensis, Maytenus versluysii, Myrcia curassavica, Ouratea guildingi*, and *Ximenia americana,* as well as the four species that have sufficient adult trees in the reproductive size-class (P): *Crateva tapia, Sabal lougheediana, Sideroxylon obovatum* and *Handroanthus billberghii.*

It will probably also go a long way towards reversing the decline (D) in the following species by allowing regeneration to take place: *Croton niveus, Cynophalla hastata, Geoffroea spinosa, Guiacum officinalis, Guaiacum sanctum, Gapira fragrans, Jaquinea arborea, Krugiodendrum ferreum, Maytenus* *tetragona, Quadarella indica* and *Schoepfia schreberi*. Nevertheless, this is not known with certainty such that in Table 5 it is indicated that propagation might ("M") be necessary in order to reverse the declining trends displayed by these species. For *M. tetragona* a new location appears to have been found where the species is already successfully regenerating (A. v. Proosdij, pers. comm.) which may suggest that simple removal of goats will indeed be sufficient for this species to gradually recover.

Propagate and reintroduce the rarest (category "?") species into (grazer-)protected areas (Table 5). These are the extremely rare: *Amyris ignea, Clusia rosea, Cynophalla linearis, Cyrtocarpa velutinifolia, Eugenia procera, Guettarda roupalifolia, Malpighia glabra, Melicoccus bijugatus, Monilcarpus tenuisliqua* and *Psidium sartorianum*.

Additional surveys are needed to map the further occurrence and distribution of the above and other rare and endangered plant species. Based on those results some species may be able to be removed from the list for protection or intervention in the case that large unknown extensive populations were to be found. Such an outcome is, however, highly unlikely. More likely such surveys will map valuable trees to serve as:

- Seed sources and genetic diversity and
- Target suitable habitat and established individuals around which the same species can be planted in fenced-off plots to create reproductive clusters of these trees in different areas of the island.
- Declare exceptionally large trees as legally protected natural monuments.

Key Threats and Management Implications

Table 5 provides and overview of the key threats and management implications.

Threats:

- Overgrazing
- Erosion
- Climate change (threat: prolonged droughts interspersed with heavy rains that exacerbate erosion)
- Limited knowledge on the ecological needs of many species

Management Implications:

- Reduce livestock densities (see e.g., Debrot, 2016; Debrot et al., 2018).
- Propagate rare and endangered species and plant them in protected areas. In 2006 and 2007, the successful reintroduction of rare native and drought-resistant berry- and fruit-bearing tree species began in the Washington Slagbaai National Park. Goats were removed from Klein Bonaire (687 ha) in the early 1980s (Debrot, 1997), and a reforestation project was carried out from 2006-2009, resulting in the return of fauna (Debrot, 2013). NGO Echo recently reforested four one-hectare fenced areas. During the 2016-2017 rainy season, 3,000 trees were planted in three of these areas. The results of these efforts have never been assessed or made available. Earlier, 3 hectares were restored at Echo's center in Dos Pos, where 500 trees were planted (personal communication, Lauren Schmaltz, NGO Echo). Livestock exclosures by means of fencing around very rare adult tree species can also lead to effective propagation of these species as the seedlings then have better survival prospects.

Table 4. *Recommended management and policy interventions for the 38 rare trees discussed for Bonaire. NNN = no native name. H = healthy, R = recovering, P = potential, D = declining, U = unknown, tree sample size is too low (<25). Y = yes, M = maybe, N = no. "Tabebuia" to read: "Handroanthus"; "pseudoguildingii" to better read: guildingi as it is most likely not pseudoguildingi.*

	Scientific name	Local name(s) on Bonaire	Diagnostic population structure type (H, R, P, D, U)	Current legal protection Y/N	Management Measures Needed		
					add. grazer exclusion needed	artificial propagation needed	add. legal protection needed
1	Amyris ignea	NNN	U	Y	Y	Y	N
2	Bursera simaruba	Pal'I sia kòrá	R	N	N	N	Y
3	Celtis iguanaea	Bèshi di yuana; Pal'I djuku	U	Y	N	N	N
4	Clusia rosea	Kuchiu; Kuchua	U	N	Y	Y	Y
5	Crateva tapia	Ishiri	P	Y	Y	N	N
6	Croton niveus	Bara blanku	D	N	Y	M	Y
7	Cynophalla hastata	Pal'I lora; Pal'I tambú	D	N	Y	M	Y
8	Cynophalla linearis	NNN	U	N	Y	Y	Y
9	Cyrtocarpa velutinofolia	NNN	U	N	Y	Y	Y
10	Eugenia procera	NNN	U	N	Y	Y	Y
11	Euphorbia cotinifolia	Manzalina bobo	U	Y	Y	Y	N
12	Ficus brittonii	Palu di mahawa	U	Y	Y	Y	N
13	Geoffroea spinosa	Pal'I taki	D	Y	Y	M	N
14	Guaiacum officinale	Wayaká	D	Y	Y	M	N
15	Guaiacum sanctum	Bera; Burobari	D	Y	Y	M	N
16	Guapira fragrans	NNN	D	Y	Y	M	N
17	Guapira pacurero	Mafobari; Mushibari	R	Y	N	N	N
18	Guettarda roupalifolia	NNN	U	N	Y	Y	Y
19	Jacquinia arborea	Huku	D	N	Y	M	Y
20	Krugioodendron ferreum	Koubati	D	Y	Y	M	N
21	Malpighia glabra	Shimaruku machu	U	N	Y	Y	Y
22	Manihot carthagenensis	Marihuri	R	Y	N	N	N
23	Maytenus tetragona	Dakawa	D	Y	Y	M	N
24	Maytenus versluysii*	NNN	R	Y	N	N	N
25	Melicoccus bijugatus	Kenepa	U	N	Y	Y	Y
26	Monilcarpa tenuisiliqua	NNN	U	Y	Y	Y	N
27	Myrcia curassavica*	NNN	R	N	N	N	Y
28	Ouratea pseudoguildingii	NNN	R	N	N	N	Y
29	Psidium sartorianum	Guyaba bè	U	N	Y	Y	Y
30	Quadrella indica	Oliba machu	D	N	Y	Y	Y
31	Sabal lougheediana*	Cabana	P	Y	Y	N	N
32	Schoepfia schreberi	NNN	D	Y	Y	M	N
33	Sideroxylon obovatum	Plaka chikí; Rambèshi	P	N	Y	N	Y
34	Spondias mombin	Oba	U	Y	Y	Y	N
35	Tabebuia billbergii	Kibrahacha	P	N	Y	N	Y
36	Ximenia americana	Kashu di mondi	R	Y	N	N	N
37	Zanthoxylum flavum	Kalab(a)ri	H	Y	N	N	N
38	Zanthoxylum monophyllum	Bosua	H	Y	N	N	N

Recent developments:
At present Stinapa Bonaire is reporting success in reducing goat abundance in the Washington-Slagbaai National Park, Bonaire. Stinapa has also bought out the grazing rights for the Washington half of the park such that now the park has full say regarding the goats in the park. On Saba, the longstanding goat culling effort has intensified since 2020 such that, according to authorities, by the end of 2024, only about 500 remain at large. Stenapa of St. Eustatius has recently rekindled efforts to close off the Boven National Park area to goats and cattle after those efforts were prematurely discontinued in the recent past. Also, Stenapa has some efforts in place to reforest native trees at the beach of Zeelandia and former agricultural land on the "Cultuurvlakte" east of the airport.

Table 5. *Overview of key threats to the rare trees of Bonaire and implications for management.*

Key threats		Management implications
Overgrazing	The presence of high densities of free-roaming livestock is a major threat to rare plants, as it severely hinders or nearly entirely prevents the regeneration of many species.	• Remove livestock, especially from protected areas, to densities of 0.1 goat per hectare or less. • Exclusion of livestock from important areas through fencing and control. • Ban livestock from protected nature areas. • Identify and propagate rare and endangered species and plant them in protected areas. • Develop sustainable alternatives to replace traditional extensive livestock farming.
Invasive species	Invasive species naturally possess traits that the island flora is poorly adapted to. This often results in excessive competition, leading to the replacement, eradication, or distortion of the flora.	• Develop and implement an Invasive Alien Species Strategy and Action Plan (see Smith et al. 2014) to prevent the introduction of potentially invasive species.
Urbanisation	Urbanization consumes space and does not allow for natural native plant growth. Often, important nature areas are attractive for development purposes, further pressuring endangered flora because of habitat fragmentation and quality degradation (e.g., terraced landscapes in central Bonaire).	• Implement nature policy and zoning plans to ban construction within key nature areas. • Law enforcement. • Awareness campaigns.
Climate change	This phenomenon will be accompanied by rising sea levels, changes in rainfall, and an increase in average temperature. This will particularly affect coastal vegetation and vegetation types that rely on dew formation.	• Participate in international forums and projects to reduce global greenhouse gas emissions. • Protect large, contiguous nature areas with suitable corridors, allowing flora space to develop. • Combat erosion.

Data Quality and Completeness

There is a significant lack of knowledge about the status, characteristics, and occurrence of very rare tree and (more generally) plant species. This brings the risk that developments may occur in areas where it is not known that a critically endangered species exists. For instance, a rare new bromeliad species (*T. balbisiana*) was recently discovered on the island in an area that had already been designated for residential development. More detailed information, covering larger parts of the island are dearly needed.

There is sufficient knowledge to classify many species as endangered and to make rough management prioritizations as done here. However, the lack of information and knowledge can lead to suboptimal priorities regarding what to protect or propagate and which area is most important to protect. There are not enough data to make precise prioritizations or really to monitor population trends. While there is a foundation of practical experience for carrying out successful replanting (Debrot, 2015), there is too little known about how to propagate many species (e.g., van der Burg et al., 2014). In this regard, the recent cultivation experiences of the local nursery organizations Echo and Terra Barra are considered very valuable. However, nursery propagation and outplanting may be complicated and challenging so the use of fencing around the few remaining adult trees to protect natural seedlings may be more cost-effective.

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14 Conservation State of the Terrestrial Molluscs of the Caribbean Netherlands

Van Leeuwen, S. J. and Neckheim, C. M. 2025. From: State of Nature Report for the Caribbean Netherlands, 2024. WUR report C001/25.

Status

Table 1. *Known status of land mollusc species and subspecies for Bonaire, St. Eustatius and Saba, Netherlands Caribbean as based on listed publications and one publication in prep. Abundance: R = rare, U = uncommon, C = common, A = abundant. Habitat: C = calcareous rocks, L = lowlands, F = forest/ rainforest, H = human environment.*

Family	Name	Abundance Bonaire	Abundance Saba	Abundance St Eustatius	Habi-tat	Status
Achatinidae	*Allopeas gracile* (T. Hutton, 1834)	R	R	U	H, L	Exotic
Achatinidae	*Allopeas micra* (A. d'Orbigny, 1835)	R	U	C	C,L,F	Indigenous
Achatinidae	*Beckianum beckianum* (L. Pfeiffer, 1846)		C	C	L, F	Indigenous
Achatinidae	*Cryptelasmus canteroiana cienfuegosensis* Pilsbry, 1906		R		H	Exotic
Achatinidae	*Lissachatina fulica* (Bowdich, 1822)	R		A	H	Exotic
Achatinidae	*Neosubulina harterti* E.A. Smith, 1898	U			C	Endemic Bonaire
Achatinidae	*Obeliscus plicatellum* (Guppy, 1868)		R		L	Endemic Lesser Antilles
Achatinidae	*Obeliscus swiftianus* (L. Pfeiffer, 1852)			R	L, F	Indigenous
Achatinidae	*Opeas hannense* (Rang, 1831)		U	U	L, F	Exotic
Achatinidae	*Paropeas achatinaceum* (L. Pfeiffer, 1846)	U			H	Exotic
Achatinidae	*Subulina octona* (Bruguière, 1789)	R	A	A	H,L,F	Exotic
Amphibulimidae	*Amphibulima patula* (Bruguière, 1789)		R		F	Endemic Lesser Antilles
Annulariidae	*Bonairea maculata* (Baker, 1924)	U			C	Endemic Bonaire
Annulariidae	*Tudora aurantia* aurantia (Wood, 1828)	C			C, L	Endemic Bonaire
Annulariidae	*Tudora aurantia wassauensis* Baker, 1924	A			C, L	Endemic Bonaire

Bulimulidae	*Bulimulus fraterculus fraterculus* (Potiez & Michaud, 1838)		U	C	L, F	Endemic Lesser Antilles
Bulimulidae	*Bulimulus guadalupensis* (Bruguière, 1789)		C	U	H,L,F	Indigenous
Bulimulidae	*Bulimulus lehmanni* (L. Pfeiffer, 1865)		R		F	Endemic Lesser Antilles
Bulimulidae	*Mesembrinus elongatus* (Röding, 1789)	C		U	L, F	Indigenous
Cerionidae	*Cerion uva bonairensis* Baker, 1924	A			C, L	Endemic Bonaire
Euconulidae	*Guppya gundlachii* (L. Pfeiffer, 1840)		R	R	F	Indigenous
Ferussaciidae	*Karolus consobrinus* (A. d'Orbigny, 1841)	R	U	R	C,L,F	Indigenous
Gastrocoptidae	*Gastrocopta barbadensis* (L. Pfeiffer, 1853)		R	C	C,L,F	Indigenous
Gastrocoptidae	*Gastrocopta curacoana* Pilsbry, 1924	U			C, F	Indigenous
Gastrocoptidae	*Gastrocopta octonaria* Pilsbry, 1924	U			C, F	Indigenous
Gastrocoptidae	*Gastrocopta polyptyx* Pilsbry, 1916		R		L	Indigenous
Gastrocoptidae	*Gastrocopta servilis* riisei (L. Pfeiffer, 1852)	R	R		C, F	Indigenous
Gastrocoptidae	*Gastrocopta servilis* servilis (A. Gould, 1843)			U	L	Indigenous
Gastrodontidae	*Zonitoides arboreus* (Say, 1817)		R	R	F	Indigenous
Helicinidae	*Helicina fasciata* Lamarck, 1822		A	C	L,H,F	Endemic Lesser Antilles
Helicinidae	*Lucidella lirata* (L. Pfeiffer, 1847)	R			F	Indigenous
Helicinidae	*Lucidella striatula* (A. Férussac, 1827)			R	L,F	Indigenous
Helicinidae	*Stoastomops walkeri* Baker, 1924	U			C	Endemic Bonaire
Oleacinidae	*Melaniella gracillima sanctithomensis* (Pilsbry, 1907)		R		L	Endemic Lesser Antilles
Oxychilidae	*Glyphyalus quillensis* de Winter, van Leeuwen & Hovestadt, 2016			U	F	Endemic St Eustatius
Polygyridae	*Polygyra cereolus* (Megerle von Mühlfeld, 1818)	R			H	Exotic
Polygyridae	*Praticolella griseola* (L. Pfeiffer, 1841)	R			H	Exotic

Pristilomatidae	*Hawaiia minuscula* (A. Binney, 1841)	R			C, F	Exotic ?
Pupillidae	*Pupoides nitidulus* (L. Pfeiffer, 1839)	C		R	C,L,F	Indigenous
Sagdidae	*Hojeda* spec.			R	L, F	Indigenous?
Sagdidae	*Hyalosagda subaquila* (Shuttleworth, 1854)			U	L	Endemic Lesser Antilles
Sagdidae	*Lacteoluna selenina* (A. Gould, 1848)		R		F	Indigenous
Sagdidae	*Setidiscus crinitus* (Fulton, 1917)	R			C, F	Indigenous
Scolodontidae	*Happia* spec.		R		F	Indigenous?
Streptaxidae	*Gulella bicolor* (T. Hutton, 1834)	R			C, H, F	Exotic
Streptaxidae	*Streptartemon glaber* (L. Pfeiffer, 1850)	R	C		H, F	Indigenous, exotic on Bonaire
Streptaxidae	*Tomostele musaecola* (Morelet, 1860)		U		H, L	Exotic
Succineidae	*Succinea concordialis* A. Gould, 1848	R			H	Exotic
Succineidae	*Succinea gyrata* Gibbons, 1879	A			C, L	Endemic Bonaire + Curaçao
Succineidae	*Succinea riisei* L. Pfeiffer, 1853		R		L	Indigenous
Urocoptidae	*Brachypodella gibbonsi* Baker, 1924	U			C	Endemic Bonaire
Urocoptidae	*Microceramus bonairensis bonairensis* (E.A. Smith, 1898)	C			C	Endemic Bonaire
Urocoptidae	*Pseudopineria viequensis* (L. Pfeiffer, 1856)		R		H	Indigenous
Valloniidae	*Pupisoma dioscoricola* (C. B. Adams, 1845)	R		R	L, F	Indigenous
Valloniidae	*Pupisoma macneilli* (G. H. Clapp, 1918)		R	R	L, F	Indigenous
Veronicellidae	Veronicellidae	R	R		H, F	Exotic
Zachrysiidae	*Zachrysia provisoria* (L. Pfeiffer, 1858)	R	C		H, L	Exotic
	Total number of taxa	31	28	23	57	

Sources: Haas, 1960; Haas 1962; Clench 1970; Breure, 1974; Hovestadt, 1980; Van der Valk, 1987; Van Leeuwen et al., 2015; Van Leeuwen and Hewitt, 2016; De Winter et al., 2016; Hovestadt & van Leeuwen, 2017; Neckheim, 2021, Van Leeuwen et al., 2023; Van Leeuwen et al., 2025.

Terrestrial molluscs are a very characteristic element of the biodiversity of the Caribbean islands, because of the large number of endemic and range restricted species. This is related to the geographical isolation of the populations and the low mobility of terrestrial molluscs. In total, 57 species of terrestrial molluscs are known from the BES islands at the moment (table 1): 31 from Bonaire, 28 from Saba and 23 from St. Eustatius.

In this table and in the rest of this chapter, we use Bonaire to refer to Bonaire and Klein Bonaire together. The mollusc fauna of Kleine Bonaire consists of a subset of species that occur on Bonaire, including several island endemics (Hovestadt and Van Leeuwen, 2017; Van Leeuwen et al., 2023; Van Leeuwen et al., 2025). The knowledge of the terrestrial molluscs of Bonaire is fairly up to date now, thanks to the study of this group during the relay expedition of Stinapa and Naturalis Biodiversity Center to the island in 2023 (Van Leeuwen et al., 2023; Van Leeuwen et al., 2025). However, the taxonomic status of several endemic taxa is unclear yet. Of most endemic taxa it is not clear whether they are species or subspecies (and endemic taxa to Bonaire only), or if they should be lumped with similar endemic taxa from Aruba and/or Curaçao (and become range restricted taxa for the ABC-islands). For conservation reasons this is relevant to know. Hopefully DNA analysis can help to solve this problem in the future.

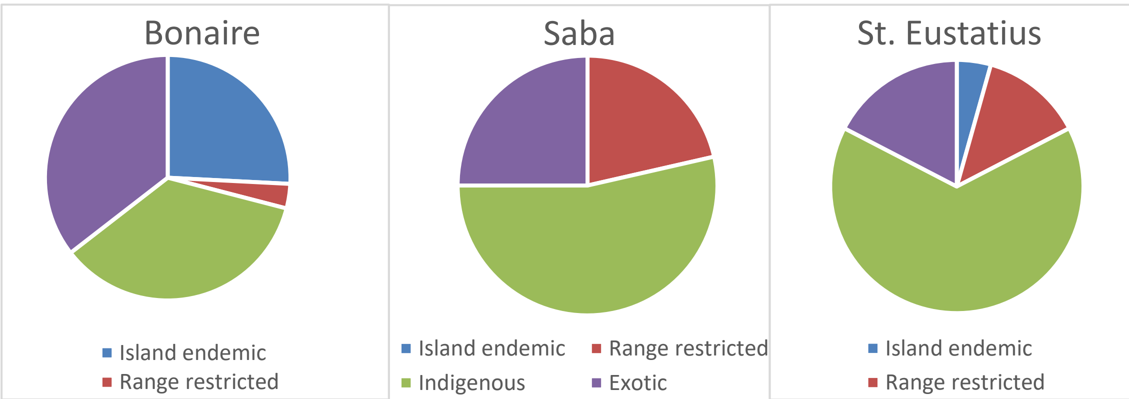
The knowledge published so far about the terrestrial mollusc fauna of St. Eustatius and Saba is shown in table 1. These lists are based on older publications and publications based on a limited amount of fieldwork. For Saba the list is based on Haas, 1960; Haas, 1962; Clench, 1970; Van Leeuwen et al., 2015 and Neckheim, 2021. For St. Eustatius table 1 is based on Haas 1960 and 1962; Hovestadt, 1980; Van der Valk, 1987; Van Leeuwen & Hewitt, 2016; and for *Lissachatina fulica* personal observations of the first author in February, 2025.

The lists seem to be incomplete yet, and more species might be expected for both islands when an actual overview will be made, based on more and actual fieldwork. Typical of this is that Neckheim (2021) found 2 new species for Saba during 1 day of field work. On the other hand, some of the species from Saba have not been reported from the island for over half of a century, but due to the limited amount of fieldwork done on this island, we are not able to conclude if these species disappeared or still live there. In 2025 terrestrial (and fresh water) molluscs will be studied more extensive during an expedition to Saba, with the aim to get a completer and more actual view of the species composition, distribution and habitat preferences.

Characteristics

Description:
The taxa known from the BES islands can be grouped into 4 categories: island endemics; range restricted endemic taxa that occur on a small number of islands, situated close to each other's; indigenous species that occur in a wider area in the Caribbean and/or American region; and exotic species from outside the Caribbean region which are likely to be introduced by humans. Figure 1 shows the species composition for each island according to these categories.

Figure 1. *Species composition of the BES islands, grouped by the distribution range of the species.*



As far as Caribbean island faunas have been studied, every island seems to host its own characteristics mollusc fauna. The distribution of endemic taxa can be limited to one island or to a small group of islands (especially some Lesser Antillean islands) but it can also be limited to a small part of one island. This makes endemic terrestrial molluscs in the Caribbean region very vulnerable.

All six Dutch Caribbean islands have a very special terrestrial mollusc fauna with a high rate of island endemics and range restricted endemic taxa. This is also the case for the BES islands (Figure 1). The fauna of Bonaire has 31 taxa, of which 8 island endemics (only occurring on Bonaire and some on Klein Bonaire) and one range restricted endemic taxa, only occurring on Bonaire, Klein Bonaire and Curaçao. One genus (*Bonairea*) is even known from Bonaire only. The genus *Tudora* is only known from the ABC islands and a nearby peninsula of Venezuela (Paraguaná).

Saba has 28 taxa, of which six are range restricted taxa (Saba has no island endemics). From St. Eustatius 23 taxa are known, of which one island endemic and three range restricted taxa. The range restricted taxa of Saba and St. Eustatius are known from a limited number of Lesser Antillean islands only. This makes it clear that terrestrial molluscs are a very characteristic and unique component of the biodiversity of the BES islands. The list of endemic taxa differs from the list made by Bos et al. (2018), due to recent advances in knowledge.

Endemic taxa deserve special attention in strategies to protect the biodiversity of the islands. An analysis of the IUCN Red List worldwide has shown that land molluscs are the species group with the highest number of documented extinctions (Lydeard et al., 2004). And within this group, island endemics are the most vulnerable (Global Invasive Species Database, 2010). This makes that land molluscs of the Dutch Antillean islands are a very important species group to consider when developing policy and management plans to protect the biodiversity of the islands.

Terrestrial molluscs of the BES islands are not protected by the CITES Convention, and they are not included in any other list of protected species, for example from the national or island government. None of the terrestrial mollusc species of these islands have been assessed for the IUCN Global Red List (IUCN, 2024). So practically nothing has been known about the status of this important and vulnerable component of the native biodiversity of the Dutch Caribbean.

Relative Importance within the Caribbean:
Monographs on the terrestrial mollusc fauna of other Caribbean islands are scarce. Examples of the last century are Puerto Rico (Van der Schalie, 1948), Guadeloupe (Bouchet & Pointier, 1998), Dominica (Robinson et al., 2009), Martinique (Delanoye et al., 2015), St. Kitts and Nevis (Breure et al., 2016), the ABC islands (Hovestadt & Van Leeuwen, 2017), and St. Martin (Hovestadt & Neckheim, 2020, with additions in Neckheim, 2021 and in Neckheim & Hovestadt, 2021). These studies show that the occurrence of island endemics and taxa with a very limited distribution is characteristic for many Caribbean islands. Distribution data in GBIF (2024) show the same.

Several taxa of terrestrial molluscs of the BES islands are limited to one island or only some Caribbean islands. The category of indigenous species refers to species that are spread over a larger group of Caribbean islands, and some of them also occur on the mainland of Venezuela, Central America or Florida (GBIF, 2024). This underpins the big importance of the terrestrial mollusc fauna of the BES islands within the Caribbean.

Ecological Aspects

Habitat:
Molluscs are relatively immobile species that have limited possibilities to react to sudden changes in their habitat or to move to another more suitable habitat. For this reason, the presence of terrestrial molluscs is a good indicator for the quality of nature in a certain area. Due to a lack of studies, very little is known about the species-specific ecology of most Antillean species. This is not only the case in the Caribbean Netherlands but in the Caribbean as a whole.

In table 1, four main mollusc habitats are distinguished: calcareous rocks, lowlands, forest/ rainforest and the human environment, based on the localities where the molluscs were observed.

On Bonaire the calcareous rocks are by far the most important habitat. All island endemic taxa and also most of the other indigenous species occur there. Molluscs need calcium carbonate to build and maintain their shells, so the limestone areas on Bonaire are much more important for land snails than volcanic areas on the island. Nearly half of the endemics of Bonaire are limited to these limestone areas only. Molluscs are vulnerable to dehydration, so most of them live hidden under stones, in holes and cracks and on places shadowed by trees and shrubs. For the same reason also, north-faced calcareous cliffs can be relatively rich in molluscs. The most important area for endemic molluscs on Bonaire is the Karst and Cave Reserve. The endemic taxa *Cerion uva bonairensis, Tudora aurantia aurantia* and *Tudora aurantia wassauensis* do also live in limestone areas, but in lower numbers they also occur in volcanic areas. These habitat preferences can not only be seen at the level of individual taxa, sister taxa of the same genus on Curaçao and Aruba have the same habitat preferences (Hovestadt & Van Leeuwen, 2017).

Snails of the family Succineidae are extremely vulnerable for dehydration. They are land molluscs, but they only live in the vicinity of water or moist places.

On Saba and St. Eustatius calcareous soils are absent or very sparse. On these islands the lowlands with natural vegetation and the forest/ rainforest are the most important habitats for endemic and indigenous mollusc species. The only island endemic on St. Eustatius, *Glyphyalus quillensis*, lives in the rainforest on the crater bottom of the Quill and just below the rim of the crater. These locations are relatively moist and densely forested.

Many taxa from lowlands and from forests/rain forests live in the leaf litter layer or under dead wood, where they are well-protected against dehydration. Other species live on the trunks, branches and leaves of trees and shrubs. The indigenous *Mesembrinus elongatus* is arboreal and lives on a limited number of plant species (Van Buurt, 2016).

What the land mollusc taxa of the BES islands exactly eat is poorly studied for most species, and the way they feed is just globally derived from family characteristics for most species. In general, it is known that many of them eat detritus, algae, fungi or dead plant material, and thus contribute to the composting of the soil. Other species live on trees, shrubs and herbal plants where they scrape algae or lichens from the plants without causing serious damage. Some taxa are plant eaters that eat from the plant stems and leaves, while others are predators or scavengers. For example, snails of the family Scolodontidae are grazers and/or predators on mobile prey and also Streptaxidae feed on mobile prey (Molluscabase, 2024). In turn, the land snails provide a valuable food source for insects (especially ground beetles), centipedes, frogs and birds, and maybe also other reptiles may eat snails.

The biology and life cycle of most taxa of the BES islands is not exactly known but only derived from the general family level. Many terrestrial molluscs are hermaphrodites, which means that they possess both male and female reproductive organs and may be capable of self-fertilization. But for example, Tudora's are of separate sex (males being much smaller than females).

More is known about the food preferences and biology of species that are harmful to agriculture or human health, and invasive species. For example, more detailed studies have been done on the Giant African snail *Lissachatina fulica*, several species of the slug family Veronicellidae and the Cuban garden snail *Zachysia provisoria*. The results, including plant species and other things they prefer to eat and their reproduction, are summarized in factsheets in the Cabi Compendium (Cabi, 2018).

Minimum viable population size: a minimum viable population (MVP) means a 95% probability of survival over the next 100 years (Frankham et al., 2018, Traill et al., 2007). The MVPs for molluscs are unknown. However, the following things can be said with a fair degree of certainty:

There should be little local concern at present for the common and abundant mollusc species. This is also the case for some endemic taxa: on Bonaire the endemic *Cerion uva bonairensis, Tudora aurantia aurantia* and *Tudora aurantia wassauensis* have healthy populations. The endemic and colourful *Helicina fasciata* is abundant on Saba. This species is much less abundant on St. Eustatius, but the population seems to be viable there too (however it became highly endangered on St. Martin).

On Bonaire four endemic taxa are uncommon, which means they have a limited distribution, and they occur in low numbers. Uncommon island endemics on Bonaire are *Neosubulina harterti, Bonairea maculata, Stoastomops walkeri* and *Brachypodella gibbonsi*.

On Saba and St. Eustatius, the abundancy in table 1 is a rough estimation only, due to the much scarcer data available about the current distribution. The only island endemic of St. Eustatius, *Glyphyalus quillensis*, is uncommon. It lives in low numbers in a very limited area. On Saba five endemic species are rare or uncommon: *Obeliscus plicatellum, Amphibulima patula, Bulimulus fraterculus fraterculus, Bulimulus lehmanni*, and *Melaniella gracillima sanctithomensis. Amphibulima patula* seems to be restricted to the higher parts of Mount Scenery and the elfin forest. On St. Eustatius *Hyalosagda subaquila* is an uncommon endemic species.

Moreover, 8 indigenous species are rare or uncommon on Bonaire, 11 on Saba and 11 on St. Eustatius. Some of these species occur on two or three of the BES islands. The following indigenous species are uncommon or rare on all BES islands where they occur: *Obeliscus swiftianus, Guppya gundlachii, Karolus consobrinus, Gastrocopta curacoana, Gastrocopta octonaria, Gastrocopta polyptyx, Gastrocopta servilis riisei, Gastrocopta servilis servilis, Zonitoides arboreus, Lucidella lirata, Lucidella striatula, Setidiscus crinitus, Succinea riisei, Pseudopineria viequensis, Pupisoma dioscoricola* and *Pupisoma macneilli*. These taxa might have small and vulnerable populations although soms taxa are very small and might have been overlooked on Saba and/or St. Eustatius.

All species that are listed as 'rare' in table 1 have seldom been observed. Some species from Saba have been found most recently more than 50 years ago. For these species, it is unsure if they still survive on the island or if they have disappeared. This is the case for *Cryptelasmus canteroiana cienfuegosensis, Gastrocopta polyptyx, Gastrocopta servilis riisei, Pseudopineria viequensis*, and *Succinea riisei*. However, the Conservation State of the regionally more-widespread indigenous species remains totally unknown due to the lack of quantitative assessments of terrestrial molluscs in the Caribbean region.

Present Distribution and Reference Values

There is a major difference in the land mollusc fauna composition between Bonaire on the one hand, and Saba and St. Eustatius on the other (Table 1). There is relatively little overlap in species, and the overlapping species are all exotic species or species with a wide distribution in the Caribbean region. This can be attributed to the large distance between Bonaire, situated in the southern part of the Caribbean Sea, and Saba and St. Eustatius situated in the northern part of the Lesser Antilles. Other reasons are the differences in geology and climate. Bonaire has an arid climate, an abundance of limestone habitat next to volcanic rocks and a low maximum altitude. Saba and St. Eustatius are volcanic islands with consequently mainly volcanic soils, much higher maximum altitudes and higher humidity.

The overlap in terrestrial mollusc fauna between Saba and St. Eustatius is large, with about half of the taxa being found on both islands, including two species that are endemic to the Lesser Antilles.

Although Bonaire is much closer to the mainland of South America than Saba and St. Eustatius, Bonaire has by far the highest number and proportion of island endemics and range restricted taxa (Figure 2).

While Bonaire has the largest number of island endemics, Saba and St. Eustatius have the highest species richness per unit of surface area (Table 2). This is remarkable, especially when one considers the likelihood that even more species occur on these islands than are currently known. Bonaire is much larger in surface area, but the island is much flatter, so it has fewer different "climatic" microhabitats than Saba and St. Eustatius. The fact that the Bonaire fauna has many more endemics can be explained by the much higher age of Bonaire compared to Saba and St. Eustatius, which are fairly recent volcanoes from the Tertiary, while Bonaire was formed in the much older Miocene.

Table 2. *Surface area, altitude and number of species of each island.*

	Bonaire	Saba	St. Eustatius
Surface in km2	288	13	21
Highest peak in meters	240	887	601
Number of terrestrial molluscs	31	28	23
Number of mollusc/km2	0.1	2.2	1.1

Source: Wikipedia (2024)

The overlap between the island faunas and the mollusc fauna of Venezuela is limited. Of the 115 taxa of Venezuela (Wikipedia, 2024), only 10 taxa also occur on Bonaire. Half of these are introduced exotic species. One would expect that the overlap of the fauna of Venezuela with Saba and St. Eustatius is smaller than with Bonaire, but that is not the case. Together Saba and St. Eustatius have 13 taxa that also occur in Venezuela, of which 8 indigenous and 5 exotic species.

Assessment of National Conservation State

Trends in the Caribbean Netherlands

Nothing is known about population trends in the Caribbean Netherlands. There is no systematic monitoring system for molluscs of the BES islands and the available data are insufficient to calculate trends. For Bonaire a baseline was created for the first time in 2023. For Saba and St. Eustatius no baseline has yet been made. It is highly advisable to create these in the near future.
Prior to these baselines, no comparable assessments or data are available. Maybe the material collected by Dr. Pieter Wagenaar Hummelinck in the period 1930 - 1973 (Wagenaar Hummelinck, 1981) may serve to create some kind of reference point, but it is unsure if this will work. His material is stored in Naturalis Biodiversity Center, but until now only a small part of his material has been identified, and the data are not digitalized. So, the available data do not allow to calculate any trends.

Quantitative data to calculate trends are also lacking on other Caribbean islands where the taxa with a restricted range occur, but on St. Martin two of the range restricted endemics from the BES islands (*Bulimulus fraterculus* and *Helicina fasciata*) have become endangered (Van Bussel, 2022).

Exotic species

Due to the growing amount of transport of people and goods the number of exotic mollusc species is growing. Exotic species can compete with indigenous species for food and space. Some exotic species will live an unobtrusive life, others can be harmful to indigenous molluscs, or even to agriculture and human health. The growing number of exotic species was clearly visible on Bonaire (Van Leeuwen et al., 2023; Van Leeuwen et al., 2025) and it is likely that this is also happening on Saba and St. Eustatius. Shortly after their introduction, the distribution of exotic species will mainly be limited to the human environment, but this may change over time. Several exotic species became well established. For example, the introduced species *Subulina octona* has become very widely distributed over St. Eustatius and Saba (personal observations of authors), where it was even found in natural areas like the crater bottom of the Quill and the top of Mount Scenery.

An exotic mollusc species that needs special attention is the African Giant snail *Lissachatina fulica* that was introduced unintentionally on St. Eustatius about a decade ago. Recently *Lissachatina fulica* was also unintentionally introduced on Bonaire. This species is a plant eater that can be harmful to agriculture and garden plants. This snail can also be a risk to human health. It is a potential host of the roundworm, *Angiostrongylus cantonensis* (Chen, 1935) or Rat lungworm, which can cause meningoencephalitis and/or eosinophilic meningitis in humans, diseases that can lead to blindness and death (Smith, 2005). Second, the snail can be a potential host of the nematode *Angiostrongylus costaricensis* Morera & Céspedes, 1971 which causes abdominal angiostrongylosis, a zoonotic disease that occurs from the southern United States to northern Argentina (Thiengo et al., 2007; Fontanilla, 2010). And third, the snail can carry the bacterium *Aeromonas hydrophila* (Chester) Stanier, 1943, that can cause a variety of bacterial infections (bacterioses) in humans, including osteomyelitis, septic arthritis, tonsillitis, and meningitis (USDA 1982, cited by Smith 2005). The transmission of nematodes and bacteria from snails to humans does not only occur through eating raw or undercooked snails. Even snail slime (mucus) on unwashed hands, on unwashed lettuce and snail-contaminated drinking water can be sources of the bacteria and nematodes. Factsheets are available with extended background information about the species, the risks, policy guidelines and literature references (Cabi, 2018; Van Leeuwen, 2023).

Trials on St. Eustatius showed that it seems feasible to control or even eradicate the African Giant snail using a combination of snail baits and handpicking, combined with dedicated monitoring (Debrot et al., 2016). However, these methods need to be applied consistently and during a very long period to eradicate this species completely. So, notwithstanding these efforts the species is still present on St. Eustatius, and it recently entered Bonaire. Notwithstanding the excellent prospects for eradication that existed on St Eustatius early in the invasion process, the measures taken on St. Eustatius were not strong enough and depended too much on the voluntary co-operation of inhabitants. Due to this, the situation has since grown out of hand. By early 2025, the species was abundant on Sint Eustatius and had spread across almost the entire built-up area. The species even approached the edge of the Quill National Park. Because the Giant African snail eats a lot of plants, this could have a negative effect on the shrub and herb layer in the park. It also appears that the residents of Sint Eustatius were not well informed about the health risks (personal observations S. J. van Leeuwen, February 2025).

Reference values for population size and distribution on Bonaire: During the relay expedition to Bonaire in 2023, over 300 snail localities were surveyed. For each locality was noted which species were present or absent. The number of localities where each species was present is used to estimate the abundancy of species on the island (table 1). The following categories are used: 1-10 localities: rare; 11-40 localities: uncommon; 41-100 localities: common; 101-200 localities: abundant. Distribution maps based on this survey are published in Van Leeuwen et al. (2025).

Reference values for population size and distribution on Saba and St. Eustatius: unknown. The abundances in table 1 are nothing more than very rough estimates based on first impressions by the authors. Additional fieldwork might show that some species are much more abundant or rare than expected at the moment.

Recent developments: The New Guinea Flatworm *Platydemus manokwari*
A serious threat to land snails on Bonaire is the discovery of the New Guinea Flatworm *Platydemus manokwari* on Bonaire during the relay expedition in 2023 (De Waart and Van Leeuwen, 2024a, 2024b; De Waart et al., 2025). On INaturalist.org, this flatworm was also reported from Saba and the French part of St. Martin. Recently the species was also observed at the Dutch part of St. Martin and on Curaçao (De Waart et al., 2025). This indicates that the species is spreading further in the Caribbean. The New Guinea Flatworm is ranked in the IUCN top 100 most invasive alien species in the world because it is an effective predator that poses a serious threat to native snails (Global Invasive Species Database 2010). On some Pacific islands this flatworm was an important factor in the extinction of several indigenous mollusc species (Sugiura & Yamaura, 2009). Some years ago, this species also appeared in Florida, and now the populations of endemic tree snails are declining rapidly (Lopez et al., 2022; personal observations by Johan van Blerk and Steve Rosenthal). These tree snails are most similar in size and behaviour to the indigenous *Mesembrinus elongatus* and also the island endemic *Cerion uva bonairensis*, and have approximately the same size. However, the flatworm also preys on much smaller species like *Zonitoides arboreus* and Clausiliidae (Kaneda et al., 1990).

On Bonaire and Curaçao the flatworms were found in garden centres, and that makes it plausible that the worms came with imported potted plants. Many imported plants on the Dutch Caribbean islands come from Florida, where the growers are in the area where Florida's tree snails are disappearing. Wageningen University understands the urgency and developed an action plan to generate more knowledge about the current distribution of the flatworm on Bonaire and to find out what measures can be effective to eradicate or control the population of these flatworms.

The New Guinea Flatworm is not only a risk for molluscs, but also for human health. The reason is that it can be the host of the Rat lungworm, a nematode that can cause meningitis in humans (Thunnissen et al., 2020). What the effect of the flatworm will be on the mollusc populations on the BES islands, is yet unknown. However, given from what can be learned from its effects elsewhere, a "wait and see" approach is not advisable. If still feasible, full eradication should be strived for and tools need to be developed with which to combat and control this new threat.

Assessment of distribution: Unfavourable-inadequate
An analysis of the IUCN Red List worldwide has shown that land molluscs are the species group with the highest number of documented extinctions (Lydeard et al., 2004). And within this group, island endemics are the most vulnerable (Global Invasive Species Database, 2010). As shown in table 1, the BES islands host many mollusc taxa that are island endemics or have a very limited distribution range. And within this small geographical range, some taxa only occur in a very specific habitat.

Assessment of population: Unfavourable-inadequate (differs per taxa)
This assessment differs per taxa. Of the taxa listed as abundant or common it may be expected that they are fairly safe in terms of population size and distribution (Table 1). As shown in table 1, each island also has a number of rare indigenous taxa: 7 on Bonaire, 14 on Saba and 9 on St. Eustatius. The population of these taxa is likely to be small and for this reason they are vulnerable. The caveat here is that on Saba and St. Eustatius these estimations about the abundancy are based on relatively poor and partly older data. More field work may reveal that the abundance is different than currently thought.

Assessment of habitat: Favourable
Presently, sufficient habitat is available for the terrestrial molluscs on the BES islands. However, the loss of suitable habitat is an important risk for the future, due to the growing population and urbanization, overgrazing and the effects of hurricanes (that might occur more frequently due to climate change).

In general, habitat quality of mollusc areas seems to be adequate. Reforestation projects and measures to diminish the grazing by cattle may lead to further improvement of the habitat quality. The caveat here is that on Saba and St. Eustatius this assessment is only tentative and based on relatively poor and partly older data. For example, there no research is done on the impact of the hurricanes on molluscs that live in the elfin forest. More field work may reveal that the mollusc habitat was more affected than currently thought.

Assessment of future prospects: Uncertain, potentially unfavourable-bad
The introduction of the New Guinea Flatworm on Bonaire and Saba is a big potential risk for the molluscs on the islands. If infected potted plants are also imported to St. Eustatius without adequate control measures, there is a high risk that the New Guinea Flatworm will also reach this island. Or maybe it has already happened without being noticed. At this moment it is still unknown how this flatworm will become distributed over the islands, if ways can be developed to eradicate of control the species, and if not, on what species it will preferably predate and in which degree mollusc taxa will be affected. Saba and St. Eustatius have a more humid climate then Bonaire, so the potential impact on these islands might be stronger than on Bonaire, but field data are not available yet.
Another future risk is the deterioration or loss of habitat due growing urbanization and changing vegetation in natural areas due to overgrazing and hurricanes. A summary of the assessment of national Conservation State is given in table 3.

Table 3. *Diagnostic scores for the four different State of Nature criteria for the Terrestrial molluscs of the Caribbean Netherlands as well as well as an overall Caribbean Netherlands conservation assessment for the year 2024.*

Aspect terrestrial molluscs	2024
Distribution	Unfavourable-bad
Population	Unfavourable-bad
Habitat	Favourable
Future prospects	Unfavourable-bad
Overall Assessment of Conservation State	Unfavourable-bad

Comparison to the 2018 State of Nature Report

This is the first CS assessment made for terrestrial molluscs in the Caribbean Netherlands and hence no comparison can be made to any earlier assessments.

Recommendation for National Conservation Objectives

National conservation objectives:

a) Increase local awareness about the uniqueness of the local mollusc fauna that includes many unique endemic taxa. Be proud of them.
b) Protect the endemic mollusc species and their habitats to safeguard this unique and characteristic part of the biodiversity of the BES islands.
c) Locate the most important areas for endemic and indigenous molluscs on each island and prevent the habitats from becoming lost or deteriorated.

Subgoals:

e) Try to develop methods to eradicate or control the New Guinean flatworm *Platydemus manokwari*. Monitor the distribution and effects of the land flatworm on Saba and Bonaire and investigate whether the species also occurs on St. Eustatius. Also consider measures to prevent the import of more exotic species, especially with the import of potted plants.
f) Identify for each island the most important areas for endemic mollusc taxa. When these are situated outside the protected national parks, develop island land-use plans to safeguard sufficient mollusc habitat and vegetation in a natural state. Give priority to protect mollusc-rich limestone areas on Bonaire where the island endemics live, and the type localities of endemic taxa, because these are outside the national parks. For this reason, consider extending the protected area of the Bonaire Karst and Cave Reserve by including the upper part of the limestone terrace and the areas in between the caves in the protected zone.
g) Conduct a plan to fill the most important knowledge gaps for the terrestrial mollusc faunas. Give priority to a complete and actual inventory of the terrestrial mollusc faunas of Saba and St. Eustatius, and the clarification of taxonomic questions by DNA research.
h) Incorporate attention to endemic molluscs in nature education programs.
i) Strengthen and renew measures to control the Giant African snail *Lissachatina fulica* on St. Eustatius, with special attention to the border zones of the Quill National Park, and develop a strict eradication plan for Bonaire, where the distribution of this snail is still limited. Monitor the effect of the measures.
j) Take measures to inform the inhabitants of St. Eustatius, Saba and Bonaire about the health risks of the Giant African snail *Lissachatina fulica* and/or the New Guinea Flatworm *Platydemus manokwari* and inform them how to act when they observe these species.
k) Develop a quantitative monitoring system for the mollusc faunas of the BES islands.

Key Threats and Management Implications

On Bonaire, a big risk is that important areas for endemic molluscs are outside protected nature areas. The important habitat of calcareous rocks, forested limestone platforms and limestone cliffs are mainly situated outside the Washington Slagbaai National Park. The Bonaire Cave and Karst Reserve is among the richest mollusc areas, both in number of species and in population size (number of individuals). The name suggests an area that is protected, but in fact only the interior caves are protected, not the coral platform with indigenous trees above and surrounding the caves that is so important for the molluscs. The risk of this poor protection is that there is no legal basis to act against plans that may contribute to the decline or even extinction of endemic species.

On Saba and St. Eustatius, the most important mollusc areas are within the Saba National Land Park (which has been part of the Mount Scenery National Park since 2018) and the Quill and Boven National Parks, which have a protected status and are managed by nature management organisations.

Another risk is a lower vegetation cover in forests, due to overgrazing and hurricanes. On Bonaire the vegetation was highly affected by the grazing of goats and donkeys (De Freitas et al., 2005). On Saba and St. Eustatius the vegetation is also affected by grazers and moreover the forests, especially the very humid elfin forests, were negatively affected by the hurricanes in 2017 (De Freitas et al., 2014; De Freitas et al., 2016; Van Andel et al., 2016; Eppinga and Pucko, 2018; Jansen, 2020; Van Proosdij, in prep.). The freely roaming goats mainly eat from the plants and young trees in the undergrowth, and they also eat dead leaves. Consequently, a drier microclimate is created on and in the soil. Most land snails are highly dependent on a moist, undisturbed microclimate and they are very sensitive to dehydration. Overgrazing causes this microhabitat to be disturbed or to disappear, resulting in the decline of terrestrial molluscs and other soil animals. This risk mainly applies to forest / rain forest species. However, no quantitative baseline is available, and no recent field work has been done to study the effects of hurricanes on terrestrial molluscs on the BES islands. However, on St. Martin four species of terrestrial molluscs may already have been lost and at least three others became highly endangered (Van Bussel, 2022). However, also on Sint Martin more research is needed to know exactly how many species are endangered. The main causes of the decline in populations include habitat loss due to hurricanes, pollution and construction.

Data Quality and Completeness

A systematic monitoring program for molluscs on the BES islands is lacking. A basic requirement for the assessment of the mollusc fauna of an island is that it is know which species live on each island, how they need to be named and how they are related to similar taxa from other islands. Secondly: knowledge about their habitat preference and distribution over the islands. Thirdly, a reference point in the past is needed to get a better insight in development of the mollusc faunas: which species disappeared or were newly established on the islands, which species are declining or increasing. Table 4 shows the current situation.

Table 4. *Knowledge about molluscs of the BES islands.*

	Bonaire	Saba	St. Eustatius
Complete overview of species	+	-	-
Clear taxonomy	-	+/-	+/-
Information about distribution over the island	+	-	-
Information about habitat preference	+	-	-
Reference point in the past	-	-	-

The only island for which a complete and actual overview of species is available, including quantitative data for one year (2023) is Bonaire (Van Leeuwen et al., in prep.). However, the taxonomy of the endemic mollusc taxa of this island in relation to similar endemic taxa from Curaçao and Aruba is unclear (see the first paragraph in this chapter, and for more detailed information Hovestadt and Van

Leeuwen, 2017 and Van Leeuwen et al., in prep.). In the future, molecular analysis might help to clear these questions.

For Saba and St. Eustatius, the knowledge gap is much larger. Although there are several incidental field observations and some publications about the molluscs of the islands, the species list in table 1 is likely to be incomplete for these islands and some species have not been reported for over half a century. Also, the habitat preferences and the abundance could only be roughly and tentatively described and is unfortunately based on poor data. No quantitative data are available, and no distribution maps have been published. The taxonomy is clear for most, but not all, species known from these islands yet. On Saba, a thorough inventory is planned for early 2025.

For all three islands a quantitative reference point in the past is lacking. Perhaps (but not for sure) the material that Wagenaar Hummelink collected and that is stored in Naturalis Biodiversity Center, will allow to create a kind of reference point in the past. This requires that the material will be identified, digitalised, analysed and published.

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15 Conservation State of the Butterflies of the Caribbean Netherlands

Debrot, A. O., Madden, H. and Boeken, M. 2025. From: State of Nature Report for the Caribbean Netherlands, 2024. WUR report C001/25.

Status

Butterflies are a colourful component of the biodiversity of the Antilles, and with a total of 12 range-restricted taxa (species or subspecies “endemic” to a small distributional range that includes one or more of the Caribbean Netherlands), contribute significantly to the islands’ status as a planetary hotspot of biodiversity (Myers et al., 2000; Mittermeiers et al., 1999). The biogeography of Antillean butterflies remains an active area of study (e.g., Gemmell et al., 2014, Lalonde et al., 2018), because the largely one-dimensional geographic distribution of the island chain presents a simplified system in which to study processes of dispersal and speciation through time and space (Fontenla, 2003). Butterfly abundance and diversity is highly sensitive to environmental variables such as plant diversity and microhabitats correlated to elevational differences and humidity. Because of this, butterflies may be especially useful as bio-indicator species (Osborne et al., 1999, Miller et al., 2011). In spite (and possibly because) of the fact that many species are endangered and have small population sizes and/or limited distributions, due to a lack of studies, very little is known about the species-specific ecology of most Antillean species. This is a key knowledge gap that may seriously hamper their conservation, not only in the Caribbean Netherlands but in the region as a whole. Table 1 provides a basic list of the species recently documented for the Caribbean Netherlands islands of Bonaire, St. Eustatius and Saba (BES).

Characteristics

Description

Overall, few published studies exist on the butterflies of these islands but several patterns can still be discerned. First is the fact that faunas differ greatly between Bonaire on the one hand and Saba and St. Eustatius jointly on the other hand. For Bonaire in the Leeward Dutch Caribbean and Saba and St. Eustatius in the Windward Dutch Caribbean, it is certain that in pre-colonial times more butterflies of forests and humid habitats (like Heliconiinae, Charaxinae, Papilionidae and Coliadinae) would have been present than are currently documented. The effects of aridification and deforestation has unquestionably taken its toll on all three islands, resulting in a decrease in forest and moist forest habitat and an increase in disturbed and arid habitats. This is certainly reflected in an altered composition of the butterfly faunas on all three islands. For St. Eustatius, the only island for which significant quantitative sampling has been done, Pieridae were the most numerically abundant group of butterflies (48%), followed by Lycaenidae (26%) and Hesperiidae (12%) and smaller numbers of both Heliconiinae (6%) and Charaxinae (5%) (Debrot et al., 2020). This differs significantly from the trademark faunal characteristics of the Antilles which have an overall notably higher contribution of species from the Papilionidae, Coliadinae, and Nymphalinae families than continental South America. However, clearly St. Eustatius (as well as Saba) is largely missing these typical West-Indian butterfly families (Debrot et al., 2020). As for Bonaire, its fauna is a reduced subset of the fauna documented from Curaçao (Debrot et al., 1999, Debrot and Miller, 2004, Miller et al., 2003). This means that if forest restoration occurs on Bonaire, many of the butterflies now only known for Curaçao may be able to re-settle successfully on Bonaire.

Additional expected species for Bonaire (upon further study but especially if vegetation recovery occurs) are any of the following 22 species already known from the most nearby butterfly source-island of Curaçao: *Danaus erisimus, Mechanites polymnia, Historis acheronta, Hamadryas feronia, Dynamine mylitta, Eunica monima, Junonia neildi, Chlosyne saundersii, Vanessa cardui, Reoka marius, Ministrymon azia, Strymon megarus, Electrostrymon nubes, Cyclargus huntingtoni, Zerene cesonia, Anteos maerula, Aphrissa statira, Pyrrhopygopsis socrates, Polythrix octomaculata, Chiomara asychis, Cymaenes tripunctus,* and *Heliopetes domicella*. Of these two would also be (or may be*) restricted-range taxa (N. Venezuela, Venezuelan Isl.): *C. huntingtoni,* and *S. megarus ssp*.* (Debrot et al., 1999).

Similarly for Saba and St. Eustatius, many additional species can be expected to blow over from one island to the other and successfully establish themselves if effective forest recovery and restoration takes place. Additional expected species for Saba or St. Eustatius (upon more field effort but especially if vegetation recovers) are any of the following 15 species that are already known from the most nearby butterfly source-islands of St. Maarten*, St. Kitts and Nevis** or both***: *Anteos maerula**, Antillea pelops**, Battus polydamus***, Chiomara asychis**, Chlorostrymon simaethis*, Choranthus vitellius*, Ephyriades zephodes?**, Junonia neildi***, Phoebis agarithe**, Polites dictynna**, Pyristia daira palmira**, Pyristia leuce*, Panoquina panoquinoides***, Siproeta stelenes***. (Debrot et al., 2020). Of these, the following six; *A. pelops, C. vitellius, E. daira palmira, E. zephodes, P. dictynna*, and *P. leuce*, would all be Antillean restricted-range species (Bos et al., 2018; Smith et al., 1994).

The number of range-restricted taxa is much higher for Saba and St. Eustatius compared to Bonaire (nine range-restricted versus only three). Very large proportions of the butterfly faunas on all three BES islands concern rare and/or uncommon species for which the Conservation State is likely poor. Even several species more widely known as typical of disturbed habitat (which includes the strand, i.e.: "beach" community) are now rare or restricted to selected habitat, notwithstanding extensive disturbed habitat on any of the islands (eg. *Dryas iulia* and *Pyristia elathea*). When even the "weediest" butterflies are rare, this should be taken as a clear indication of the poor state of vegetation and paucity of flora needed to support a rich and stable butterfly fauna. The poor and/or questionable Conservation State also is the case for most of the endemic taxa which should be considered conservation priorities. For instance, of the seven St. Eustatius butterflies that are range-restricted ("endemic"), all but two (i.e., five) are uncommon or rare. Likewise, of the six Saba butterflies that are range-restricted, three are uncommon or rare, and for three data are insufficient to be able to give an informed opinion.

Table 1. *Known status of butterfly species for Bonaire, St. Eustatius and Saba, Netherlands Caribbean. As based on listed publications and as updated based on observations by M. Boeken, and H. Madden). Status: I = invasive, R = range restricted in addition to naturally occurring, N = naturally occurring. Abundance: R = rare, U = uncommon, C = common, P = patchily abundant, A = abundant almost all over. Habitat: S = strand ("beach"), D = disturbed, W = woodland, MW = moist woodland, R = rain/mist forest.*

		Bonaire			Statia			Saba		
Size (km2)		288			21			13		
Max. altitude (m)	status	241	abundance	habitat	602	abundance	habitat	887	abundance	habitat
Danainae										
Danaus plexippus	I	x	C	D	x	P	D	x	U	D
Heliconiinae										
Dryas iulia warneri	R	-			x	R	W	x	R	W
Dryas iulia alicionea	N	x	R	MW						
Agraulis vanillae insularis	N	-			x	C	D	x	C	D
Agraulis v. vanillae	N	x	C	D						
Heliconia c. charitonia	N	-			x	P	R	x	C	MW

		Bonaire			Statia			Saba		
Size (km2)		288			21			13		
Max. altitude (m)	status	241	abundance	habitat	602	abundance	habitat	887	abundance	habitat
Heliconius erato hydara	N	x	R	MW	-			-		
Riodiniidae										
Theope virgilius	N	x	R	W	-			-		
Charaxinae										
Anaea troglodyta minor	R	-			x	C	W	x	R	W
Marpesia petreus	N	-			x	R	W	x	R	W
Hypolimnus misippus	I	x	R	W	x	R	W	-		
Junonia zonalis	N	x	R	W	x	U	W	x	U	D
Junonia neildi	N	-			-			-		
Anartia jatrophae	N	x	R	MW	x	C	D	x	R	D
Biblis hyperia	N	-			x	P	R	x	R	R
Vanessa cardui	I	-			x	R	D	-		
Lycaenidae										
Ministrymon ligia	R	x	U	D	-			-		
Chlorostrymon maesites	N	-			-			x	R	W
C. simaethis	N	x	R	W	-			-		
C. telea	N	x	P	W	-			-		
Strymon a. acis	N	-			x			x	U	D
Strymon columella	N	-			x			-		
Strymon b. bubastes	N	x	A	D	-			-		
Strymon bubastus ponce	N	-			x	C	D	x	U	D
Electrostrymon angerona	R	-			x	R	W	x	R	W
Leptotes c. cassius	R	x	U	S	-			-		
Leptotes cassius catilina	R	-			x	U	W	x	A	D
Brephidium exilis spp.	R?	x	R	S	-			-		
Hemiargus h. hanno	N	x	A	D	-			-		
Hemiargus hanno watsoni	N	-			x	A	D	x	A	D
Cyclargus thomasi woodruffi	N	-			x	R	W	-		
Pieridae										
Glutophyrissa drusilla boydi	R	-			-			x	U	D
Glutophyrissa d. drusilla	N	x	R	W	-			-		
Ascia monuste	N	x	R	W	-			-		
Ascia monuste virginia	N	-			x	C	D	x		
Pyristia elathea	N	x	U	D	x			-		
Pyristia gratiosa	N	x	U	W	-			-		
Pyristia lisa	N	x	C	D	x	C	D	x	C	D
Pyristia proterpia	N	x	U	W	-			-		
Krycogonia lyside	N	x	C	W	-			-		
Phoebis agarithe	N	x	U	W	-			-		
Phoebis argante	N	x	R	W	-			-		
Phoebis sennae	N	x	C	D	x	C	D	x	C	D
Rhabdodyras trite	N	-			-			x	R	W

		Bonaire			Statia			Saba		
Size (km2)		288			21			13		
Max. altitude (m)	status	241	abundance	habitat	602	abundance	habitat	887	abundance	habitat
Papilionidae										
Papilio demoleus	I	-			x	R	D	x	R	D
Hesperiidae										
Chioides catillus	N	x	C	W	-			-		
Epargyreus zestos	N	-			-			x	U	W
Polygonus l. leo	N	-			x	P	R	x	U	W
Polygonus savigny punctus	R	-			x	R	W	-		
Urbanus dorantes	N	x	U	D	-			-		
Urbanus proteus	N	-			x	C	W	x	U	W
Urbanus obscurus	R	-			x	C	W	x	U	W
Gesta gesta	N	x	C	D	-			-		
Zopyrion satyrina	N	x	R	D	-			-		
Ephyriades arcas	N	-			x			x	?	?
Pyrgus adepta	N	x	A	D	-			-		
Pyrgus oileus	N	-			x	C	D	x	C	D
Hylephila phyleus	N	x	U	D	x	C	D	x	U	D
Atalopedes flaveola	R	x	R	S	-			-		
Lerodea eufala	N	x	A	D	-			-		
Wallengrenia ophites	R	-			x			x	C	D?
Calpodes ethlius	I	-			-			x	U	D
Panoquina lucas	N	-			x			x	U	W
Panoquina panoquinoides	N	x	R	S	-			-		
Total		34			32			31		
Uncommon sp.		8			2			12		
Rare sp.		14			8			8		
Endemic		3 +1?			7			6		

In conclusion:

- There are large differences in butterfly faunas between leeward Bonaire and the two windward islands of Saba and St. Eustatius.
- On all three islands current butterfly faunas are impoverished compared to former pristine conditions due to massive deforestation and aridification. Key Antillean butterfly families are almost fully missing from the faunas of St. Eustatius and Saba.
- There are lists of additional species to be expected in case future forestation is achieved.
- Saba and St. Eustatius have much larger numbers of range-restricted (endemic) butterfly taxa than Bonaire.
- On all three islands large portions of the fauna concern rare or very rare species, for which their future status on these islands is highly uncertain and which require further study and, where feasible, directed conservation action.

Relative Importance within the Caribbean

Most of the Antillean endemic butterfly species have their centres of distribution around the larger and higher islands of Cuba, Hispaniola and Jamaica (Scott, 1972). The number of species present is also highly dependent on island size and the range of microhabitats present (Ricklefs and Lovette, 1999). As a consequence, the number of endemic species represented on these island is limited (in comparison to the wider Caribbean) but still respectable compared to the Netherlands, which has a much larger total surface area than the BES islands but far fewer endemic butterflies (only one endemic subspecies, Lycaena dispar batava compared to nine endemic taxa for Saba and St. Eustatius and three additional subspecies for Bonaire).

Ecological Aspects

Habitat: four habitat zones and two almost fully separate faunas

The distribution of butterfly abundance was noticeably linked to the range of different habitats represented; hence it is no accident that butterflies are often considered a good ecological "indicator" group (Table 1). These habitat differences can not only be seen at individual species level, but also at the familial level in accordance with known general familial characteristics (Debrot et al., 2020). For instance, Debrot et al. (2020) found that Pieridae were the most abundant family of butterflies on St. Eustatius, amounting for almost half of all butterflies detected, and were well-represented in all habitats. This family is characterised by many large, strong and fast-flying species, and can often even be encountered far offshore. Lycaenidae are small butterflies, adapted to surviving in dry and resource-limited habitats and were the next most abundant group on St. Eustatius, commonly encountered in all habitats except the crater of the Quill, a dormant volcano. Heliconiinae include many species that are forest dwellers and weak fliers, and on St. Eustatius these were notably more abundant in the sheltered vegetated habitats. Charaxinae, which are typically less powerful fliers than Pieridae but stronger fliers than Heliconiinae, appeared somewhat more common in moist, sheltered vegetated habitats than the drier, more open and windswept habitats. The relative impoverishment of the present butterfly faunas of all three islands is likely due to the great extent and persistence of rural anthropogenic- and livestock-related deforestation. This tends to decrease both the coverage and quality of evergreen and moist forest habitat types (Freitas et al., 2004; 2014; 2016) and increase the coverage of arid and disturbed habitats.

Minimum viable population size: a minimum viable population (MVP) means a 95% probability of survival over the next 100 years (Frankham et al., 2014; Traill et al., 2007). The MVPs for insects are unknown. However, three things can be said with a fair degree of certainty:

1. There should be little local concern at present for the most common, widespread and abundant "weedy" butterfly species that thrive in disturbed, deforested, degraded and desertified lands.
2. The BES islands have suffered and continue to suffer unsustainable land degradation, erosion and aridification due to uncontrolled grazing by introduced livestock (especially goats).
3. Most range restricted and island endemic species appear to be rare and/or have very small and vulnerable populations.

Present Distribution and Reference Values

There is a major difference evident in the butterfly species composition between Bonaire on the one hand, and Saba and St. Eustatius on the other (Table 1). There is actually very little overlap in species. This can be ascribed to the distance between Bonaire, situated in the arid leeward islands, and Saba and St. Eustatius situated in the less-arid northern Lesser Antilles. The overlap in species between Saba and St. Eustatius is large, with most species being found on both islands.

Secondly, even though Bonaire is much larger in surface area, it is a much flatter island with fewer “climatic” microhabitats than Saba and St. Eustatius. Even though all islands have a comparable number of species, Bonaire has a much lower species richness per unit of surface area (Table 1).

The third major point is that Saba and St. Eustatius possess a higher number and proportion of the fauna that is range restricted at the species or subspecies level (Table 1). The greater proportion of unique diversity for Saba and St. Eustatius can be understood in their higher degree of isolation from South America. Bonaire on the other hand is situated relatively close to South America with consequently that more species are more-widely found.

Assessment of National Conservation State

Trends in the Caribbean Netherlands: unknown
Data are deficient for trends as no quantitative or semi-quantitative studies were available prior to this assessment and the status of all range-restricted (“endemic”) species on other range islands remains unknown. It is likely that species in disturbed and arid habitats have increased in abundance and distribution on the islands, while those species requiring moist and forested woodlands have decreased in abundance. This is also the future expected trend. The poor status of many of the species currently known from the Caribbean Netherlands has likely become worse due to loss of larval hostplant species and reduced forestation caused by chronic and uncontrolled grazing by feral livestock (e.g., Freitas et al., 2005; 2014; 2016, Debrot et al., 2018; Lagerveld et al., 2015; Debrot, 2016). Only for Saba have densities of feral livestock never been quantified but these have long been known to be problematic as well.

Reference values for population size and distribution on St. Eustatius: unknown

Recent developments: four key threatening developments

Until the 1950s, small-scale agriculture was widespread on these islands. Consequently, at those times, the problem of roaming feral livestock was highly controlled throughout the Caribbean Netherlands, also by law (e.g., Debrot, 2016) and much less of a problem than today. However, since the 1950s and the demise of small-scale local agriculture, the need to limit roaming livestock has been ignored and in recent decades this has become a major problem on all three islands (Debrot et al., 2018). Such uncontrolled, high densities of roaming livestock have certainly had a major harmful effect on floral diversity and woodlands and forests in general, making larval host plants less abundant, and nectar food sources less abundant and less predictable. Despite several recent (and continued) efforts to address the roaming livestock problem, success has been variable and only temporary. Several, small-scale initiatives for reforestation are underway on the islands and, if scaled up using the right species, this could improve conditions for the butterfly faunas.

Another fairly recent development pertaining to Saba has been the almost total loss of lowland forest (De Freitas et al., 2016). This loss was due to a mid-1990s outbreak of a plant pest but since then recovery of forest has been negligible due to roaming grazers (goats) which prefer the lowland areas. For St. Eustatius the urbanization of limited dry evergreen habitat on the slopes of the Quill is reducing the amount of critical butterfly woodland habitat.

Finally, climate change is unstoppably for these islands (Debrot and Bugter, 2010; IPCC, 2022) and will have major consequences in the coming decades. What this will mean can only be surmised at this point due to a lack of baseline data and quantitative monitoring. However, in general what can be expected in the coming decades will be the gradual loss of the butterflies of moist forests and woodlands, as well as difficult uphill migration of the butterflies of dry and disturbed habitats.

Assessment aspects of natural area of distribution: Unfavourable-bad
On none of the islands is "gross" habitat shortage currently a major limitation to the butterfly fauna in general. Large parts of all islands are still forested and fairly safe from deforestation associated with urbanization. The main problem is habitat quality degradation due to goats preventing forest recovery and endangering the many plants required by butterflies, not only as a food supply for adults but also as host plants for larvae.

One threat at present on St. Eustatius to keep in check is the recent large drive to urbanize the slopes of the Quill volcano whereby much (dry-evergreen) butterfly habitat is being destroyed (e.g., Knippenga Estate). Fortunately, so far these initiatives have not threatened all lower Quill slope habitat and there still remains sufficient scope for controlling and limiting the loss of butterfly habitat with greater attention to land-use planning.

Assessment aspect population: Unfavourable-bad (for almost all woodland species and species of moist habitats)
This assessment differs per species. While a few species can be listed as either abundant, common or patchily distributed and, therefore, fairly safe in terms of population size and distribution (Table 1), a large portion of the butterflies is either rare and/or very rare and therefore also highly vulnerable to local eradication.

Assessment aspect habitat: Favourable
In general, habitat quality of those areas with butterflies (de Freitas et al., 2014; 2016) is clearly degraded compared to early colonial conditions but, on the other hand, probably improved in recent decades due to reduced agricultural activity. The only major persistent negative pressure is uncontrolled and excessive grazing by roaming livestock (e.g., Debrot et al., 2015; Madden, 2020).

Assessment aspect future prospects: Unfavourable-bad in the long-term
Given the apparent intractability of the roaming livestock problem, the drive towards urbanization of much critical Quill-slope forest habitat, and the inexorable long-term climate change impacts, the long-term prospects for woodland and humid-forest butterfly species seems bleak.

Table 2. *Summary overview of the status of the butterflies of the Caribbean Netherlands (Bonaire, Saba, St. Eustatius) in terms of different conservations aspects.*

Aspect (for the many rare and range-restricted species)	2024
Distribution	Unfavourable-bad
Population	Unfavourable-bad
Habitat	Favourable
Future prospects	Unfavourable-bad
Overall Assessment of Conservation State	Unfavourable-bad

Comparison to the 2018 State of Nature Report

This is the first CS assessment made for butterflies in the Caribbean Netherlands and hence no comparison can be made to any earlier assessments.

Recommendations for National Conservation Objectives

a) Improve forestation by reducing uncontrolled livestock grazing through a combination of culling, removal and fencing. This will benefit butterflies (and soil, erosion prevention, vegetable gardening, coral reefs, floral diversity and climate adaptation).
b) Create a stable local resource for butterfly persistence based on the required larval host plants and all-season nectar food sources. Propagate and reforest with key larval host plants

c) Develop island land-use plans to safeguard sufficient habitat and vegetation in natural state.
d) Conduct basic conservation biology studies on the rare and range-restricted butterfly species
e) Quantitative monitoring of butterfly abundance and distributions to monitor progress and effectiveness.

Key Threats and Management Implications

The major threats to butterflies are threefold.

a) The first and most immediate is grazing pressure by goats causing aridification and floral impoverishment which disrupts larval host plant availability and nectar food sources. This exists on all three islands.
b) The second is the pressure of increasing anthropogenic land use, whereby large swaths of natural habitat are being destroyed. So far, this threat principally plays a role on St. Eustatius where major building projects are concentrated in the dry-evergreen slopes of the Quill, which is a limited habitat type on the island.
c) The third threat that will unfold more gradually over time is climate change, whereby the expected warming and drying trend in the Caribbean will reduce and ultimately eliminate the rainforest and remnant elfin woodlands found at the highest altitudes on these islands and endanger the richest montane floras and associated butterfly species. At the same time, drought resistant butterflies of disturbed arid lands will become more common across the islands and (in the case of Saba and St. Eustatius) migrate to higher elevations.

Data Quality and Completeness

Current data quality and completeness are sufficient to document the clearly perilous Conservation State of most range-restricted butterflies and many other rare species. However, due to the lack of time-series of quantitative assessments of butterflies, data availability and quality remain inherently very poor. The least is known about the butterflies of Saba. The only island for which there is some actual quantitative data for more than one year is St. Eustatius (Debrot et al., 2020). Clearly the amount and quality of data available on the distribution and abundance of butterflies is insufficient. Furthermore, very little is known about the availability and abundance of the required larval host plants on all three islands.

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16 Conservation State of the Lesser Antillean Iguana of St. Eustatius

Van den Burg, M. P., Mitchell, A. and Debrot, A. O. 2025. From: State of Nature Report for the Caribbean Netherlands, 2024. WUR report C001/25.

Status

Table 1. *Overview of the international Conservation State of the Lesser Antillean Iguana.*

Name				IUCN category	SPAW Annex	CMS Annex	CITES Appendix
Scientific	Common	Local	Dutch				
Iguana delicatissima	Lesser Antillean Iguana	iguana	Antilliaanse leguaan	CR	2	-	II

Classified as Vulnerable until 2009, the poor Conservation State of wild *Iguana delicatissima* populations caused its downscaling to Endangered in 2010 (Breuil et al., 2010), and more recently to Critically Endangered in 2018 as its regional decline continuous (van den Burg et al., 2018a). This is also reflected by the move from SPAW Annex III to Annex II in late 2023. On St. Eustatius, the native population remains present but is critically endangered by occasionally arriving non-native iguanas (*Iguana iguana* species complex) from St. Maarten, which were able to hybridize in 2016.

Characteristics

Description:
Iguana delicatissima is a large tree-dwelling lizard endemic to the Lesser Antilles. It is in severe decline due to habitat destruction, feral predators, hunting, but above all through competitive hybridization with its sister species, the green iguana, *Iguana iguana*. Data from St. Eustatius indicates animals can reach SVL up to 43 cm, tail lengths over 92 cm, and weigh over 3.4 kg (Reichling, 1999; van den Burg, unpublished data). Pasachnik et al. (2006) and Knapp et al. (2014) provide an extensive overview of literature pertaining to the species.

The Lesser Antillean iguana can still hybridize with its sister species, the green iguana, despite ~8–9 million years of divergence (Malone et al., 2017). This process of hybridization is not random, but directional towards hybrid and non-native iguanas, given native *I. delicatissima* populations disappear if non-native iguanas are not removed (van den Burg et al., 2018a). At least two factors disfavour *I. delicatissima* during this process of competitive hybridization and introgression; 1) hybrid and non-native iguanas attain larger overall body sizes, and 2) have higher fecundity compared to native Lesser Antillean iguanas (Vuillaume et al., 2015; van Wagensveld and van den Burg, 2018). The species is also vulnerable to hurricanes and can experience major mortality (Legouez, 2007) and population bottlenecks during and or soon after major hurricane events (van den Burg et al., 2022).

Relative Importance within Caribbean:
The Lesser Antillean iguana was originally found in the Lesser Antilles from Anguilla to Martinique but is rapidly disappearing from both large and small islands due to a range of factors, which include invasive alien predators; hybridization with I. iguana; and habitat loss. Populations have been extirpated on Antigua, Barbuda, St. Kitts and Nevis, Les Îles des Saintes, Marie Galante (Breuil et al., 2010; van den Burg et al., 2018, 2024a), as well as St.-Martin/St. Maarten as recently as since 1996, when the species was last reliably documented from the Colombier valley area (Breuil, 2002).

Of the remaining populations, islands where non-native iguanas remain absent are all small islets of less than 2 km2: Prickly Pear East, Les Îles de la Petite Terre, Îlet Chancel, Île Fourchue and Îlet Frégate. On all other islands where Lesser Antillean Iguanas remain present, so do non-native iguanas, and hybridization is likely on-going: Anguilla, St. Eustatius, St. Barthélemy, Basse Terre, Grande Terre, La Désirade, Dominica, Martinique, and Îlet Ramiers (Vuillaume et al., 2015; van den Burg et al., 2018a, 2018b; Pounder et al., 2020).

Although non-native *I. iguana* is present on Dominica (van den Burg et al., 2020), the resident native Lesser Antillean population is the only one that exceeds the long-term minimum viable population (MVP) size of 5,000 individuals. In recent years, pure *I. delicatissima* have been translocated from Anguilla to neighbouring Prickly Pear East, a population subsequently supplemented with translocated iguanas from Dominica (Pounder et al., 2020).

Ecological Aspects

While its precarious Conservation State would stress the need for scientific study, life history information remains limited (Pasachnik et al., 2006; Knapp, 2007) but see more recent studies (Knapp and Perez-Heydrich, 2012; Knapp et al., 2016; Warret Rodrigues et al., 2021). Information on the St. Eustatius population is provided by Debrot and Boman (2013, 2014), Debrot et al., (2013; 2014; 2022), and van den Burg et al. (2018b; 2018c; 2022).

Habitat: The Lesser Antillean iguana occupies islands of the northern Lesser Antilles from sea level to approximately 700 m on the larger islands. It can thrive in habitats ranging from mangroves to dry or humid forest, dry rocky shrub lands or manicured gardens (Legouez, 2007). Debrot and Boman (2013) found that iguanas on St. Eustatius favoured habitat at altitudes lower than 300 m, excluding about 4 km2 of habitat surrounding the 600 m high Quill volcano. Debrot and Boman (2014) report the highest iguana densities and sighting rates for St. Eustatius to be in the human-populated estate subdivisions concentrated along the north-western lower flanks of the Quill, and along the escarpment and cliffs between Oranjestad harbour and the town located above the cliffs (2.00 iguana/ha). Subsequent surveys indicated that high densities of especially large adults are also found on the oil terminal (van den Burg et al., 2018b).

Food: This species is fully herbivorous, feeding on the leaves, fruits and flowers of a wide variety of plants, and is versatile in its habitat choice. From observations on St. Eustatius, and elsewhere, as well as on the green iguanas of the Dutch Caribbean, it is evident that these iguanas can survive on very sparse vegetation and in a variety of habitats. Food availability is probably not a limiting factor on St. Eustatius; even in areas that are heavily grazed or overgrown by the invasive Coralita vine (*Antigonon leptopus*), Lesser Antillean iguanas have in fact been observed eating Coralita.

Disturbance/mortality: Two studies report on and discuss cause of death or life-threatening incidents (total of 83 cases; Debrot and Boman, 2014; van den Burg et al., 2018c). These reports indicate traffic and dogs kept in gardens as major threats. Other documented sources of iguana mortalities were starvation (or drowning) in abandoned cisterns, entanglement in fencing, hunting, and predation by domestic cats. A single cat was even observed killing two hatchlings from the same nest. Seventy-five (i.e., 90%) of the 83 endangerment or mortality events were human-related. There were two documented cases in which iguanas were killed for consumption, both incidents involved the same people.

A recent preliminary study of iguana nesting sites on St. Eustatius highlighted how the species is being overlooked as an ecosystem engineer, identifying a previously undescribed keystone species function (Thibaudier et al., 2024). This preliminary effort also highlighted the threat of goats, construction and Coralita to iguana nesting sites. Indeed, Coralita coverage across St. Eustatius is increasing and projected to become 36% (Huisman et al., 2021). This vine is known to overgrow and suffocate native vegetation, thereby further reducing the heavily goat-affected habitat available for iguanas.

Minimum size viable population: a minimum viable population (MVP) means a 5% extinction risk within 100 years. The MVP for *Iguana delicatissima* is not known with certainty but according to Breuil (2002) the long-term MVP for *I. delicatissima* populations is about 5,000 individuals. The St. Eustatius population is at less than 15% of this MPV estimate. The population appears somewhat fragmented, but a genetic assessment of population structure found no gene flow barriers to be present (van den Burg et al., 2018b). Although adult iguanas are generally very static, hatchlings can disperse several hundreds of meters of their nest within their first week(s) (unpublished data STENAPA).

Present Distribution and Reference Values

Historically, iguanas were likely present across the entire island, except for the Quill crater (ridge) and the higher slopes of the Quill where climatological variables (clouds, fog, and more rain) make this area less ideal for iguanas (de Freitas et al., 2014). As pointed out by Debrot and Boman (2013), iguanas often exploit and seek out discontinuities in habitats. The central plains area of the island consists of outstretched areas with low shrubs and grassland and provides the iguana few shelter possibilities either in the form of high vegetation or in the form of boulder fields with crevices. While the habitat would otherwise be suitable, iguanas appear not to choose these areas. Other than the higher parts of the Quill volcano, the whole of the island is essentially suitable as habitat for the iguana if local vegetation is adequate. Although the Boven National Park was reported to have high numbers of iguanas during the 1990’s and early 2000’s (Reichling, 1999; Fogarty et al., 2004), currently almost no iguana appears present inside the Boven NP (unpublished data, STENAPA).

Since 2013, at least nine non-native iguanas have unintentionally arrived on St. Eustatius from St.-Martin/St. Maarten, while one adult female is believed to have been intentionally introduced (Debrot et al., 2022; STENEPA, unpublished data). Prior to its capture, the intentionally introduced individual produced at least one clutch of F1 hybrids of which nine individuals have been captured during rapid response actions performed between 2016-2018 (Debrot et al., 2022; Figure 1). The capture of a relatively small hybrid in 2020 suggests a second hybrid clutch has hatched on St. Eustatius. Currently, only a single non-native/hybrid individual is known to be present; a large female that was sighted one time south of the oil terminal (Figure 1). Hundreds of surveying hours within the surrounding area have so far not resulted in her capture. Genetic analysis of 255 iguanas from St. Eustatius that were sampled before the discovery of the intentionally introduced animal in 2016 show no hybridization was ongoing before 2016/2017 (van den Burg et al., 2018b).

Reference values for population size and distribution on St. Eustatius: 48,000
Under favourable circumstances, iguana populations can attain high densities. Healthy populations of the Lesser Antillean iguana in the French islands have been estimated at some 60 adults/ha (Breuil, 2002). Knapp and Perez-Hydrich (2012) documented densities of 36-43 iguanas/ha for several Lesser Antillean iguana populations in Dominica. On St. Eustatius, the available area that can have optimal iguana habitat is about 16 km^2 (Debrot and Boman, 2013). With an expected virgin carrying capacity of at least 30 animals per hectare, the original St. Eustatius population size prior to human intervention is estimated to have been at least 48,000 animals, well above the 5000 MVP lower limit.

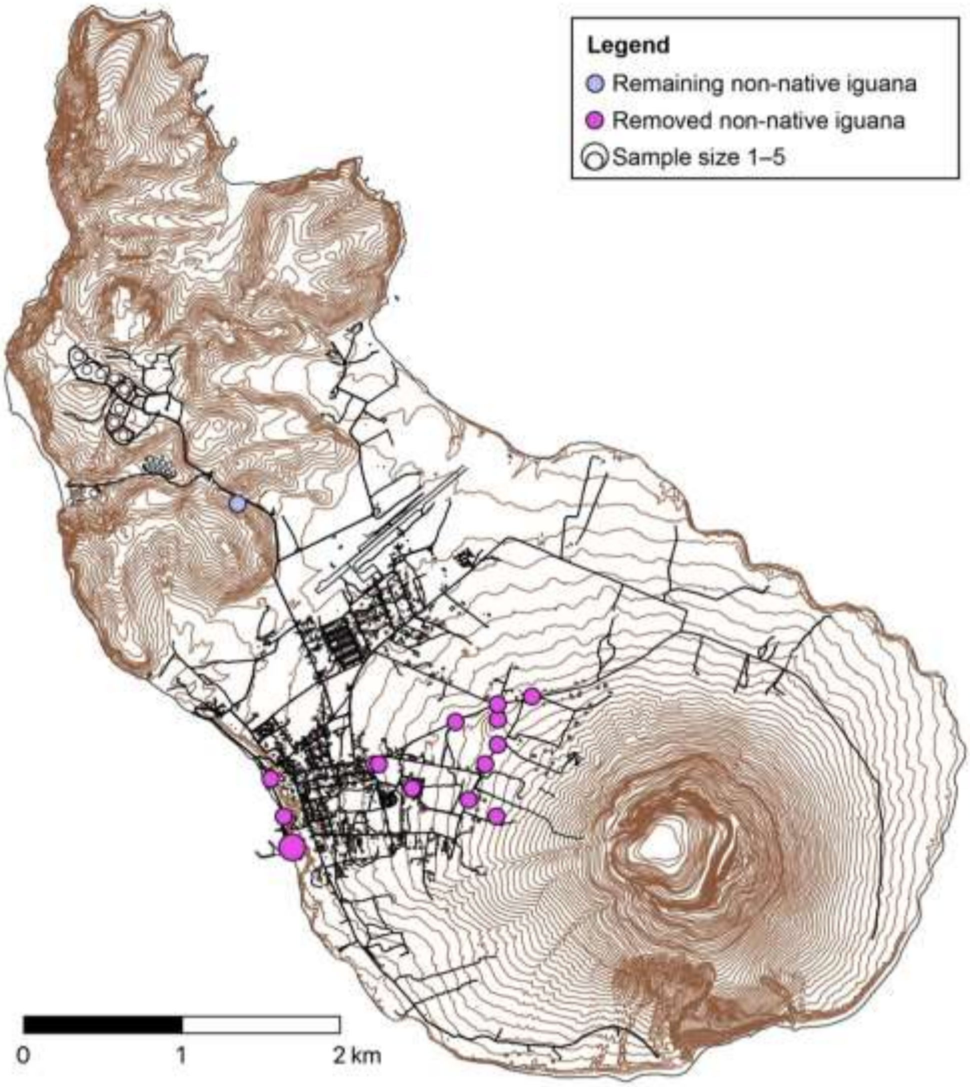

Figure 1. *Map indicating locations of captured and currently present non-native iguanas and hybrids on St. Eustatius.*

Assessment of National Conservation State

Trends in the Caribbean Netherlands
The pre-colonial natural population size for the species is estimated to have been 48,000 or higher (see below). Past population size estimates for St. Eustatius amount to about 300 animals in 1992, less than 300 animals in 2000, and about 425 (275-650) animals in 2004 (Fogarty et al., 2004). Subsequent monitoring work suggested that around 2015-2017, the population size was around or above the higher margin of the 2004 estimate (STENAPA, unpublished data). However, using both survey-transect data as well as opportunistic sightings, van den Burg et al. (2022) found that the Lesser Antillean Iguana population on St. Eustatius declined by ~25% during 2017, which was the most intensive Atlantic Hurricane season on record; both hurricanes Irma and Maria passed the island at close proximity.

Recent developments: The ongoing incursions of green iguanas remain the most severe threat to the *I. delicatissima* population on St. Eustatius. So far, the impact of these incursions has been minimalized through the capture of non-native and hybrid iguanas during a rapid action campaign implemented by a collaboration of multiple stakeholders, funded by the Ministry of LVVN and funding organizations. The NEPP for the Caribbean Netherlands assigns a high priority to invasive species problems like this one (Min. LNV et al., 2020). Currently only a single non-native iguana is known to be present on St. Eustatius. The results from these eradication efforts demonstrate the great difficulty in locating individual non-native iguanas. In the absence of improved biosecurity, both on St. Eustatius and St.-Martin/St. Maarten, a continuous and structurally funded extermination campaign is essential to protect the St. Eustatius *Iguana delicatissima* population. Failure to prevent ongoing non-native incursions and on-island extermination of green/hybrid iguanas will result in the disappearance of *Iguana delicatissima* within the coming decades. A recent morphological assessment between the *I. delicatissima* and *I. iguana* populations of respectively St. Eustatius and St. Maarten, has shown that length-dependent characters can be utilized to separate between native and pure non-native iguanas (van den Burg et al., 2024b). These are in addition to scale and colouration characters identified by Breuil (2013).

The recent inclusion of *I. delicatissima* on the SPAW Annex II increases its legal protective status under the Nature Management Bases Act Protection BES. However, knowledge to implement this protective status and assess impacts of spatial interventions, such as construction projects, are currently inadequate. This prevents impact assessments during environmental impact and ecological permit assessments.

Assessment of distribution Unfavourable-inadequate
The absence of iguanas within Boven National Park and large low-elevation areas is a cause for concern. Equally is the present low density of iguanas on St. Eustatius and the tendency of the species to seek out selected areas, particularly developed anthropogenic areas, places the species at higher risk (Debrot and Boman, 2013). This is mainly due to various sources of man-associated mortality (such as traffic and domestic dogs). An assessment of habitat restoration and potential reintroduction of iguanas into Boven NP is planned to be undertaken by STENAPA and iguana experts.

Assessment of population: Unfavourable-bad
Although a recent population estimate is absent, post-hurricane data from 2018-2019 suggests the population is presumably around or above the 2004 estimate of 425 (275-650) animals (Fogarty et al., 2004). This is far below the required MVP of 5,000 animals and means that the iguana is critically endangered on St. Eustatius. In addition, a recent genetic study identified this population to be genetically depauperate, with extremely low levels of genetic diversity and the presence of possible genetically caused morphological abnormalities (van den Burg et al., 2018b). Irrespective of the threat of hybridization, we currently lack an understanding of why the population is not increasing in size. Whilst captive breeding would increase the native population size (Debrot and Boman, 2013; Debrot et al., 2014), it should be emphasized that this is likely to be symptom treatment and will not provide understanding or a cure for the reason of population growth absence.

Assessment of habitat: Unfavourable-inadequate
There is no doubt that in the virgin state of the island, habitat suitability was much better. The species is flexible in its habitat use and the island provides habitat that can allow for expansion and recovery of the population, with habitat availability likely not limiting population growth (Debrot and Boman, 2013). However, habitat quality is likely to be highly affected by roaming goats and coralita. Additionally, availability and suitability of nesting locations may be limiting population recovery (Debrot et al., 2014; Thibaudier et al., 2024).

Assessment of future prospects: Unfavourable-bad
In the absence of improved biosecurity to halt incursions and a structural financial system for extermination efforts, it is unlikely that further hybridization can be prevented, leading to the disappearance of the population. If non-native incursions can from here on out be prevented, the low population size and seeming absence in population growth remain a major cause for concern. Of immediate concern is the Jan-2023 sighting of a non-native adult female iguana, which might have laid two hybrid clutches since that sighting; nocturnal surveys to assess their absence/presence are urgently needed.

Table 2. *Summary overview of the status of Lesser Antillean Iguana of the Caribbean Netherlands (only St. Eustatius) in terms of different conservations aspects.*

Aspect Lesser Antillean Iguana	2024
Distribution	Unfavourable-inadequate
Population	Unfavourable-bad
Habitat	Unfavourable-inadequate
Future prospects	Unfavourable-bad
Overall Assessment of Conservation State	**Unfavourable-bad**

Comparison to the 2018 State of Nature Report

Overall, the CS of *Iguana delicatissima* on St. Eustatius has become more precarious since the 2018 assessment, especially due to the continuing longterm threat of hybridization with invasive Green iguanas.

Recommendations for National Conservation Objectives

Overall: Safeguard the species from hybridization and increase population size.
Goals:

m) Biosecurity improvement at regional and local harbours to halt green iguana incursions (which will also prevent incursions of other potential threats, such as raccoons) and small Indian mongooses (*Herpestes javanicus*)
n) Urgent assessment of hybrid hatchling and juvenile presence north of Oranjestad
o) Island-wide survey and eradication of all green iguanas and hybrid iguanas
p) Perform study to understand absence of population growth
q) Perform study to provide policymakers with legally necessary knowledge for protection of for example nesting sites
r) Active measures to reduce the currently high rates of (accidental) anthropogenic-induced mortality.

Key Threats and Management Implications

The major threat to this species' survival is hybridization with green iguanas, and their continuous incursions from St.-Martin/St. Maarten. If non-native iguanas are not continuously identified and eradicated, long-term survival of a pure *I. delicatissima* population on St. Eustatius is impossible. If left unchecked, survival of this population is only feasible in captivity or through inter-island translocation; however, also on other islands/islets incursions are possible given region-wide presence of non-native iguanas (Knapp et al., 2020). Currently, immediate monitoring action is required to assess if hybrid hatchlings and juveniles are present north of Oranjestad where an adult female non-native iguana was sighted in January 2023. Management priority should focus on biosecurity improvement and continuous surveys to assess presence of non-native iguanas, resulting in their capture; methods to locate individual iguanas more effectively in heterogenic terrain and habitat should be explored, e.g., using drones.

As mentioned above, the low population size and absence in growth are another major threat. However, a head-starting strategy, through ensuring iguanas will survive until a larger size, would prevent stakeholders from understanding its reason. Hence focus should lay on studying the 1) availability and quality of nest sites, including present threats; 2) how many eggs survive to hatching (nest success); 3) subsequent survival of hatchlings. These insights will aid limiting threats to nest sites as well as hatchling and juvenile iguanas, allowing a sustainable growth in population size.

Data Quality and Completeness

Current data quality and completeness are sufficient to document the species' perilous Conservation State given the threat of green iguana incursions and hybridization. However, these data characteristics are inadequate considering only the native *I. delicatissima* population. Namely, a recent population size estimate is absent, and knowledge on the number, distribution, and threats to legally protected nesting sites is very poor; knowledge that is necessary to follow the new legal status of nesting sites given the species' recent inclusion on SPAW Annex II. Effort is best invested in green/hybrid iguana incursion prevention and understanding the absence in *I. delicatissima* population growth. Further efforts should increase public appreciation of the value of this endangered species to reduce unnecessary anthropogenic-related mortality (Debrot and Boman, 2014).

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17 Conservation State of the Saba Green Iguana in Saba

Van den Burg M. P., Madden, H. and Debrot, A. O. 2025. From: State of Nature Report for the Caribbean Netherlands, 2024. WUR report C001/25.

Status

Table 1. *Overview of the international Conservation State of the Saba Green Iguana.*

Name				IUCN category	SPAW Annex	CMS Annex	CITES Appendix
Scientific	**Common**	**Local**	**Dutch**				
Iguana iguana OR *Iguana melanoderma* (Breuil et al. 2020)	Saba Green Iguana	iguana	Sabaanse groene leguaan	CR	3	-	II

The native *Iguana iguana* population of Saba had not received nearly any scientific and or conservation attention prior to the proposed taxonomic elevation to *Iguana melanoderma* by Breuil et al. (2020). Prior to 2022, it was considered as a species of Least Concern within the IUCN assessment of *Iguana iguana* (Bock et al., 2018) but, based on its unique island genetic status, has since been individually assessed as Critically Endangered (van den Burg and Debrot, 2022). Since the discussion on its taxonomy is still ongoing, it is considered as a native population of green iguana, *Iguana iguana,* on the SPAW Annex 3.

Characteristics

Description:
The native Saban *Iguana iguana* is part of the proposed taxon *Iguana melanoderma* that is also present on Montserrat, St. Croix and St. Thomas (De Jesús Villanueva et al., 2021), and a yet undefined region in northern Venezuela (Breuil et al., 2020). Even so, the Iguana Taxonomy Working Group (2022) still is in discussion about its taxonomic status.

This account is limited to the Saba population, known as the Saba Green Iguana. In contrast to the Lesser Antillean Iguana, *I. delicatissima,* of neighbouring St. Eustatius, this species can often be seen on the ground, in addition to in trees like most populations within the *Iguana iguana* species complex. Saban Green Iguanas also often appear to seek nocturnal refuge in rock crevices and under boulders. Despite little available data, Saban iguanas have been found to reach SVL up to 43.9 cm and tail lengths over 110 cm (van den Burg et al., 2022a), which is smaller compared to max. SVL from mainland *Iguana iguana* locations that can exceed 55 cm (Fitch and Henderson, 1977). Saba Island Green Iguanas are particularly black, more so than the Montserrat population (Breuil et al., 2020). Very little has been published about the Saba population, but see Breuil et al. (2020), and some minor comments by Blankenship (1990) and Lazell (1973).

Hybridization between Saban Green Iguanas and other (invasive) members of the *I. iguana* species complex remains possible. This process is currently ongoing on Saba with multiple non-native iguanas from St.-Martin/St. Maarten having made their way to Saba (van den Burg et al., 2023, unpublished data). Exact details about the outcome of long-term on-island hybridization on Saba remains unknown but it's believed the native Saba Green Iguana population will face a similar declining effect as *I. delicatissima* if hybridization with non-native iguanas should occur more extensively (van den Burg et al., 2018a). While data on clutch size is limited, these suggests Saba Green Iguanas lay smaller clutches compared to invasive *Iguana iguana* from mainland populations (Bock et al., 2018), and more in line with other island iguana populations, like *Iguana delicatissima* (Knapp et al., 2016). Similarly to populations of *I. delicatissima*, the Saba Green Iguana population is likely very vulnerable to hurricanes (van den Burg et al., 2022b).

Relative Importance within Caribbean: Only two native populations of the *Iguana iguana* "species complex" occur in the northern Lesser Antilles which are on Saba and Montserrat. These two populations share a mitochondrial ND4 haplotype and are most closely related to iguanas from the area of Cumana on the northern coast of Venezuela (Stephen et al., 2013), which have together been proposed to be part of *Iguana melanoderma* (Breuil et al., 2020). Other samples from Venezuela have not been analyzed yet so there is currently no understanding of the possible mainland range of this melanistic group within Venezuela. Fortunately, while non-native, invasive iguanas have been identified on Saba, a genetic analysis of over 70 samples did not yet show any sign of non-native iguana presence on Montserrat (van den Burg et al., 2023).

Ecological Aspects

Habitat: Lazell (1973) reported observing iguanas all over Saba, including towards the summit of Mount Scenery (887 m). More recent assessments indicate iguanas likely occur up to a maximum elevation of ~550 m (Gerber, 1999; Breuil et al., 2020; van den Burg et al., 2022a); a discrepancy presumably caused by the former degraded and open state of habitats at higher elevations due to the former presence of plantations (Esperen, 2017). In contrast, data from recent transect surveys and opportunistic sightings in 2021 suggest iguanas occur in all vegetation types (de Freitas et al., 2016), except for the two highest occurring vegetation types at above >550 m (van den Burg et al., 2022a). Although the "*Bothriochloa* mountains vegetation type" on the north side of Saba (de Freitas et al., 2016) was not assessed, its occurrence at elevations below 500 m suggests iguanas are likely present there as well, though presumably at low densities.

Food: Similarly to *Iguana delicatissima* this species is fully herbivorous, feeding on the leaves, fruits and flowers of a wide variety of plants, and is versatile in its habitat choice. Observations on Saba show that the animals can inhabit areas with sparse vegetation (van den Burg et al., 2022a), such as the lower southeast slopes of the island with a high percentage of grass. Food availability is probably not a limiting factor, even in areas that are heavily grazed or overgrown by the invasive coralita vine; preliminary genetic data from microbiome samples show that Saba Green Iguanas do eat some coralita.

Disturbance/mortality: As indicated by van den Burg et al. (2022a), recruitment appears to be low within the Saba population, which is likely being affected by the large feral goat population (Lotz et al., 2020) and the island-wide feral cat population (Debrot et al., 2014). Feral goats are known to have strong negative impacts on nesting sites, both being able to destroy the site itself as well as to trample the incubating nests (Alberts, 2004), whilst cats are known to predate, even multiple hatchlings from the same nest (van den Burg et al., 2018). A study on nest site availability and quality, as well as recruitment of young animals is urgently needed.

Within a one-month period during August-September of 2021, we recorded three large adults that had become victims of car collisions. Given that highest iguana densities occur in urban habitat, we recommend a study on road-mortality during the nesting season, when female iguanas migrate outside their home range and need to cross roads to reach nest sites.

Minimum size viable population: a minimum viable population (MVP) means a 5% extinction risk within 100 years. Although a MVP for small-island populations within the *Iguana iguana* species complex has not been proposed, the proposed MVP for *Iguana delicatissima* can be used as a substitute; 5000 individuals (Breuil, 2002). The population of Saba is larger than this proposed MVP, estimated at >6,000 (van den Burg et al., 2022a).

Present Distribution and Reference Values

In pre-historic times, when the species arrived on Saba, it likely established itself across the entire island, except for the higher slopes of Mount Scenery. Especially the windward side of the island has suitable and high-quality habitats for the iguana under natural conditions, while instead the habitats on the northwest-to-north side (the leeward side) often fall in the shade of clouds that surround Mount Scenery. Currently, and in contrast to in former times when forests dominated at lower altitudes (de Freitas et al., 2016), much of the east and southern lower elevations (>400 m) are barely vegetated or have large patches of grass due to the large feral goat population; iguanas are present, and even in high densities, but only together with large boulders (for shelter) and suitable vegetation. It remains unclear why there are so few iguanas in Spring Bay, despite its highly heterogeneous landscape and vegetation structure.

In 2021, morphologic and genetic data of 58 iguanas demonstrated the presence of non-native iguanas on Saba. A rapid action campaign was employed to identify their distribution across the island and start their removal. Prior to 2024, eight non-native iguanas have been removed, and non-native presence has been confirmed from the immediate and approximate vicinity of the Fort Bay harbour, as well as the northern part of Windward Side village, and along the road towards Zion's Hill (Figure 1). During December 2024 another 14 non-native iguanas were removed from these two regions on Saba, whilst at least another six non-native iguanas remain present. At least another nine iguanas are of doubtful origin and an island-wide assessment to understand the presence and distribution of non-native iguanas is urgently needed.

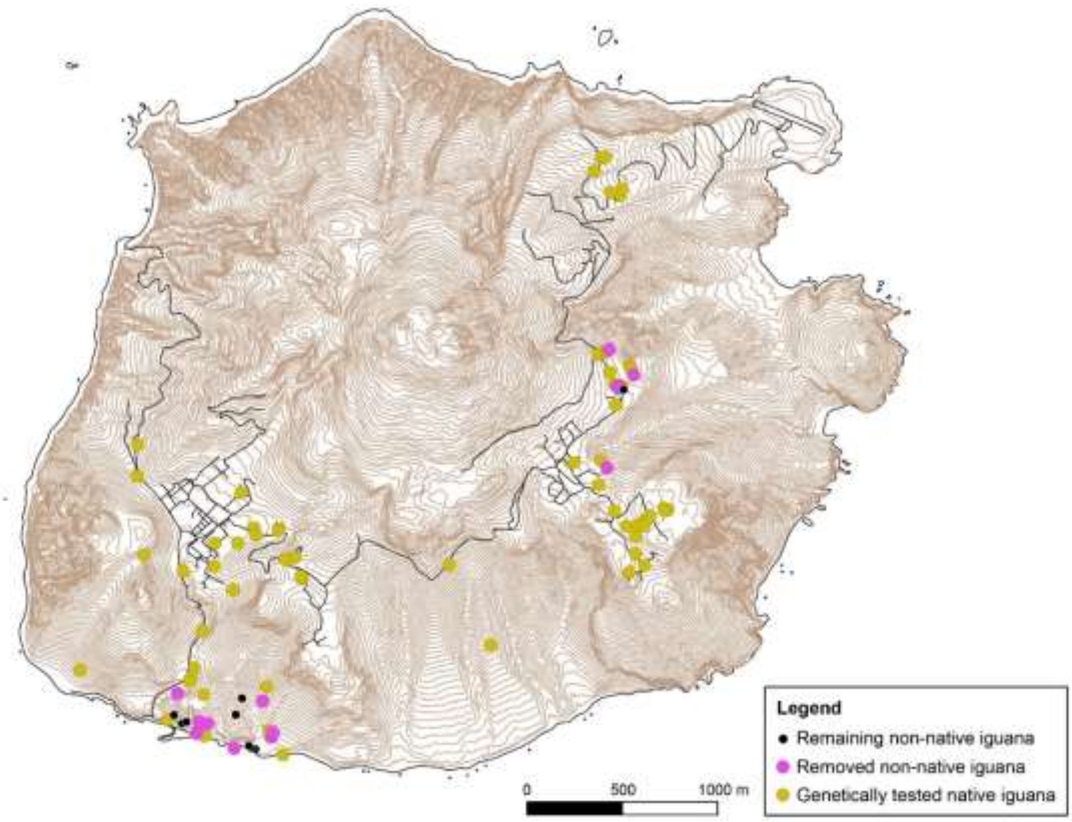

Figure 1. *Distribution of non-native and genetically assessed native iguanas on Saba.*

Reference values for population size and distribution on Saba: 15,000

Apart from elevations above 550 m, available habitat for the iguana population is present across ~11.75 km2. Van den Burg et al. (2022a) found that iguana densities are highest in urban habitat (15 ind/ha) and the *Aristida-Bothriochloa* habitat (8 ind/ha) which runs along the southern lower slopes of the island (de Freitas et al., 2016). These densities are much lower than found for other populations, like in a recent study from a native population in Colombia (46-75 ind/ha; Ramos et al., 2023), or from the non-native population in Puerto Rico (223 ind/ha; López-Torres et al., 2012). As habitats across Saba have been affected by feral goats, it is not possible to assess how "natural" the current iguana densities are, or to what extend it is recovering from the major hurricane season of 2017 (van den Burg et al., 2022b). A conservative density of 30 ind/ha for areas below 500 m, excluding the north(western) region of the island, would result in a pre-human island population estimate of ~15,000 animals.

Assessment of National Conservation State

Trends: The exact arrival of the melanistic iguanas on Saba might have been through translocation by prehistoric Amerindian inhabitants (Breuil et al., 2020; van den Burg et al., in prep). As only a single and first population assessment has currently been performed, no more recent population trend can be identified at this stage. However, compared to pre-colonial times, we believe the population size has declined; Currently, the population is likely recovering from the major hurricane season of 2017, which reduced the *Iguana delicatissima* on neighbouring St. Eustatius presumably by 25% (van den Burg et al., 2022b).

Recent developments: Troubling is the recent discovery of non-native iguanas on Saba which are present in two different areas; around the Fort Bay harbour, and on the northern edge of Windward Side towards Zion's Hill. Morphological data suggests hybridization is already ongoing. A few morphological characteristics have so far been identified that can aid identification of non-native iguanas in the field, but more study is necessary for 100% accurate identifications (van den Burg et al., 2023). Preliminary data from cloacal samples of both native and non-native iguanas from Saba furthermore suggests that non-native iguanas have introduced bacteria (e.g., *Devriesea agamarum* and *Mycoplasma iguanae*) and ectoparasites which have spilled over to the native population. This is similar as on St. Barthelemy where non-native iguanas from St. Martin/St. Maarten have introduced the bacterium *Devriesea agamarum* (Hellebuyck et al., 2017).

In recent years it has become evident that Saba Green Iguanas have been illegally taken from Saba and transported to St. Maarten (van den Burg and Weissgold, 2020). Genetic analyses of melanistic iguanas in the pet trade proof that illegal trade has occurred and show that the non-native iguana population on St. Maarten has been used to "white-wash" illegal wild-caught Saba Green Iguanas for trade purposes (Mitchell et al., 2022). Conservation and iguana experts have called for a complete halt of the live trade in *Iguana* both from and between Caribbean islands (van den Burg et al., 2022c).

Assessment of distribution: Favourable
The unique and endangered Saba Green Iguana is present throughout most of the island, with most animals occurring at medium elevations, between 180-390 m (van den Burg et al., 2022a). There is an apparent distribution gap on the eastern slopes below 350 m, with areas which hold far less iguanas compared to similar areas on the southern slopes (see below). Highest densities are found in and around urban areas, which likely results in anthropogenically-induced conflicts and mortality, though no study has yet assessed these threats.

Assessment of population: Favourable
Our preliminary estimate of current population size indicates minimally 6,000 iguanas occur on Saba (van den Burg et al., 2022a). This is marginally above a proposed MVP of 5000 for the closely related *I. delicatissima* (Breuil, 2002). As hurricanes can reduce iguanid population sizes by 25% (van den

Burg et al., 2022b), a direct hit by a major hurricane could quickly see the population size fall below the MVP. Periodic population size monitoring should be given priority. No study on the genetic diversity of the native population has yet been performed.

Assessment of habitat: Favourable
Although habitat availability is not limiting the iguana population, habitats on Saba have been negatively affected by a large feral goat population for decades (Lotz et al., 2020). An eradication campaign is ongoing through which many goats have already been removed, with reforestation and recovery of understory vegetation (hopefully) underway. However, goats are extremely prolific and annual removal rates typically need to be well above 50% to achieve significant reductions within a 4–5-year timeframe (Debrot, 2016). As goat reductions since 2020 appear to have removed about 90% as per the end of 2024, it means that removal rates on Saba have been suitably high. Much of the southeastern and eastern mid- and low-elevation slopes are degraded by goats and iguanas are much less abundant if present at all. Once (if) restored, these areas could sustain high numbers of iguanas as shown by the higher densities of iguanas typical of less-degraded areas within similar vegetation and elevation. Habitat quality for now does not seem to be the major limitation to iguana population size or distribution nor a long-term threat.

Assessment of future prospects: Unfavourable-bad
In the absence of improved biosecurity to halt incursions and a structural financial system for extermination of non-native iguanas, it is unlikely that further hybridization can be prevented. This is predicted to ultimately cause the local extinction of this unique population.

Table 2. *Summary overview of the status of the Saba Green Iguana in the Caribbean Netherlands (only Saba) in terms of different conservations aspects.*

Aspect of Saba Green Iguana	**2024**
Distribution	Favourable
Population	Favourable
Habitat	Favourable
Data quality and completeness	Unfavourable-bad
Future prospects	Unfavourable-bad
Overall Assessment of Conservation State	**Unfavourable-bad**

Comparison to the 2018 State of Nature Report

This is the first CS assessment made for the Saba Green Iguana and hence no comparison can be made to any earlier report.

Recommendations for National Conservation Objective

Safeguard the species from hybridization and non-native incursions, study the presence and impacts of bacterial and parasite pathogens spillover to native reptile species, as well as gain a better understanding on factors endangering nesting and recruitment in this unique melanistic iguana population within the *Iguana iguana* species complex.

Goals:

a) Improvement of regional and local biosecurity to halt non-native iguana incursions; urgently needed prior to imports for construction of Black Rock harbour
b) Island-wide survey and eradication of all non-native iguanas and hybrid iguanas
c) Better understand pathogen threats due to bacteria and parasite spillover from non-native iguanas to native reptilian species
d) Map and study nest sites and recruitment; allowing protection and mitigation during spatial development projects

Key Threats and Management Implications

The major threat to the survival of the Saba Green Iguana is the presence of non-native green iguanas and potential hybrids, including their continuous incursions from (principally) St.-Martin/St. Maarten and associated with the importation of goods. The NEPP for the Caribbean Netherlands assigns a high priority to invasive species threats like this one (Min. LNV et al., 2020). If non-native iguanas are not continuously identified and eradicated, long-term survival of a pure and unique melanistic *Iguana iguana* population on Saba is impossible. These non-native iguanas have introduced invasive and detrimental bacteria as well as ectoparasites that can harm both the native iguana population as well as other native reptiles. A broad study is urgently necessary to understand the diversity of introduced pathogens like bacteria and viruses, as well as their spread to native reptilian species on Saba including the unique native Saba Green Iguana. A further anthropogenetic threat is the illegal trade in the Saba Green Iguanas, which have been taken to neighbouring St. Martin/St. Maarten for subsequent shipments across the globe. More strict cargo control between Saba and St. Maarten could prevent further illegal trade. Lastly, no study on iguana nesting availability and distribution has been performed on Saba, which is impeding legal protection and conservation of these sites. Likely nesting sites have been negatively impacted by the feral goat population, thereby reducing iguana recruitment. Nesting locations should be rapidly mapped so these can be assessed in terms of quality and functionality and provide a baseline data set for further studies (e.g., on recruitment). We note that it is unknown whether large communal nesting sites are present on Saba, and whether those are present in areas (to be) identified for future spatial development, e.g., the Black Rock harbour. It is unknown whether the presence of iguana nesting sites has been assessed within an environmental impact assessment for the Black Rock harbour project.

Data Quality and Completeness

Knowledge about the presence of non-native iguanas is sufficient to document the poor Conservation State of the Saba Green Iguana, however most other population details remain un(der)studied. For example, the presence, quality and distribution of nesting sites remains a high-importance conservation issue, including the recruitment rates for young animals within the population. Small iguanas and iguana nests may also fall prey to the many rats present on the island. As this is a baseline assessment, there has yet not been any continuous or repeated effort to monitor population size or trends. An assessment of anthropogenic mortalities, which can be high as shown by data from St. Eustatius (Debrot and Boman, 2014; van den Burg et al., 2018b), has likewise not been performed so far. Effort is best invested in the prevention of further non-native/hybrid iguana incursion by stringent control at ports of entry, the removal of any or all non-native iguanas, and an assessment of present non-native bacteria and diseases. Further efforts should be directed towards knowledge on nesting and recruitment characteristics.

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18 Conservation State of the Bridled Quail-Dove in the Caribbean Netherlands

Madden, H. 2025. From: State of Nature Report for the Caribbean Netherlands, 2024. WUR report C001/25.

Status

Table 1. *Overview of the international Conservation State of the Bridled Quail-dove.*

Name				IUCN category	SPAW Annex	CMS Annex	CITES Appendix
Scientific	**Common**	**Local**	**Dutch**				
Geotrygon mystacea	Bridled Quail-Dove	-	Grote Kwartel-duif	LC	-	-	-

The current IUCN classification of the Bridled Quail-Dove *Geotrygon mystacea* is Least Concern (LC). The species is assumed to be declining across its range (Boal and Madden, 2021). However, except for one published study from St. Eustatius (Rivera-Milán et al., 2021), recent quantitative estimates of the species that would enable insights into population trends are fully lacking.

Characteristics

Description: The Bridled Quail-Dove (*Geotrygon mystacea*) is similar in appearance to the Key West Quail-Dove (*Geotrygon chrysia*) to which it is most-closely related but a bit more brownish rather than reddish or purple on the upper dorsal areas. It is a medium-sized dove, about 28–31 cm (11–12 inches) in length. Plumage is primarily dark brown, with a distinctive green iridescent sheen and shades of blue/violet on the nape, neck and upper back. The flight feathers are cinnamon-red. A striking white stripe runs from the bill, below the eye, giving it the 'bridled' appearance.

Relative Importance within the Caribbean: extremely high
The Bridled Quail-Dove is endemic to the West Indies. Once common and widespread throughout the U.S. Virgin Islands and Lesser Antilles, currently only scattered remnant populations exist (Steadman et al., 2009). Its current range extends from Puerto Rico and the Virgin Islands in the north-eastern Caribbean to St. Lucia in the south, however it does not occur on all islands. In the Dutch Caribbean it is a breeding bird on Saba and St. Eustatius. One adult was confirmed from neighbouring St. Martin (Brown and Newman, 2007) but thought to have originated from elsewhere and brought in by a storm. The species is absent from Anguilla and St. Barthelemy (Boal and Madden, 2021). In Puerto Rico, its eastern-most island of occurrence, where it shares the island with the more-widely distributed Key-West Quail-Dove and Ruddy Quail-Dove (*G. montana*), it is listed as a rare breeding priority species for conservation.

Ecological Aspects

Habitat: The species occupies montane tropical forests with a dense understory and deep leaf litter layer (Raffaele et al., 1998). It has been observed in gulleys and along slopes (Robertson, 1962; Diamond 1973; Chipley, 1991; McNair et al., 2005; Boal, 2018) but may also forage in leaf litter under forests of sea grape (*Coccoloba uvifera;* Boal, 2018). On St. Eustatius it occurs only in the Quill

National Park at elevations above 200m (Rivera-Milán et al., 2021). On Saba, Bridled Quail-Doves primarily occupy rainforest and transitional forest habitats at elevations ~350-600 m (Madden, 2024).

Food: Bridled Quail Doves are primarily granivorous and frugivorous. They forage exclusively on the forest floor for fallen fruits and seeds, including *Roystonea regia*, *Momordica charantia* L., *Calophyllum antillanum*, *Arthrostylidium capillifolium*, *Croton spp.*, *Solanum torvum Sw., Murraya paniculata, Eugenia* spp., *Amyris elemifera, Bursera simaruba,* and *Capparis* spp. (Seaman, 1966; Chipley, 1991; Yntema et al., 2017; Rivera-Milán et al., 2021; Boal and Madden, 2021). They occasionally also consume small mollusks and reptiles (Boal 2008).

Minimum viable population size: a minimum viable population (MVP) means a 5% extinction risk within 100 years. Determining the MVP for Bridled Quail-Doves involves various factors such as genetic diversity, environmental variability and demographic parameters. MVP sizes should be large enough to maintain genetic diversity and ensure population resilience to environmental stochasticity and demographic fluctuations, which could range anywhere from a minimum of 500 to a few thousand individuals (Frankham et al., 2014). A population viability analysis has not been conducted on the species.

Present Distribution and Reference Values

Figure 1. *Current distribution (dark orange) of the Bridled Quail-Dove (source: Birds of the World).*

Figure 1 shows the current distribution of *Geotrygon mystacea*. It is a native breeder on many Caribbean islands, including Saba and St. Eustatius, whose montane forest habitats support local breeding populations (Boal and Madden, 2021). Bridled Quail-Doves do not occur on the leeward islands of Bonaire, Curaçao or Aruba (Birdlife International, 2024).

Reference values for population size and distribution:
Rivera-Milán et al. (2021) estimated the St. Eustatius population declined from 1,038 (± 156) individuals in 2016 and 2017 to 238 (± 98) individuals in 2019. In 2022 the species had further declined to 55 ± 20 individuals (Madden unpubl. data). Follow-up surveys in 2023 and 2024 suggest there has been no recovery of the local population (STENAPA, unpubl. data). General bird monitoring data from Saba confirm the species is present in two of the three vegetation types surveyed (Madden, 2024).

Bambini et al. (2017) estimated the 2016 Montserrat population at 411 individuals (min 250–max 853); Boal's (2018) population estimate for Guana Island (British Virgin Islands) was 429 (± 127 SE) individuals; Levesque et al. (2020) detected Bridled Quail-Doves in 19.8% of island-wide bird monitoring on Guadeloupe between 2014 and 2019. Similarly, Jean-Pierre et al. (2022) suggest Bridled Quail-Doves are common and relatively abundant in Guadeloupe based on camera trap data. Askins and Ewert (2019) consider Bridled Quail-Dove a rare species in St. John (U.S. Virgin Islands), where standardized bird counts were insufficient for assessing population size or trends.

Assessment of National Conservation State

Trends in the Caribbean Netherlands:
Due to data deficiency, regional population trends are mostly unknown, although the species is in severe decline on St. Eustatius (Rivera-Milán et al., 2021). Based on bi-annual landbird monitoring data, it is thought to be stable on Saba (Madden, 2024). Predation of Bridled Quail-Dove eggs and chicks by non-native cats, rats and mongooses is thought to negatively affect local populations across the species' range (Boal and Madden, 2021). Fortunately, the mongoose which is present on St. Martin is absent from Saba and St. Eustatius, but cats and especially rats are especially abundant in its most important habitat on Saba (Debrot et al., 2014). It might be especially vulnerable to nest predation as it typically nests in vines, shrubs and trees at low height. In addition, native predators may include Pearly-eyed Thrashers (*Margarops fuscatus*), land crabs (Gecarcinidae), birds of prey and snakes such as *Alsophis* spp. (Rivera-Milán and Schaffner, 2002; Boal and Madden, 2021).

Recent developments:
Local populations are vulnerable to severe hurricane events (Wauer and Wunderle 1992, Boal and Bibles 2020). This is the case on St. Eustatius where the local breeding population in the Quill National Park has experienced a drastic decline following two category 5 hurricanes in 2017 (Rivera-Milán et al., 2021). While local conservation measures are being implemented across the dove's habitat (Erroi pers. comm.), the species is currently at risk of extirpation.

Assessment of distribution: Favourable
While scattered populations of Bridled Quail-Doves remain in the Lesser Antilles, Puerto Rico and Virgin Islands, individuals are not known to fly between islands. This makes breeding populations vulnerable to local disturbance (e.g., habitat alteration/loss, hunting, predation, urbanization, agriculture), but also to large-scale events such as hurricane impacts, volcanic eruptions, and climate change (Wauer and Wunderle, 1992; Dalsgaard et al., 2007; Oppel et al., 2014; Boal and Bibles, 2020; Rivera-Milán et al., 2021).

Assessment of population: Unfavourable-bad
Due to data deficiency, the species' current regional population trend has not been assessed but is likely decreasing due to predation by invasive mammals and habitat loss/alteration/degradation (Boal and Madden, 2021). Possibly with exception of Guadeloupe, all available studies are indicative of fairly small and vulnerable populations (St. Eustatius, Guana Island, St. John's, Montserrat). Unfortunately, on most islands where the species still exists, there are little to no published studies that quantify the rate or intensity of population declines (except for St. Eustatius; Rivera-Milán et al., 2021). The current trends in most breeding populations are thus unknown.

Assessment of habitat: Unfavourable-inadequate
Bridled Quail-Doves exclusively inhabit forested areas (Boal and Madden, 2021), with a preference for tropical rainforest vegetation (Jean-Pierre et al., 2022). On St. Eustatius they occur only in the Quill National Park; on Saba they can be found in the rainforest of Mount Scenery and transitional forested areas (between the rainforest and dry forest; Madden, 2024). Breeding adults construct nests in vines, shrubs and trees typically at a low distance from the ground (Boal and Madden, 2021).

Table 2. *Summary overview of the status of the Bridled Quail-dove in the Caribbean Netherlands (only Saba and St. Eustatius) in terms of different conservations aspects.*

Aspect Bridled Quail-Dove	**2024**
Distribution	Favourable

Population size	Unfavourable-bad
Habitat	Unfavourable-inadequate
Future prospects	Unfavourable-inadequate
Overall Assessment of Conservation State	**Unfavourable-bad**

Comparison to the 2018 State of Nature Report

This is the first CS assessment made for the Bridled Quail-dove in the Caribbean Netherlands and hence no comparison can be made to any earlier report.

Recommendations for National Conservation Objectives

Bridled Quail-Dove populations are largely island-bound and not known to travel between islands (Boal and Madden, 2021). This dependence on and sensitivity to local habitats makes the species extremely vulnerable to changes in them. The Caribbean Netherlands 2020-2030 Nature Environment Policy Plan lists the Bridled Quail-Dove as a flagship species (Annex 1; Min. LNV et al., 2020). It is recommended to enact local legislation to confer legal protection of Bridled Quail-Doves on Saba and St. Eustatius, as well as their nesting and foraging habitats (DCNA Bird Conservation Group, 2022). Conservation strategies could include feasibility studies into invasive predator removal or control (e.g., rats), roaming ungulate removal, and habitat restoration (Boal and Madden, 2021; SAP).

Key Threats and Management Implications

Disturbance: Bridled Quail-Doves are hunted on Guadeloupe (Levesque et al., 2020, Jean-Pierre et al., 2022), Montserrat (Hilton et al., 2006) and St. Kitts and Nevis (Cooper et al., 2011), however the impact of this on local populations is unknown. On St. Eustatius and many other islands where the Bridled Quail-Dove exists, habitat loss and degradation from free-ranging livestock (Madden, 2020) and/or anthropogenic activities is widespread (Skipper et al., 2013).

Predation: Predation pressure on breeding populations on Saba and St. Eustatius has not been quantified but invasive rats *(Rattus rattus*) and cats *(Felis catus*) are assumed to impact Bridled Quail-Dove eggs and chicks (Boal and Madden, 2021). To date there are no reports of mongooses being present on Saba or St. Eustatius (Nellis and Everard, 1983). No information exists about number of eggs laid, eggs hatched, or chicks fledged on either island.

Climate change: Increasing sea-surface temperatures have resulted in more frequent, severe hurricanes across the northeastern Caribbean in recent decades (Hernandez et al., 2024). The 2017 hurricane season was particularly detrimental for Bridled Quail-Doves on St. Eustatius (Rivera-Milán et al., 2021), and hurricane impacts are likely to continue to negatively affect populations. The initial, short-term impacts (i.e., direct mortality) of hurricanes on populations appear less severe than the long-term impacts (i.e., habitat destruction, increased risk of predation, and loss of food resources; Wauer and Wunderle, 1992). Given that Bridled Quail-Doves do not fly between islands (Boal and Madden, 2021), any loss of or damage to forest habitats would negatively impact local populations. During hurricanes, birds may be swept out to sea or to nearby islands (Brown and Newman, 2007). Bridled Quail-Doves are sensitive to ash-fall from volcanic eruptions (Dalsgaard et al., 2007; Oppel et al., 2014).

Genetic isolation: Since there is no evidence of natural movement between islands, breeding populations on Saba and St. Eustatius are thought to be quite genetically isolated. Genetic assessment of the species across its range would allow insights into dispersal rates and the potential for recolonization following extirpation (Boal and Madden, 2021).

Data Quality and Completeness

Research and monitoring of Bridled Quail-Doves remains generally lacking, as do insights into the species' ecology. Long-term quantitative assessments of the total number of individuals, estimated nesting pairs and nest survival would provide more accurate insights into the status of current breeding populations (Boal and Madden, 2021; DCNA Bird Conservation Group, 2022). Standardized counts and collection of demographic data would allow a better assessment of population sizes and trends on Saba and St. Eustatius; banding and telemetry studies could provide insight into seasonal movements and resource limitations/requirements (Boal and Madden, 2021).

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19 Conservation State of the Red-billed Tropicbird in the Caribbean Netherlands

Henkens, R. J. H. G., Madden, H. and Leopold, M. 2024. From: State of Nature Report for the Caribbean Netherlands, 2024. WUR report C001/25.

Status

Table 1. *Overview of the international Conservation State of the Red-billed tropicbird.*

Name				IUCN category	SPAW Annex	CMS Annex	CITES Appendix
Scientific	**Common**	**Local**	**Dutch**				
Phaethon aethereus	Red-billed tropicbird	-	Roodsnavel-keerkringvogel	LC	-	-	-

The current IUCN classification of the Red-billed Tropicbird *Phaethon aethereus* is Least Concern (LC). The species does not approach the thresholds for vulnerable (VU) under the criteria for range size, population size and population decline. Despite ongoing population declines, the trend is not yet believed to be sufficiently rapid to approach the threshold for vulnerable (Birdlife International, 2019).

This species Conservation State does not reflect the status of the different subspecies. The Caribbean subspecies *P.a. mesonauta* likely warrants a 'higher' IUCN-Conservation State based on evidence of severe predation in Caribbean breeding colonies (Debrot et al., 2014; Terpstra et al., 2015; Boeken, 2016; Eggermont and Madden, 2019; Orta et al., 2019; Madden et al., 2022).

Characteristics

Description: Superficially resembling a tern in appearance, the Red-billed Tropicbird has mostly white plumage with some black barring on the upper wings and back, a black mask and a red bill. It measures 90-105 cm, including its 46-56 cm long tail streamers. It has a wingspan of 99-106 cm (Hauber, 2014; Orta et al., 2017).

Relative Importance within the Caribbean: very high
The Red-billed Tropicbird is a seabird of tropical waters in the Caribbean Sea, Atlantic, Pacific and northern Indian Oceans. Three subspecies are recognized in different oceans, of which *P.a. mesonauta* is the most numerous. This subspecies breeds in both the Pacific and Atlantic Ocean and has one of its strongholds in the Caribbean region, where it breeds on various islands (Geelhoed et al., 2013). Circa 35% of the Caribbean population of *P.a. mesonauta* nest on Saba and St. Eustatius, (Koelega et al., 2020). The relative importance of the Caribbean Netherlands is therefore very high.

The subspecies has occasionally been recorded on and around the Dutch southern Caribbean islands (Bonaire, Curaçao and Aruba), although it is not a native breeder on these islands (Wells et al., 2022; Birdlife International, 2024).

Ecological Aspects

Habitat: Red-billed Tropicbirds are loosely colonial breeders, nesting in rocky crevices, under boulders, at the base of trees (Lee and Walsh Mc Gehee, 2000) and in sandy burrows (Geelhoed et al., 2013), preferentially on cliffs where take-off is easy (Birdlife International, 2024). They lay a single egg which is incubated by both parents for 40-45 days; upon hatching the chick remains in the nest for 80-90 days (Boeken, 2016; Madden et al., 2022). The breeding season on Saba and St. Eustatius runs from November-July (Geelhoed et al., 2013), with a peak from January-April (Sarmento et al., 2014).

Non-breeding Red-billed Tropicbirds are pelagic foragers that spend their lives at sea. During the breeding period, adults often disperse widely to forage for prey resources (Birdlife International, 2024). Bio-logged adults with small chicks at a breeding site on St. Eustatius travelled a maximum distance of 953.7 km from the colony, with an average trip length of 176.8 (± 249.8) km (Madden et al., 2022), while tracked birds breeding on Saba travelled a maximum distance of 553.7 km, with an average trip length of 117.2 ± 144.6 km (±SD) (Madden et al., 2023).

Food: Investigating the foraging preferences of tropical seabirds provides crucial information about their ocean habitat affinities as well as prey choice. However, until recently, foraging studies of Red-billed Tropicbird populations in the Caribbean had been scarce. Madden et al. (2022, 2023) tracked chick-rearing adults using GPS devices and sampled regurgitates at nest sites on St. Eustatius and Saba. The studies suggest that adults are solitary hunters, foraging extensively in search of Exocoetidae (flying fish), and other pelagic species, in patches of marine habitat with higher primary productivity. Diet samples from Saba were high in flying fish (70.73%), but also included squid (Loliginidae; 9.76%), flying gurnards (Dactylopteridae; 2.44%) and ray-finned fish (Carangidae; 2.44%; Madden et al., 2023). Diet samples from St. Eustatius were dominated by flying fish (59.5%) and needle fish (Belonidae; 14.9%; Madden et al., 2022).

Disturbance: foraging birds are entirely pelagic in all seasons, while breeding birds nest on steep, rocky cliff faces (Boeken, 2016, Madden et al., 2022). Disturbance by humans, including tourists, in these habitats is not considered to be significant.

Minimum viable population size: a minimum viable population (MVP) means a 5% extinction risk within 100 years. Determining the MVP for Red-billed Tropicbirds, or comparable pelagic seabird species, is not clear, since this involves various factors such as genetic diversity, environmental variability and demographic parameters. However, MVP sizes should be large enough to maintain genetic diversity and ensure population resilience to environmental stochasticity and demographic fluctuations, which could range anywhere from a minimum of 500 to a few thousand individuals (Frankham et al., 2014). A population viability analysis has not been conducted on the species.

Present Distribution and Reference Values

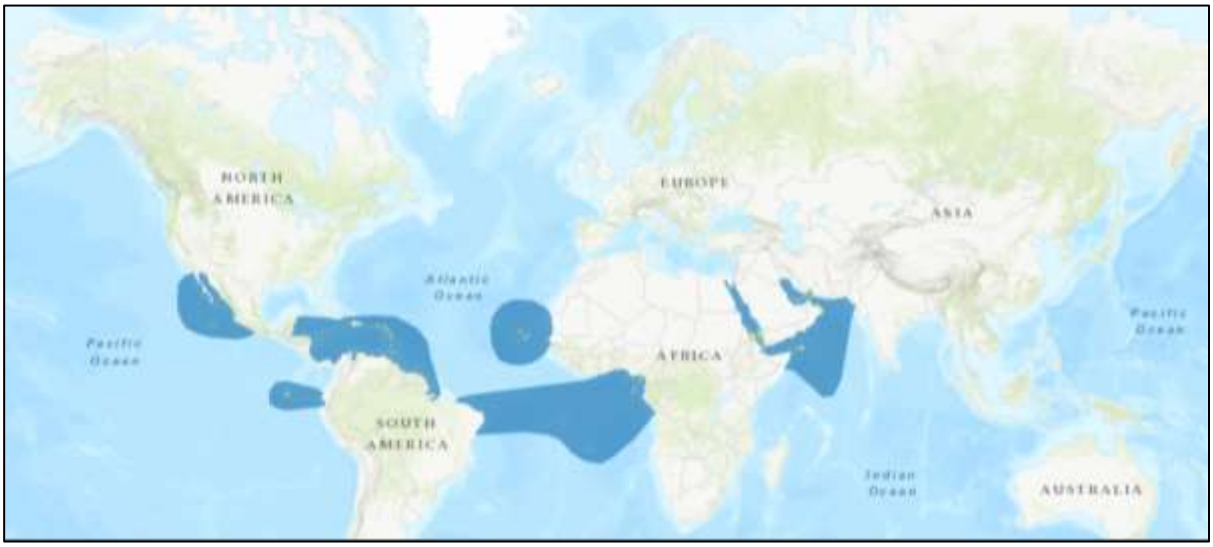

Figure 1. *Current distribution (dark blue) of the Red-billed Tropicbird (Phaethon aethereus; Birdlife International, 2024).*

The figure above shows the current distribution of *Phaethon aethereus*. The subspecies *P. a. mesonauta* breeds in the the Caribbean. It is a native breeder on many Caribbean islands, including Anguilla, Saba and St. Eustatius which support globally significant breeding populations (Soanes et al., 2016; Madden et al., 2022; Madden et al., 2023). The species does not breed on the leeward islands of Bonaire, Curaçao or Aruba (Birdlife International, 2024).

The global population is estimated at 16,000 - 30,000 individuals, and the most recent estimate of the Caribbean population (individuals) was 4,721 (EPIC 2011). However, based on estimates from Saba (Boeken, 2016) and St. Eustatius (Madden, 2019), the regional population is likely higher. Specifically, the Saba population is 55.0 – 63.5% of the most recent estimated regional total, and 13 – 15% of the estimated global population (BirdLife International, 2023); the St. Eustatius population is approximately 12 – 21.2% of the most recent estimated regional total and 3 – 10% of the estimated global population (BirdLife International, 2023).

Assessment of National Conservation State

Trends in the Caribbean Netherlands: The (sub)species current population trend has not been quantified; however, it is suspected to be in decline (Birdlife International, 2019). Evidence suggests that predation of Red-billed Tropicbird eggs and chicks by cats and rats is particularly severe in Caribbean breeding colonies, where globally important populations occur (Madden et al., 2022, 2023; Leopold and Boeken, 2020; Orta et al., 2019; Boeken, 2016).

In 2011/2012 and 2019/2020, the breeding success of Red-billed Tropicbirds (*Phaethon aethereus mesonauta)* was monitored on Saba (Boeken, 2016; Leopold and Boeken, 2020). Fledging success in two small colonies (approximately 100–300 nests) was zero in 2011/2012 due to cat predation, while in 2019/2020 some chicks successfully fledged. Breeding success in the island's largest colony (approximately 1,000 nests, both in 2011/12 and in 2019/20) was 65% in 2011/2012 (Boeken, 2016) but had decreased in 2019/2020 (Leopold and Boeken, 2020). Overall the 2019/2020 breeding population was smaller compared to 2011/12, despite the positive impacts of temporary cat culling and only a temporary halt to the cat TNR program (in which more than 1000 cats had been set loose already by 2014; see Debrot et al., 2014) between both research periods (Leopold and Boeken, 2020).

Recent developments:
Red-billed Tropicbirds were once thought to be rare visitors to the southern Caribbean, however, there has been an increase in sightings in recent years. Reasons for this are not clear but could be related to a shift in flight patterns due to changes in prey distribution, large-scale weather patterns, or climate change. It may also be related to an increase in skilled birders, or improved accessibility to online databases for archiving sightings of rare birds (DCNA, 2023).

Assessment aspects of natural area of distribution: Favourable
Red-billed Tropicbirds are a mobile species, and the Caribbean subspecies *P. a. mesonauta* is widely distributed across the Caribbean region.

Assessment aspect population: Unfavourable-inadequate
The species' current population trend is not known but is likely decreasing due to predation by cats and rats, as well as habitat loss. Unfortunately, in most cases, there is little to no published literature available that quantifies the rate or intensity of population declines. The current trend in most breeding colonies is thus unknown. Despite this, the species is suspected to be in decline across the region.

Assessment aspect habitat: Unfavourable-inadequate
The species' main habitats can be divided into foraging habitat at sea and breeding habitat on Saba and St. Eustatius. Although formerly a native breeder on St. Maarten, the species no longer exists there.

Assessment aspect future prospects: Unfavourable-inadequate
The predation pressure on breeding colonies on both Saba and St. Eustatius requires urgent and continued management efforts. The potential direct and indirect impacts of climate change are not clear but are thought to threaten breeding and foraging habitats of the species.

Recommendations for National Conservation Objectives

National conservation objectives:
Due to their pelagic nature and extensive foraging ranges (Madden et al., 2022; 2023), Red-billed Tropicbirds cross multiple political boundaries and exclusive economic zones, thus requiring regional as well as local protection.

It is recommended to enact local legislation to confer legal protection of Red-billed Tropicbirds, as well as their nesting sites (Madden, 2023). This should entail the culling of cats and rats and the termination of the TNR program (Debrot et al., 2023).
The Caribbean Netherlands 2020-2030 Nature Environment Policy Plan (NEPP) lists the Red-billed Tropicbird as a protected species (Annex 1; Min. LNV et al., 2020).

Table 2. *Summary overview of the status of the Red-billed Tropicbird in the Caribbean Netherlands (only Saba, St. Eustatius) in terms of different conservations aspects.*

Assessment of Conservation State	
Aspect	**2024**
Distribution	Favourable
Population	Unfavourable-inadequate
Habitat	Unfavourable-inadequate
Data quality and completeness	Unfavourable-inadequate
Future prospects	Unfavourable-inadequate
Overall Assessment of Conservation State	**Unfavourable-inadequate**

Comparison to the 2018 State of Nature Report

This is the first CS assessment made for the Red-billed tropicbird in the Caribbean Netherlands and hence no comparison can be made to any earlier report.

Key Threats and Management Implications

Predation by cats and rats: predation by cats and rats is currently the primary threat faced by breeding Red-billed Tropicbirds on Saba and St. Eustatius. However, the culling of cats may give rise to an increase in the number of rats, meaning both should be suppressed simultaneously. However, due to the size and nest defence the Red-billed Tropicbird is less sensitive to rat predation than many other smaller and more docile seabirds (Van Halewijn and Norton, 1984, Campbell, 1991). On Saba the Red-billed Tropicbird colonies only really came into trouble after unwanted cats started being let loose into the wild instead of being humanely euthanized as part of a Trap-Neuter-Release campaign. TNR (Trap-Neuter-Return) was introduced to Saba to avoid euthanizing unwanted cats but the massive collateral animal suffering (of nesting seabirds) (Debrot et al, 2014; Terpstra et al., 2015) was not taken into consideration. More recently, there have been several initiatives which included (or varied between) both euthanizing and TNR of cats. Currently, this combination is used as well, where kittens are re-homed and aggressive adult cats are euthanized. These isolated activities of private veterinarians, not of an established conservation policy. Legal registration and required neutering of all pets has been proposed in the past. Relevant animal welfare legislation is currently being developed on Saba. Additionally, a new program is implemented focused on trapping and euthanizing cats around the Tropicbird nesting sites (in combination with the use of a rattus-specific poison to remove rats). So, feral cats are included in the current invasive species control pilot project between PES and SCF and will be targeted for removal in the coming years. There does not seem to be a strong public opinion against the removal of cats (PES, pers. comm.). Studying islander attitudes, Debrot et al. (2014) indeed found that when asked to choose, Sabans greatly valued the life of a tropicbird over the life of a cat. Therefore, Debrot et al., (2022) have recommended a more species-inclusive perspective on animal welfare that also takes collateral animal suffering in consideration that is caused by letting cats loose in the wild.

Climate change: Increasing sea surface temperatures have resulted in more frequent, severe hurricanes across the northeastern Caribbean in recent decades (Hernandez et al., 2024). While it is not clear what this means for Red-billed Tropicbirds, severe storms can lead to mass mortality of seabirds. Whereas some pelagic seabird species have been observed flying into the eye of the hurricane, or to avoid approaching storms to minimize risks (Weimerskirch & Prudor, 2019; Lempidakis et al, 2022), there are reports of mass mortality due to starvation or wrecks during/after heavy storms (Hass et al., 2012; Clairbaux et al., 2021). Besides this, the indirect impacts of climate change may affect prey availability, since flying fish (tropicbirds' main food source) are thought to be increasingly vulnerable to climate change-related stressors (Butt et al., 2022; Putri et al., 2023). Such shifts in prey distribution, abundance and availability associated with changing marine conditions could increase Red-billed Tropicbird breeding stress and reduce overall productivity (Madden, 2023).

Genetic isolation: Since there is no evidence of movement between nesting islands, Leopold and Boeken (2020) suggest that the breeding populations of Saba and St. Eustatius are genetically isolated, despite their geographic proximity (approx. 25 km). If this is the case, the risk of local extirpation increases, strengthening the need for management actions. Furthermore, high fidelity in general may cause inbreeding depression or other genetic risks in the future.

Effects of hurricanes: Given that Red-billed Tropicbirds exhibit high fidelity to their nest cavities (Madden 2019), any loss of or damage to nest sites would negatively impact local populations. Increasingly, birds may be lost at sea if hurricane frequency and intensity increase.

Data Quality and Completeness

Research and monitoring efforts on Red-billed Tropicbirds have increased in recent years, which have provided additional insights into the species' ecology. However, detailed, long-term quantitative assessments of the total number of individuals, estimated nesting pairs or apparently occupied nests, and additional birds on Saba and St. Eustatius are required. Without these it is difficult to assess whether the active breeding populations are decreasing, increasing or stable (Madden, 2023).

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20 Conservation State of the Breeding Terns of the Caribbean Netherlands

Debrot, A. O., Bertuol, P., DeAnda., D., Boeken, M. and van Slobbe, F. 2024. From: State of Nature Report for the Caribbean Netherlands, 2024. WUR report C001/25.

Status

Seabirds, among which terns and especially the migratory species, are considered one of the most highly threatened groups of vertebrates with almost half of the species listed as globally threatened or near threatened (Phillips et al., 2023). However, according to the IUCN classification, all species of terns known to breed within the Caribbean Netherlands (which excludes the Dutch islands of Aruba, Curaçao and St. Martin) are classified as LC species: "Least Concern." An "LC" species is one that has been evaluated by the IUCN but does not qualify for a higher protection status. As a result, such species are not considered “globally” threatened. However, the situation is different when viewed from a regional Caribbean perspective. From a regional standpoint, *Sterna dougalli* is considered "Endangered," *Sterna hirundo* as "Critically Endangered," *Sternula antillarum* as "Vulnerable," *Thalasseus maximus* as "Endangered," and *Thalasseus sandvicensis eurygnatha* as "Vulnerable" (Schreiber, 2000; USFWS, 2010).

Table 1. *Overview of the international Conservation State of the breeding terns of the Caribbean Netherlands. DD = data deficient, CE = critically endangered, EN = endangered, LC = least concerned, VU = vulnerable. None of the species has a CITES listing.*

Name				Island	IUCN category	Regional categ.	SPAW Annex	CMS Annex
Latin	common	local	Dutch					
Sterna dougalli	Roseate Tern	Bubi chiki	Dougalls Stern	Bonaire	LC	EN	2	2
S. hirundo	Common Tern	Bubi chiki	Visdief	Bonaire	LC	CE		
Sternula antillarum	Least Tern	Meuchi	Amerikaanse Dwergstern	Bonaire	LC	VU	2	
Thalasseus maximus	Royal Tern	Bubi chiki	Konings stern	Bonaire	LC	EN		
T.s. eurygnatha	Cayenne Tern	Bubi chiki	Amerikaanse Grote Stern	Bonaire	LC	VU		
Anous stolidus	Brown Noddy		Bruine Noddy	Saba	LC	LC		
Onychoprion anaethetus	Bridled Tern		Brilstern	Saba	LC	LC		
Onychoprion fuscatus	Sooty Tern		Bonte Stern	Saba	LC	DD		

Characteristics

Description
Terns are medium-sized seabirds, usually with grey or white plumage and often with a partially black head. They are elegant birds with long tails and long wings. Most terns hunt fish by diving. They often hover above the water surface to locate their prey. Terns breed in colonies.

Roseate Tern (Dougall's Tern): The plumage of this 38 cm long bird is white. It has a black crown, a long V-shaped tail, and orange legs. In the summer, the beak is black with a red base, while it is dark in winter. In the summer, the bird has a rosy belly with a blue-gray upper side.

Common Tern: The common tern reaches a body length of about 35 cm. It is a slender bird with a black cap and a deeply forked tail. The beak is orange-red with a black tip, and the legs are red.

Least Tern (American Little Tern): The least tern is the smallest tern (21 cm) in North America and is closely related to the European little tern. It is a small tern, 21-25 cm long. It has a black cap, but the forehead is white. The legs are yellow, and the beak is also yellow but with a black tip. The tail is short.

Royal Tern: The bird is 42 to 49 cm long. It has a long and deeply forked tail that is white on top. The orange beak is sturdy, and the head has a typical black cap with a crest.

Cayenne Tern (American Great Tern): This large tern (36-41 cm total length) has a black cap and crest and a yellow-orange and often curved beak.

Brown Noddy: Adults are dark brown with a white crown and forehead and range in sizes between 38-45 cm in total length and 75-86 cm. in wingspan.

Bridled Tern: Adult birds have a gray back, a black cap and white forehead (eBird.org). A medium-sized tern with a total length of 30-32 cm and a wingspan of 77-81 cm.

Sooty Tern: Adults have a black back and cap and a white forehead. It is a large tern with average sizes of 33-36 cm in total length and a wingspan of 82-89 cm.

Relative Importance Within the Caribbean

Very important for the Common Tern, the Cayenne Tern, and the Least Tern (Bonaire).

Bonaire
The Leeward Islands of the Netherlands Antilles, including Bonaire, have traditionally been known as a regionally important nesting area for at least three species of terns: the Common Tern, the Cayenne Tern, and the Least Tern. The Royal Tern is considered regionally endangered (Schreiber, 2000), as is the Roseate Tern (USFWS, 2010), but they are not found here in large numbers. The number of terns breeding on Saba, including offshore rocks of Saba, has generally been very small but possibly fairly consistent. On St. Eustatius, no terns are or have ever been known to breed.

Roseate Tern: This species is found worldwide, with an estimated global population of about 100,000 breeding pairs (Wetlands International, 2015). It is not known whether the population is increasing or decreasing, but there are currently no indications of concern for the species. The Caribbean subpopulation appears to be part of a larger metapopulation (Bradley and Norton, 2009). Throughout the region, colonies of this species are rather small, declining, and/or abandoning their historical breeding grounds (Birdlife International, 2015). Bradley and Norton (2009) estimated the Caribbean breeding population at about 5,400 pairs in 2007. Given the generally incomplete and qualitative nature of the data for most locations and the tendency of the species to change breeding sites relatively quickly, there is no clear indication that the population has increased or decreased in the past 30 years. Nevertheless, the species is considered endangered in the region (but not globally!) based on the small total number of individuals (USFWS, 2010). The relative significance of Bonaire as a breeding site within the Caribbean is unknown due to a lack of data but is likely limited. Newest insights are that about 30 pairs have been breeding quite consistently in southern Bonaire (Bertuol, pers. comm.).

Common Tern: The total population of this species in the West Atlantic area exceeds 200,000 breeding pairs. The total population in both South and North America together is probably more than 32,000 breeding pairs. Good protection in the U.S. in recent years has resulted in a substantial expansion of breeding colonies. The total breeding population for the Caribbean is about 960 pairs (Lee & Mackin, 2009a). This is a small number of individuals, making the Common Tern a regional priority species. Schreiber (2000) categorizes the species as critically endangered (CE) for the Caribbean region. Within the region, the ABC islands (including Bonaire) have always played an important role as a breeding area. However, today, this involves relatively few individuals.

Least Tern: The total population of this species on the American continents is probably more than 32,000 pairs, also due to good protection in the U.S., which has led to an increase in the number of individuals. For the Caribbean, the total is about 4-5 thousand breeding pairs (Lee & Mackin, 2008a). Schreiber (2000) classifies the species as vulnerable (VU) in the Caribbean. Many local breeding populations are declining due to beach development for tourism and recreational pressure. The ABC islands, including Bonaire, have always played an important role as a breeding site for this species. In 2002, Debrot et al. (2009) documented more than 790 breeding pairs on the island, 180 of which were in the Cargill area. For 2014, 2015, and 2016, Stinapa documented 581, 519, and 245 breeding pairs in the Cargill area, respectively. The importance of the Cargill area seems to have increased but in more recent years the numbers seem to be declining both in the Cargill area of southern Bonaire and in the Washington-Slagbaai National Park (Bertuol, pers. comm.). More research is needed.

Royal Tern: According to Lee & Mackin (2009b), the total population for the Northwest Atlantic area is about 70,000 breeding pairs (Kushlan et al., 2002). Thus, this species is not globally threatened. For the Caribbean, the breeding population is estimated at about 1680 breeding pairs. Therefore, for the Caribbean, the species should be considered endangered (EN) according to Schreiber (2000). The species has been eradicated on various islets of the U.S. Virgin Islands. The regional role of Bonaire as a breeding island is unknown due to a lack of data but may be greater than previously thought based on counts in 2014-2016. However, recent counts (P. Bertuol, pers. comm.) suggest a relatively stable breeding population of about 255 nests (count of 2022).

Cayenne Tern: This tern is found from Puerto Rico throughout the southern part of the Caribbean Sea and along the coast of South America to Argentina. There are no reliable population estimates for the species available. The Caribbean breeding population is estimated at 5 to 6 thousand breeding pairs (Lee and Mackin, 2009c). Therefore, the Cayenne Tern is considered vulnerable (VU) in the Caribbean (Schreiber, 2000). Within the region, the ABC islands (notably Bonaire) have always played an important role. In 2022, a total of 467 nests with eggs and/or chicks were counted in the Cargill area of southern Bonaire (P. Bertuol, pers. comm.).

Figure 1. *Nesting Royal Terns with chick and a Cayenne Tern in the Cargill salt pans of southern Bonaire. Photo: D. DeAnda.*

Saba

Brown Noddy: Boeken (2018) recorded this species nesting on Diamond Rock (17°38'51''N, 63°15'22''W) and Green Island (17°38'57''N, 63°13'47''W) off Saba, with an estimated total of 30–35 breeding pairs in June and July 2012. This agrees with Lee and Mackin’s (2009c) estimate of 30–60 pairs.

The Brown Noddy breeds worldwide on tropical and subtropical islands. Due to the large worldwide population size, it is considered a Least Concern species (Birdlife International 2020). Within the Greater Caribbean the species breeds with upwards of 12 thousand breeding pairs principally in the Us Virgin Islands, the Bahamas and the Jamaican offshore islands (Chardine et al. 2000a). Individual breeding colonies are generally small (20-200 pairs) (Chardine et al., 2000a).

Bridled Tern: Voous (1983) reported “at least 25 pairs” on Diamond Rock while Lee and Mackin (2009a) estimated for Saba “52–70 pairs in 3 colonies.” During June–July 2012, Boeken (2018) counted the nesting birds on Green Island eight times by spotting scope. No more than 10–15 nests; most were scattered in the lower parts of Green Island below the much more abundant Sooty Terns. During a boat trip near Diamond Rock on 14 July 2012, Boeken further observed 20–30 birds sitting on and flying around the rock. No other nesting site is known around Saba. The total number of breeding pairs appeared to be fewer than 50 during the 2012 season.

With an adult size of 30-32 cm, this species has a worldwide breeding population of upwards of 600 thousand breeding pairs and is considered a Least Concern species (Birdlife International, 2019). Within the Greater Caribbean this species nest with about 4-6 thousand breeding pairs, mostly in the Bahamas, Jamaica and US Virgin Islands (Chardine et al., 2000b).

Sooty Tern: Voous (1983) was unsure of the breeding status; Lee and Mackin (2008c) reported 15–30 breeding pairs in a single colony. Boeken (2018) observed a few birds on Diamond Rock in July 2012, while on Green Island he counted a colony of 75–85 nests in June and July 2012. On 30 June and 7 July 2023, Boeken found a previously unknown colony of about 70 Sooty Terns (*Onychoprion fuscatus*) nesting at Red Cliff on Saba's mainland (eggs and chicks observed).

The Sooty Tern with an adult size of about 43 cm, breeds worldwide on tropical and subtropical islands. Due to the large worldwide population size it is considered a Least Concern species (Birdlife International 2020). Within the greater Caribbean there are a few 100 thousands of breeding pairs (Saliva, 2000). Recent work shows that the largest nesting colony of Sooty terns in the Atlantic (Ascencion Island) has undergone a long-term decline in breeding pairs amounting to 84% decline between 1942 and 2005 (Hughes et al., 2017). The authors point out that even though total worldwide populations remain large and well-above the LC threshold, more research is needed providing more-detailed local assessments (Hughes et al., 2017) to redraft a more accurate global assessment. Within the Eastern Caribbean, seabird nesting (principally the Sooty Tern and Brown Noddy) on the important seabird island of Aves Island (Venezuela) has also drastically declined throughout the 20th century (Heatwole et al., 2022). The island now has permanent human disturbance and is rapidly declining in size due to a combination of factors such as especially climate change and sea level rise. Consequently, it is likely that this formerly key nesting area will continue to decline in importance and ultimately be lost (Heatwole et al., 2022).

Table 2. *Various estimates for the breeding terns of Saba, including new counts of breeding pairs for 2023.*

Common name	Latin name	Voous (1983)	Lee & Mackin (2008)	Boeken (2012)	Diamond Rock 2023	Green Island 2023	Red Cliff 2023
Sooty Tern	*Onychoprion fuscatus*	Breeding unsure	15-30	75-85	5	30-50	30-40
Bridled Tern	*Onychoprion anaethetus*	$\geq$25	52-70	<50	20-40	10-15	
Brown Noddy	*Anous stolidus*	Probably breeding	30-60	30-35	5-10	10-20	

For Saba breeding of the Sooty Tern seems to have increased over time. Compared to 2012, in 2023 Boeken counted fewer nesting birds on Green Island, which was found to be much more eroded than before. Part of this colony may have moved to Red Cliff; another possibility is that breeding on Green Island was early in 2023 year, and some birds moved away already.

The estimated numbers of nesting Bridled Terns and brown Noddies seems to have remained stable, but also for these two the breeding season was in its final stage. It is possible that more animals might have been counted if the (2023) surveys had been done at the end of May instead of only at the end of June-beginning of July.

Ecological Aspects

Habitat: Terns feed on fish they catch at sea. They rest and breed on bare patches of ground such as beaches, unpaved roads, and among rocks.

Food: All the described tern species feed on small fish and/or other small swimming prey found at the surface of the water in open seas and/or coastal areas of the island. Very little is known about the feeding ecology of these species in the Caribbean.

Disturbance: All terns are highly sensitive to disturbance (Debrot et al., 2009). They lay their eggs on the ground in bare beach habitats. As a result, breeding colonies are highly vulnerable to recreational disturbance from beachgoers and hikers, as well as predators, particularly free-ranging invasive species such feral cats, pigs and unleashed dogs. Videos, camera traps and monitoring by P. Bertuol has shown how a single cat can wipe out a colony of 31 Common Tern nesting pairs in a single night.

Minimum Viable Population Size The sizes of the MVPs (Minimum Viable Populations) for these eight breeding tern species are unknown. Based on the well-known population fluctuations of seabird populations, it can be stated that the IUCN rule of thumb is too low for terns. Therefore, the MVP is set at 5,000 birds per species. The island subpopulations of these far-flying species are likely to be considered a genetic unit regionally. This means that the MVP should be considered at a regional scale rather than an island scale. On a regional level, the population sizes for the Roseate Tern, the American Least Tern, and the Cayenne Tern are more or less at the required MVP level and may be considered as regionally Vulnerable according to the criteria of Schreiber (2000), even though the status of Roseate Tern has since been upgraded to Endangered (USFWS, 2010). For the Common Tern and the Royal Tern, the regional populations are far below the required MVP level and these species should respectively be considered as regionally, Critically Endangered and Endangered (Schreiber, 2000). For the Brown Noddy, Bridled Tern and Sooty Tern the sizes of the regional breeding populations are well above the 5,000 MVP level (Chardine et al., 2000a, b; Saliva, 2000) and the species may be considered as Least Concern at the regional level.

Present Distribution and Reference Values

The distribution of terns on Bonaire is as follows:

- **Roseate Tern:** The species nests annually with about 30 pairs breeding in the Cargill area of southern Bonaire. It has not been recently seen breeding in Slagbaai, Goto or elsewhere (P. Bertuol, pers. comm.).
- **Common Tern:** Based on monitoring (2014-2017) this species breeds with about 20 pairs annually in southern Bonaire and with 2-3 pairs in the Salina of Slagbaai. It may also be breeding in Goto.
- **Least Tern:** 750 pairs (around Klein Bonaire and the rest of the island's East Coast, the saltworks and Pekelmeer); yearly it breeds with around 300 nests in the Cargill saltworks and Pekelmeer in southern Bonaire (2014-2017, 2022 (but numbers may have recently declined (P. Bertuol, pers. comm.).
- **Royal Tern:** Between 400 and 650 nests annually in the Cargill saltworks and Pekelmeer in southern Bonaire (P. Bertuol, pers. comm.)
- **Cayenne Tern:** 3000-4000 pairs (mainly Pekelmeer).

For Saba the numbers of nesting birds are roughly as follows:

- **Sooty Tern:** 65-95 pairs (Saba: Diamond Rock, Green Island, Red Cliff). Numbers may have increased compared to the past.
- **Bridled Tern:** 30-55 pairs (Saba: Diamond Rock, Green Island)
- **Brown Noddy:** 15-30 pairs (Saba: Diamond Rock, Green Island)

No terns are known to nest on St. Eustatius.

Assessment of National Conservation State

Trends in the Caribbean Netherlands
Within the Caribbean Netherlands, breeding terns are only of regional or international significance on Bonaire. Due to the lack of structured monitoring and otherwise very scant data collection (but with the notable exception of Saba) this report provides very little in the way of a true update compared to the last SoN report of 2018 (Debrot and Bertuol, 2018). Some recent developments are suggested for

Bonaire (such as increases in breeding for the Royal Tern, declines for the Least Tern and some stabilization for the Cayenne Tern, but little can be said with much certainty as these are based on field experience and no data is available. For Saba our new data suggest some increase in the nesting population of the Sooty Tern while the number of nesting Brown Noddies and Bridled Terns has remained stable.

Recent Developments:
In Bonaire, egg poaching for human consumption used to be a problem long ago, but that is no longer the case. Today, the two main threats are human disturbance from recreation and predation, particularly by invasive predators such as free-roaming dogs, feral cats, and the expanding Laughing Gull, which benefits from the availability of unlimited food at the landfill (Debrot and Sybesma, 2000, Debrot et al., 2009).

Strolling through nature with (unleashed) dogs is a typical Dutch national pastime and the large influx of Dutch nationals to Bonaire over the last 10 years means that coastal areas of vital importance to nesting terns are experiencing rapidly growing disturbance (and likely nest mortality) by such recreation (Bertuol, pers. obs.). It will be especially critical to control the growing disturbance of nesting areas and colonies by this form of recreational disturbance.

On the positive side, Cargill has recently been experimenting with the creation of small artificial nesting islets within the flooded areas for salt production to limit or exclude invasive predators like cats and rats. This innovative approach to seabird management appears promising (Bertuol et al., 2015; Simal et al., 2022) but needs to be continued and perfected. Recent work by Simal et al (2022) shows much higher fledging success for terns nesting on islands rather than along shorelines where invasive predators (cats, rats) have easy access to nests.

Developments per species
Roseate Tern: In 2016, Stinapa counted 33 breeding pairs within the Cargill area in the south of Bonaire (Bertuol, in prep.). This is the first time since the 1960s that the species has been recorded breeding on Bonaire. Elsewhere, breeding islands are often subject to dynamic changes due to erosion and/or vegetation (particularly invasive plants) (USFWS, 2010). The species also shows a strong preference for nesting on small islets (USFWS, 2010). It is possible that the species was more numerous in the past but recent data remain lacking.

Common Tern: The number of breeding pairs of this species on the ABC islands (Debrot et al., 2009) indicates that, as previously noted by van Halewyn and Norton (1984), these islands have been and remain an essential breeding area within the Caribbean. On these islands, the species shows a strong preference for nesting on small islets (Debrot et al., 2009). The number of such islets on Bonaire is likely much fewer now than historically, particularly due to the levelling and flooding of large parts of southern Bonaire for salt production. It is suspected that the current breeding population of this species may be much lower than before the 1960s. In 2002, Debrot et al. (2009) counted 39 breeding pairs for Bonaire, 30 of which were within the Cargill area. In 2014 and 2015, the number in the same area was only 9 and 11, respectively. It is unclear if this represents a natural fluctuation or a possible declining trend.

Least Tern: For Bonaire, Debrot et al. (2009) estimated the breeding population of this species at about 790 pairs in 2002, widespread over 49 locations along the coasts of the island including Klein Bonaire. This accounts for more than 10% of the breeding population of the entire Caribbean (van Halewyn and Norton, 1984). This species is the least vulnerable to disturbance and very flexible regarding nest site choice and colony size. The current size of the nest population is likely indicative of the historical level. However, the species is very vulnerable to predation. Of the 29 colonies of this species monitored with cameras from 2013 to 2015, 20 had visits from predators. In the majority (13 cases), the predator was a cat, in 4 cases a Yellow-crowned Night Heron (*Nyctanassa violacea*), in two cases a Laughing Gull (*Leucophaeus atricilla*), and in one case an unidentified predator (Bertuol, in prep.). Casual observations (P. Bertuol, pers. comm.) suggest that numbers breeding in the Cargill saltworks of southern Bonaire may have recently declined. Work by Simal et al. (2022) demonstrate the great added-value of nest-island construction as an effective way to exclude predators and boost nesting success in this species.

Royal Tern: In 2002, there were 85 breeding pairs on the island, while in 2016 there were 209 breeding pairs in the Cargill area. In 2022, a total of 255 nests with eggs and or chicks were counted (P. Bertuol, pers. comm.) so it appears that breeding for this species on Bonaire has stably grown in recent years.

Cayenne Tern: This species still breeds on Bonaire, albeit in significantly reduced numbers. In 1969, 3,000-4,000 breeding pairs were recorded in the salt extraction area (Voous, 1983), while in 1982 only 600 pairs were found in the Goto Lake. In 1999, Adrian Del Nevo recorded 170 breeding pairs within the salt extraction area, while in 2002 Debrot et al. (2009) found only 150 pairs at Goto, but no nesting birds elsewhere on the island. There is enough data to conclude that the number of breeding pairs has drastically decreased over the past half-century. Since 2014, however, the species seems to be slowly increasing again. In 2014, 2015, and 2016, there were conservatively estimated 160, 540, and 750 breeding pairs in the Cargill area, respectively (Bertuol, pers. comm.). In 2022, a minimum of 467 nests were counted in the Cargill saltworks and Pekelmeer area of southern Bonaire (P. Bertuol, pers. comm.), so breeding by this species seems to have stablilized on Bonaire. Work by Simal et al. (2022) demonstrate the great added-value of nest-island construction as an effective way to exclude predators and boost nesting success in this species.

Sooty Tern: Breeds in low numbers on Saba. Seems to have increased in numbers breeding since 2008.

Bridled Tern: Breeds in low numbers on Saba. No demonstrable developments since 2008.

Brown Noddy: Breeds in low numbers on Saba. No demonstrable developments since 2008.

Assessment of Distribution: Favourable
The historically available breeding habitat is still available, with few exceptions. Most of it is part of internationally protected Ramsar sites, IBAs (Important Bird Areas), or island-level (planned but not legally) protected areas. The Conservation State and management challenges for all these areas are discussed by Geelhoed et al. (2013).

Assessment of Population: Unknown
Apart from the American Least Tern, which is still very numerous and possibly still present in historically high numbers, the current numbers of the other species are small and/or have decreased compared to a few decades ago. For the Cayenne Tern, it can be definitively shown that the number of breeding pairs for Bonaire has drastically decreased over the past 50 years. For the species highly dependent on islets (the Roseate Tern and the Common Tern), the currently low number of breeding pairs is likely partly due to the loss of small nesting islets due to the levelling and flooding of large areas for salt production in southern Bonaire. Very hopeful is that the salt production company (Cargill) has been constructing small nesting islets for terns since 2014/2015. This may partially compensate for past habitat loss. The number of breeding pairs of the Royal Tern has probably always been limited.

Assessment of Habitat: Unfavourable-inadequate
The available breeding area is currently sufficient for the existing and increasing number of breeding terns. Most breeding habitats have some degree of legal or planned recognition as important conservation areas and are often designated as island-protected areas or RAMSAR-recognized areas. However, actual management often falls short. As a result, disturbance remains a growing problem, some important areas are not well managed, and there is no control over free-roaming invasive predators, such as cats, which can cause enormous damage. In the past century, much natural habitat was lost or severely altered for salt production in southern Bonaire. Recently, this loss has been

partially mitigated by proactive interventions, such as artificial nesting islets and management from the relevant company, Cargill Salt Bonaire BV, in collaboration with STINAPA.

Assessment of Future Prospects: Unfavourable-inadequate
Bonaire experienced the greatest decline in breeding terns in the second half of the 20th century. Since then, there have been no further large-scale alarming developments. The increased awareness among key players (STINAPA and Cargill) means that while the situation cannot yet be called healthy, there is new future perspective. However, it will be especially critical to control the growing disturbance of nesting areas and colonies (due to growing recreational disturbance by the growing human population) and to control invasive predators. The threat of uncontrolled growth in recreational disturbance as signalled previously by Debrot and Sybesma (2000) and Debrot et al. (2009) and which has occurred on Bonaire during the last 10 years urgently needs to be harnessed (Bertuol, pers. obs).

Invasive species (particularly rats and cats) are the most important threat to most seabirds worldwide (Dias et al., 2019). For the neighbouring island of Curaçao, limited active intervention targeting the key invasive predator (cats) has been found to be very successful. After removal of all (but one) cats for the island of Klein Curaçao, within roughly 20 years the number of breeding seabirds increased from a single species to nine species and from a documented maximum of 140 pairs to over 650 pairs (Debrot et al., 2023).

Table 3. *Diagnostic scores for the four different State of Nature criteria for the five breeding terns of Bonaire as well as well as an overall conservation assessment for the year 2024.*

Aspect of terns Bonaire 2024	**Roseate Tern**	**Common Tern**	**American Least Tern**	**Royal Tern**	**Cayenne Tern**
Distribution	Favourable	Favourable	Favourable	Favourable	Favourable
Population size	Unfavourable-bad	Unfavourable-bad	Unknown	Favourable	Unfavourable-inadequate
Habitat	Unfavourable-bad	Unfavourable-bad	Unfavourable-inadequate	Favourable	Unfavourable-inadequate
Future prospects	Unfavourable-bad	Unfavourable-inadequate	Unfavourable-inadequate	Unfavourable-inadequate	Unfavourable-inadequate
Overall Assessment of Conservation State	**Unfavourable-bad**	**Unfavourable-bad**	**Unfavourable-inadequate**	**Unfavourable-inadequate**	**Unfavourable-inadequate**

Table 4. *Diagnostic scores for the four different State of Nature criteria for the five breeding terns of Saba as well as well as an overall conservation assessment for the year 2024.*

Aspect of terns Saba	**Brown Noddy**	**Bridled Tern**	**Sooty Tern**
Distribution	Favourable	Favourable	Favourable
Population size	Unfavourable-bad	Unfavourable-bad	Unfavourable-bad
Habitat	Unfavourable-bad	Unfavourable-bad	Unfavourable-bad
Future prospects	Unfavourable-inadequate	Unfavourable-inadequate	Unfavourable-inadequate
Overall Assessment of Conservation State	**Unfavourable-bad**	**Unfavourable-bad**	**Unfavourable-bad**

Comparison to the 2018 State of Nature Report

Overall, in comparison to the 2018 assessment, no major changes can be identified for the CS of the breeding terns of Bonaire. Also, as this is the first CS assessment made for the breeding terns of Saba and hence no comparison can be made to any earlier report.

Recommendations for National Conservation Objectives

Nature policy should focus primarily on the three tern species for which the ABC islands (including Bonaire) have historically been important breeding islands in the Caribbean region.

Key conservation goals:

- Increase/restoration of the current number of breeding pairs of the American Least Tern;
- Preservation of current number of breeding pairs of Roseate Tern on Bonaire;
- Increase/restoration of the number of breeding pairs of the Royal Tern;
- Increase the number of breeding pairs of the Common Tern.

Subgoals:

- Protection and preservation of the total available habitat;
- Habitat improvement through the construction of artificial nesting islands which have been shown to greatly boost nest success (Simal et al., 2022);
- Reduction of human disturbance (using a combination of awareness, signage, seasonal closures and the requirement to leash dogs when strolling through sensitive coastal areas).
- Control of invasive predators (Debrot et al., 2023)
- Support adherence to rules and regulations through public awareness campaigns about the impact of feral cats, pigs and unleashed dogs.

Key Threats and Management Implications

The significance of Bonaire as a breeding island for terns has drastically declined in the past (Debrot et al., 2009) but is now starting to show improvement. However, in the last five years practically no significant new information has become available and consequently this report provides little in terms of a real update compared to our last report (except for Saba). The current core threats are summarized in Table 5.

Table 5. *Listing of different threat categories to nesting terns, their predominant cause and the ensuing management implications.*

Key threat		Management implications
Habitat loss:	The filling in of saliñas with eroded sediment makes these shallower and reduces water coverage which increases vulnerability to disturbance and access by terrestrial invasive predators (cats, rats, pig, dogs).	• Combat erosion at the source (roaming livestock) • Restore water depth and coverage in critical saliña areas surrounding breeding sites • Land-use zoning to protect against deleterious land use
Disturbance:	Disturbance is extremely dangerous to nests during the breeding season	• Seasonal area closures, requirements for leashing dogs, control of invasive species • Laws and enforcement • Awareness
Nesting habitat degradation:	Different species of terns nest preferentially on small islands. Loss of islands due to erosion sedimentation, excess	• Construction of artificial nesting islands • Protection and maintenance of existing nesting islands.

	vegetation encroachment (such as in Lac Bay) and or salt industry modifications of water levels degrades or destroys critical tern nesting habitat.	• Preventing excess vegetation on nesting islands.
Predation:	Invasive terrestrial predators such as cats, rats and pigs can be disastrous to tern breeding success.	• Management of invasive species around key nesting areas. • Construction and restoration of nesting islands to reduce or eliminate access by invasive predators (see Simal et al., 2022).

Data Quality and Completeness

Current state of data collection: The current state of knowledge is sufficient to identify the main threats and formulate essential recommendations. However, it is inadequate for accurately tracking trends and developments or evaluating the effectiveness of management practices as is common in the Netherlands. NGO STINAPA, in collaboration with Cargill, collected valuable data from 2014-2016, providing new insights. However, this monitoring has not been continued. Only for southern Bonaire has additional data been collected in 2022. For the rest of Bonaire also no new data on tern breeding numbers has been collected since the last 2017 assessment (Debrot and Bertuol, 2018). For the island of Saba some updates in breeding tern data have fortunately been realised. For St. Eustatius this is not needed as terns are not known to breed there.

Challenges in Monitoring: Terns are known to regularly change their breeding locations. This implies that population monitoring focused solely on specific sites is insufficient to follow island-wide breeding trends. Comprehensive monitoring across the entire island and throughout the breeding season is necessary (as first conducted by Debrot et al., 2009) to provide quantitative insights. This approach, however, can be relatively expensive and labour-intensive. It is even likely that breeding birds may even move to different islands based on excessive disturbance or shortages in food supply. So monitoring on a more regional level makes the most sense.

Recommendations for Monitoring:

- Conduct island-wide surveys every five years. This would represent a significant advancement and should be sufficient to document long-term population trends and evaluate the effectiveness of management practices.
- Implement monitoring efforts that cover the entire breeding season and the whole island. This will account for the terns' tendency to shift their breeding sites and provide a more accurate picture of population dynamics.
- Ensure that monitoring is sustainable and continuous, potentially involving local stakeholders and volunteers to mitigate costs.
- Encourage data sharing and collaboration between NGOs, government agencies, and academic institutions to enhance the quality and completeness of the data collected.

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21 Conservation State of the Bats of the Caribbean Netherlands

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Status

Bats play key roles in terrestrial ecosystems, with different species likely helping to control insects (Petit 1996; Kunz et al., 2011; Riccucci & Lanza, 2014; Tuneu-Corral et al., 2023), other species serving as key pollinators and dispersers of plants like agave and columnar cacti (Petit, 1995; 1996; Kunz et al., 2011) and yet, other fruit-eating species playing an important role in seed dispersal for certain other plants (Barlow et al., 2000; Ortega & Castro-Arellano, 2001; Kunz et al., 2011; Pedersen et al., 2018a). On islands, they are often the main or only surviving native mammalian species and often fulfil keystone functions in terms of pollination and seed dispersal (Cox, 1992; Petit, 1995; Mickleburgh et al., 2002). Such is also the case for all three of the Caribbean Netherlands islands, which have no other surviving native mammals than bats.

The current and recent fossil bat fauna of the West Indies amounts to 57 species, of which 27 species are either locally or fully extinct (Morgan, 2001). Looking specifically at the Lesser Antillean fossil bats, these amount to one (fully) extinct species and three locally extinct species, which can fortunately still be found surviving in the Greater Antilles (Morgan, 2001). The assessment by Morgan (2001) further shows that specialized or obligate cave-dwelling species are the most vulnerable to extinction and the role of past climate change appears quite important. This suggests that the impending climate change caused by humans (IPCC, 2022) is also a big risk factor for the present surviving cave-dwelling species. Notwithstanding many documented extinctions, new bat species (Larsen et al., 2012, Moratelli et al., 2017) and subspecies (Larsen et al., 2017) continue to be described from the Caribbean.

The most recent addition to the known bat fauna of the Caribbean Netherlands is the Visored Bat, *Sphaeronycteris toxophyllum*, known from a single individual collected in Bonaire in 2023 (Simal & Nassar 2025). The species is very rare throughout its large range in South America (Solari, 2018a) and could either be vagrant or recently introduced via shipments of merchandize from Venezuela. Other likely vagrants for Bonaire might be *Ametrida centurio* and *Pteronotus davyi*, for which, respectively, a former or current local population seems very doubtful. As calm waters with small fish suitable for hunting by *N. leporinus* are expansive on the island, the apparent absence of this species, notwithstanding, ample observations by experts is puzzling (F. Simal, pers. obs.). On Curaçao the species is only known to roost in seaside caves and overhangs, a habitat which does not seem present on Bonaire and might explain the absence of the species.

Table 1. *Known Conservation State of bat species for Bonaire, St. Eustatius and Saba, Netherlands Caribbean. Our relative ranking of abundance categories is based on the metric "bat captures per net-night (BNN) as documented by Genoways et al. (2007) for Saba, Pedersen et al. (2018a) for St. Eustatius and Simal et al. (2021) and Simal and Nassar (2025) for Bonaire. R: range-restricted species; R* = range-restricted subspecies; W: widespread. A = abundant; C = common; U: uncommon; V = very uncommon. Many bats can eat a variety of foods but in our table only the principal diet preference is indicated unless other food types are also commonly taken.*

Latin name	common name	distribution	daytime roost	main food habits	spec or gen.	Bonaire	Statia	Saba	worldwide trend	IUCN
Ardops nichollsi montserratensis	Antillean Tree Bat	R	trees, higher island elevations	Frugivore	spec.	-	V	V	unknown	LC
Artibeus j. jamaicensis	Jamaican Fruit-eating Bat	W	hollow trees, caves	Frugivore	gen.	-	A	U	Stable	LC
Ametrida centurio	Little White-shouldered Bat	W	dense foliage	fruits, insects, nectar	gen.	V	-	-	unknown	LC
Brachyphylla c. cavernarum	Antillean Fruit-eating Bat	R	principally caves	omnivore	gen.	-	C	C	unknown	LC
Glossophaga longirostris elongata	Miller's Long-tongued Bat	W	light and dark caves, also buildings	nectarivore	spec.	C	-	-	unknown	LC
Leptonycteris curasoae	Curaçaoan Long-nosed Bat	W	principally dark caves but also man-made structures	nectarivore	spec.	C	-	-	decreasing	V
Molossus m. molossus	Pallas's Mastiff Bat	W	man-made structures	insectivore	gen.	-	C	C	unknown	LC
Molossus molossus pygmaeus	Pallas's Mastiff Bat	W	man-made structures	insectivore	gen.	V	-	-	unknown	LC
Monophyllus plethodon luciae	Insular Single-leaf Bat	R	caves, man-made structures, higher elevations	nectar, fruits, insects	?	-	V	V	unknown	LC
Mormoops megalophylla intermedia	Ghost-faced Bat	R*	warm caves	insectivore	spec.	C	-	-	decreasing	LC
Myotis nesopolus nesopolus	Curaçao Myotis	R	warm caves	insectivore	spec.	U	-	-	unknown	LC
Natalus t. tumidirostris	Trinidadian Funnel-eared Bat	R*	warm caves	insectivore	spec.	U	-	-	unknown	LC
Natalus s. stramineus	Mexican Funnel-eared Bat	R	warm caves	insectivore	spec.	-	-	U	unknown	LC

Latin name	common name	distribution	daytime roost	main food habits	spec or gen.	Bonaire	Statia	Saba	worldwide trend	IUCN
Noctilio leporinus	Greater Bulldog Bat	W	coastal caves	calm water piscivore	gen.	V	-	-	unknown	LC
Pteronotus davyi	Davy's Naked-backed Bat	W	warm caves	insectivore	spec.	V	-	-	stable	LC
Sphaeronycteris toxophyllum	Visored Bat	W	caves/trees	frugivore	spec.	V	-	-	unknown	LC
Tadarida brasiliensis antillularum	Mexican Free-tailed Bat	W	caves and man-made structures	insectivore	gen.	-	V	C	stable	LC
Confirmed species per island						10	6	7		
Common or abundant (C, A)						3	3	3		
Uncommon or very uncommon (U & V)						7	3	4		

A total of 23 island occurrences of 17 bat taxa have so far been documented for the Caribbean Netherlands (ten from Bonaire, six from St. Eustatius and seven from Saba; Table 1). Of these, only nine occurrences can be characterised as "common" or "abundant" with 60% of the occurrences concerning uncommon, very uncommon or likely vagrant species. The critical state of conservation of bats in the Caribbean Netherlands should be clear. Island endemic bats, like particularly the four range-restricted species listed for Saba and St. Eustatius, are significantly more threatened than bats not restricted to islands and have also typically been much less-well studied (Conenna et al., 2017). This is also the case for the bats of the Caribbean Netherlands, most of which have been poorly studied.

The principal components of an IUCN extinction risk assessment are decline, geographic range and population size (abundance) (Le Breton et al., 2019). The available data for none of the range-restricted species is anywhere suitable for detecting decline, nor is anything really known about population size. Nevertheless, comparing bat counts to rank species according to relative abundance is a critical criterium for conservation assessment, but it is extremely difficult, and the challenges and caveats have been well-highlighted by Pedersen et al. (2009). The most simple and pragmatic metric is "bat captures per net night" (BNN) (Pedersen et al., 2009). Based on BNN, we present a preliminary relative ranking of abundance for the Caribbean Netherlands (Table 1) based on data from Genoways et al. (2007) for Saba, Pedersen et al. (2018) for St. Eustatius and Simal et al. (2021) for Bonaire.

Relative BNN values we use for a ranking (as in Table 1), and as reported by Genoways et al. (2007) for Saba were as follows: *Tadarida brasiliensis* n = 24 (33%); *Brachyphylla cavernarum,* n = 17 (23%); *Molossus molossus*, n = 11 (15%); *Artibeus jamaicensis*, n = 9 (12%); *Natalus stramineus,* n = 9 (12%); *Monophyllus plethodon,* n = 2 (2%); *Ardops nichollsi*, n = 1 (1%). Relative BNN captures we use for a ranking, and as reported by Pedersen et al. (2018) for St. Eustatius were as follows: *A. jamaicensis*, n = 124 (72%); *B. cavernarum*, n = 22 (13%); *M. molossus*, n = 22 (13%); *A. nichollsi*, n = 3 (2%); *M. plethodon*, n = 1 (1%); *T. brasiliensis*, n = 0 (0%). Relative BNN captures we use for a ranking, and as reported by Simal et al. (2021) for Bonaire were as follows: *Leptonycteris curasoae*, n = 2379 (51%), *Glossophaga longirostris*, n = 995 (21%); *Mormoops megalophylla*, n = 754 (16%); *Myotis nesopolus*, n = 269 (6%), *Natalus tumidirostris*, n = 234 (5%); *Pteronotus davyi*, n = 1 (<<1%). Based on bat detector data, *M. molossus* seems more common in Saba than suggested by BNN, probably because it is a fast, open area hunting bat that has less chance of getting into a mist net than species that forage lower (Noort, pers. obs.). Methodological limitations must be kept in mind and improvements or changes in our assessments will certainly occur in time as long new, better, and more sophisticated research becomes available.

The Lesser Antilles are a disaster-prone region with various risks of volcanic eruption (even for Saba, Roobol and Smith, 2004) and hurricanes (Pedersen et al., 2009; Rodriguez-Duran, 2020), which have been shown to have heavy consequences for bat populations. The regional analysis by Pedersen et al. (2009) found that Saba and St. Eustatius have relatively low numbers and abundance of bat species considering their size (along with St. Maarten) and as compared to most other islands in the northern Lesser Antilles.

For all except one species, the current IUCN listing is as "Least Concern" (LC; Table 1). However, this is only based on information regarding their occurrence on multiple islands and the assumption that they are probably abundant on the islands on which they occur. On the other hand, our assessments indicate that for both Saba and St. Eustatius at least, the scant abundance of *A. nichollsi* and *M. plethodon* can provide very little assurance against extinction risk. In fact, *A. nichollsi*, *M. plethodon* and *N. stramineus* all appear rare or (in some cases) even totally absent on St. Maarten and St. Barthelemy (Larsen et al., 2006; Genoways et al., 2006) and Nevis (Pedersen et al., 2003), while *A. nichollsi* is slightly more abundant on St. Kitts, but *N. stramineus* is altogether absent (Pedersen et al., 2005). Their scant populations on the various surrounding islands provide little reassurance against local extinction risk for bats on Saba or St. Eustatius. Recent work shows that *A. nichollsi* and *M. plethodon* fortunately appear common on Martinique (Catzeflis et al., 2019) and St. Lucia (Pedersen et al., 2018b) so that as a species their global status may be less of a concern. *Leptonycteris curasoae* is the only species with a listing other than LC (see Table 1). It is listed as “Vulnerable” (VU) by IUCN-criteria based on decreasing populations, its breeding that is limited to a few major caves in South America (3 to 4 per ABC island), the dry cactus-dominated ecosystem it depends on in South America is in decline and, for the most part, outside protected areas (Nassar, 2015, Cole & Wilson, 2006).

Characteristics

Description

Overall, few studies have been done on the bats of these islands but several patterns can still be discerned. First, the bat fauna of Bonaire, leeward Dutch island, on the one hand, and Saba and St. Eustatius, windward Dutch islands on the other, show practically no overlap in species. Only one species (the generalist insectivore, *Molossus molossus*) overlaps at species level but with different subspecies in the two island groups. In total, the islands together possess eight range-restricted bat taxa. Bonaire has four range-restricted bat taxa, of which two subspecies and one species of insectivorous bats all depend on warm caves (Table 1). In contrast, Saba and St. Eustatius have four range-restricted bat taxa, all of which at the (higher) species level and of which three are principally frugivorous and one insectivorous. Their principal daytime roosts are in caves for three species and in trees for one species.

On Saba and St. Eustatius, current bat faunas are impoverished compared to surrounding islands and compared to former pristine conditions (Pedersen et al., 2009), and this has in part been ascribed to deforestation. On St. Eustatius, deforestation was for agricultural purposes during the colonial epoch, on Saba due to die-off of a key forest tree species (Freitas et al., 2019). On Bonaire, longstanding deforestation resulted from unsustainable wood harvest during colonial times (Freitas et al., 2005) and continuing chronic overgrazing that limits forest recovery and threatens keystone cacti landscapes just as in Curaçao (Petit, 2009). On Bonaire, the only documented species that has never again been collected and which may have been a rare vagrant is *Ametrida centurio*, a tree-roosting and principally fruit-eating species that may be especially sensitive to deforestation. The individual recorded of this species probably arrived from the mainland, while foraging and losing orientation, or alternatively was accidentally transported in a merchandise ship coming from Venezuela. The same may be true for the most recently documented Visored Bat, *S. toxophyllum*.

While the range-restricted taxa listed by Bos et al. (2018) are identical to those we list here for Saba and St. Eustatius, those listed for Bonaire here differ from those presented previously by Debrot (2006). We have delisted both *Leptonycteris curasoae* and *Glossophaga longirostris* as “range-restricted” subspecies for the Leeward Dutch islands, based on the wide range of documented distribution for both species, genetic assessments (Newton et al., 2003), and the documented connectivity between the islands and the mainland of Venezuela for *Leptonycteris* (Simal et al., 2015).

We here have further added two bat subspecies and one species to the list of range-restricted taxa for Bonaire based on the limited range of the subspecies in question. The first concerns *Mormoops m. intermedia,* an otherwise wide-ranging species of which the subspecies *intermedia* is limited to a very small area in the Southern Caribbean (Rezsutek and Cameron, 1993). Secondly, we also list *Natalus t. tumidirostris* as a range-restricted bat subspecies for Bonaire. *Natalus tumidirostris* is the most geographically variable of the four continental *Natalus* species and is found in Colombia, Venezuela, the Leeward Dutch islands, Trinidad and all the way down to French Guyana (Tejedor, 2011). However, the Curaçao and Bonaire *N. tumidirostris* population, referred to traditionally as *Natalus t. tumidirostris*, has often been considered a subspecies endemic to Curaçao and Bonaire (Genoways and Williams, 1979). New analyses based on a multivariate assessment of measurements for four subpopulations showed no overlap between subpopulations and also that the bats from Curaçao, Bonaire (and a limited area of Colombia and Venezuela) are among the smallest of the species which justifies their subspecies status. Finally, the third concerns *Myotis nesopolus*, currently limited to the Leeward Dutch islands and a small portion of Venezuela (Solari, 2016). According to earlier as well as more recent work using a multivariate analysis, *M. n. nesopolus* from Bonaire and Curaçao differ from *M. n. larensis* from coastal Venezuela more than expected of subspecies in mammals (Genoways and Williams, 1979, Larsen et al., 2012).

In conclusion:

- On all three islands, very few species can be considered common or abundant. Most species are uncommon or very uncommon, and their future status on these islands can be considered quite uncertain.
- The islands of Saba and St. Eustatius have a lower-than-expected number of bat species than on average to be expected in the Antilles based on island size.
- The principal cause for low bat diversity is believed to be deforestation.
- There are large differences in bat faunas between leeward Bonaire and the two windward islands of Saba and St. Eustatius.
- Saba and St. Eustatius have more-highly range-restricted ("endemic") bat taxa than Bonaire, because of greater isolation from mainland sources of bat diversity.

Relative Importance Within the Caribbean

Bats form an ecologically important component of the biodiversity of the Lesser Antilles which amounts to 27 species (Pedersen et al., 2013). This is low compared to the number of bat species on the larger islands like Cuba and Hispaniola or on the mainland parts of the Americas. Even so, the Lesser Antillean islands have 11 endemic species typically found on just few adjacent islands and contribute meaningfully to the unique biodiversity of the region. The Caribbean Netherlands form part of the Caribbean biodiversity hotspot (Myers et al., 2000; Mittermeiers et al., 1999) and have a total of sixteen bat taxa, eight of which are range-restricted species or subspecies and eight other, more widely distributed bat taxa.

The keystone roles fulfilled by bats as pollinators, seed dispersers and insectivores have high ecosystem importance to the maintenance of terrestrial biodiversity on these islands, including many other endemic taxa. An example from Bonaire is the role nectar- and fruit-feeding bats play as key pollinators and seed dispersers of candelabra cacti (Petit, 1995), which in turn are a principal dry-season food source for the endangered parrot, *Amazona barbadensis*, and the following six endemic subspecies of birds: parakeet, *Aratinga pertinax xanthogenius*; Tropical Mockingbird, *Mimus gilvus rostratus*; Pearly-eyed Thrasher, *Margarops fuscatus bonairensis*; Bananaquit, *Coereba flaveola bonairensis*; Black-faced Grassquit, *Tiaris bicolor sharpei*; and Yellow Oriole, *Icterus nigrogularis curacoensis*.

Ecological Aspects

Habitat:
With the likely small population sizes of bats found particularly on Saba and St. Eustatius, bat species survival in these and surrounding islands depends on the ability of bats to move between islands. Yet practically nothing is known about this (Pedersen et al., 2009). Their essential habitat for long-term survival thus extends beyond any given island.

The key importance of caves for the range-restricted species of the bats of the Caribbean Netherlands is clear and caves have been indicated as a key conservation priority for bats in the islands (Genoways et al., 2007; Pedersen et al., 2013, 2018a; Simal et al., 2021). While for Bonaire the presence of bats in caves is well-established (Simal et al., 2021), for Saba and St. Eustatius, the key bat caves appear to be much more elusive (Genoways et al., 2007; Pedersen et al. 2018a). Finding the key bat caves or other roosting locations on Saba and St. Eustatius is of crucial importance, but in 2022 and 2023 Boeken and Noort made a larger effort to find, map and survey key caves, crevices and overhangs of Saba (in addition to the Sulphur mine, 19 of which two were 10 meters or deeper) and St. Eustatius (two, including the location listed by Pedersen 2018a for Venus Bay). However, signs of a few bats were only found at a single "cave" shelter (the Sulphur Mine) and one man-made shelter location (ceiling) on Saba while scientists from the Royal Netherlands Meteorological Institute incidentally encountered a few bats roosting in a shallow coastal cave near Ladder Bay. Figure 1 illustrates the results of the efforts to document important bat roosting sites for Saba. The question of where the bats of Saba and St. Eustatius are roosting and why they do not appear to use natural rock shelters on these islands is key to their protection and requires further investigation. The possible low availability of suitable caves or their low use due to preference for more ephemeral manmade structures may explain the (at least temporary) extirpation of certain bats species from these islands due to hurricane impact (Rodriguez-Duran et al., 2020).

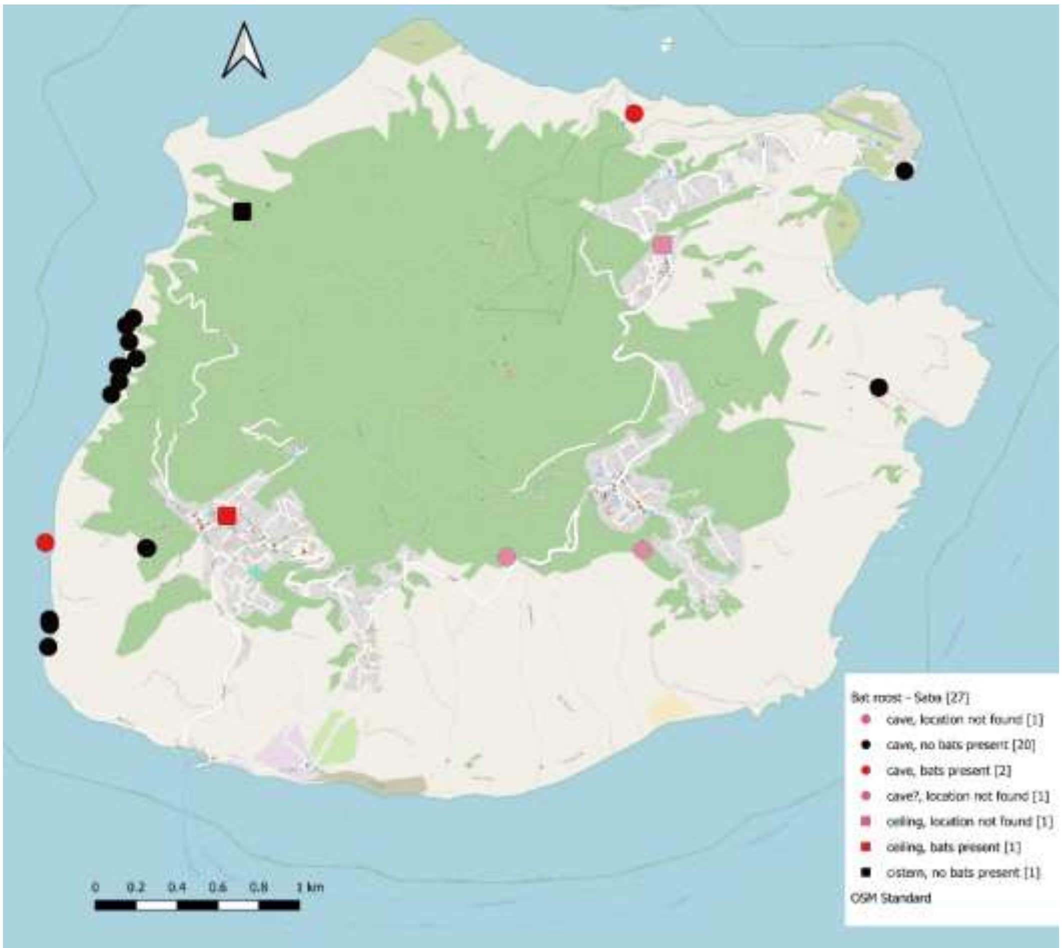

Figure 1. *Bat* roosting *site survey results for Saba in 2023, based on 27 site visits to high-potential bat roosting sites. Round = natural cave, Square = artificial/man-made structure. Black: searched potential sites with no bats detected, red: sites with a few or dozens of bats found roosting, pink: sites reported to have likely bat roosts but which were unable to find.*

Minimum viable population size: not reached for any species
A minimum viable population (MVP) means a 5% extinction risk within 100 years. MVPs for bats are unknown; however, based on the surveys that have been conducted, it is safe to say that, with exception of *L. curasoae*, which moves among the ABC islands and mainland and probably migrates seasonally to southern Venezuela and eastern Colombia (Simal et al., 2015), none of the islands of the Caribbean Netherlands have populations that even approach the 5,000 population size criterium that is typically considered a minimum for long-term survival. Therefore, survival of these species on the islands will depend critically on animals immigrating from beyond the islands. Unfortunately, for very few species is anything known about movements between neighbouring islands (e.g. Simal et al., 2015; Pedersen et al., 2009) nor about the population sizes inhabiting nearby islands. In fact, for three of the four Antillean

endemic bat species of Saba and St. Eustatius *(A. nichollsi*, *M. plethodon* and *N. stramineus)*, their status on neighbouring islands appears equally tenuous (Genoways et al., 2006; Larsen et al., 2006; Pedersen et al., 2003, 2005).

Present Distribution and Reference Values

Little can really be said with great certainty about the habitat distribution or habitat preferences of bats on the islands of the Caribbean Netherlands. Nevertheless, a few key points can be made based on the available mist-netting results and additional observations by experts.
Saba (based on Genoways et al., 2007): *T. brasiliensis* has so far only been documented from a single location in English Quarter. All *B. cavernarum* were documented from Island Gut. *Artibeus jamaicensis* was found in low numbers in the Sulphur Mine. *N. stramineus* was documented from one location at Mary's Point, but a cave could not be located. A roost of *M. molossus* was found in one building on Saba, but more are expected. Genoways et al. (2007) searched three caves on Saba without finding any bats. These were Deep Cave on Great Hill, a small overhang cave south of The Bottom and the seaside cave at Well's Bay which was reported at one time to have harboured bats. The historic collection site of Bat Hole, a small cave at Ladderberg near land Point is worth examining again as a potential daytime roost for *B. cavernarum*, if it still exists.

St. Eustatius (based on Pedersen et al., 2018a): *B. cavernarum* were seen in large numbers in a cave at Venus Bay. *A. nicholsi* and *M. plethodon* were only documented from the higher regions of the Quill. *A. nicholsi* apparently prefers the higher parts of islands (Catzeflis et al., 2019). Even though *M. plethodon* is also found in low numbers on more xeric islands, it may be dependent on the Quill in Statia because of the major impact of man on the vegetation (Pedersen et al., 2018a). According to (Catzeflis et al. (2019) *M. plethodon* is also clearly a species of higher elevations.

Bonaire (based on Simal et al., 2021): While in contrast to Saba and St. Eustatius, on Bonaire the five principal bat caves studied until now are fairly known, still none of them are inside legally protected and managed nature parks. However, this island is rich in caves and karst, deserving further exploration for possible additional caves used by bats as day, mating and maternity roosts. The real distribution of bats across the island while foraging is less well-known, because all mist-netting has focussed on the larger bat caves. For this reason, recent results may also not give proper representation of species that roost in other structures (such as buildings, cisterns, scattered crevices and trees).

Assessment of National Conservation State

Trends in the Caribbean Netherlands
Aside from a few recent baseline studies that provide at least some point of reference, nothing is known about population trends for bats in the Caribbean Netherlands. However, on Bonaire, after erecting physical barriers to keep humans out of bat maternity chambers at the five most important bat maternity roosts in 2020, ongoing research of these bat colonies has preliminarily shown an increase in the size of the colonies after three years (2021-2023) of data collection (F. Simal and J. Nassar, unpublished data). This ongoing research is a 6-year quantitative study, which collects data simultaneously at these five caves using infrared light to take images of the bat colonies during the day at two of these caves and filming their evening exits at the other three. The aim of the study is to assess the long-term effect of the barriers on the colonies.

Reference values for population size and distribution: limited
Very few quantitative assessments have been done for the Caribbean Netherlands and assessments are by nature plagued by limited site-selection and technical difficulties as accurately counting bats is extremely difficult.

Recent developments: none
No major developments aside from the several recent studies cited herein need to be mentioned. However, during 2019 and 2020, a Bonaire Caves and Karst Nature Reserve was inaugurated to protect all the known Middle terrace bat maternity roosts by gating the cave entrances. Unfortunately, since then the project has stagnated due to lack of local support and a legal status remains wanting.

Assessment of distribution: unfavourable-bad
While on Bonaire the five main roosting and pupping caves are well known, on Saba and St. Eustatius most daytime roosting locations remain unknown while those that are known appear small. One notable exception is a coastal cave at Venus Bay St. Eustatius, that has been found to (at least once) harbor about 250 *B. cavernarum* (Pedersen et al., 2018). Due to the critical nature of roosting/pupping caves, and notwithstanding the island-wide foraging distribution of bats in general, we apply a more restrictive concept of "distribution" than would, for instance, be applied to birds (which also fly all around an island but which are much less restricted to a highly specialized habitat feature like caves).

Assessment of population: unfavourable-bad
With exception of *L. curasoae*, and probably *G. longirostris*, there can be little doubt that none of the other bat species on any of the three islands meet the 5000 minimum viable population (MVP) size for long-term survival. Long-term survival of these species will critically depend on the status of these species on surrounding islands (in the case of Bonaire also in Venezuela) and the movement of bats between the various islands. Fifteen of the 22 documented species occurrences regard species to be characterized as either uncommon, very uncommon or already extirpated (Table 1). Most species are very vulnerable to local extirpation and clearly highly dependent on populations of adjacent islands.

Assessment of habitat: favourable
In general, forest habitat quality for bats is certainly to have been degraded compared to early colonial conditions but, on the other hand, probably improved in recent decades due to reduced agricultural activity (de Freitas et al., 2005; 2014; 2016). The only major persistent negative pressure is uncontrolled and excessive grazing by roaming livestock (e.g., Debrot et al., 2015; Lagerveld et al., 2015; Madden, 2020), which does not allow the much-needed forest recovery to take place. Bat caves of Bonaire presently seem to be of adequate quality while the bat roosting habitats of bats of St. Eustatius and Saba are largely unknown, destroyed (Bat Hole, Saba) or recently degraded (Sulphur Mine, Saba). Disturbance, even based on well-meaning interest, can cause serious disturbance to roosting or nursing bat colonies and is a danger that needs to be controlled by limiting human access.

Assessment of future prospects: unfavourable-bad in the long-term
Given the apparent intractability of the roaming livestock problem, the inexorable long-term climate change impacts, the already seemingly small and vulnerable island population sizes of most bats on Saba and St. Eustatius and the growing risk of disturbance of key daytime roosting habitat, long-term prospects for bats on the islands of the Caribbean Netherlands seem quite uncertain.

Table 2. *Summary overview of the status of the bats of the Caribbean Netherlands in terms of different conservations aspects.*

Aspect (for the many rare and range-restricted species)	**2024**
Distribution	Unfavourable-bad
Population	Unfavourable-bad
Habitat	Favourable
Future prospects	Unfavourable-bad
Overall Assessment of Conservation State	**Unfavourable-bad**

Comparison to the 2018 State of Nature Report

This is the first CS assessment made for the bats of the Caribbean Netherlands and hence no comparison can be made to any earlier report.

Recommendations for National Conservation Objectives

Setting priorities in conservation is essential as the number of species requiring action is large and because the costs for the necessary interventions on a species-basis typically exceed the available resources (Possingham et al., 2002). Careful choices need to be made on what to do and where to spend resources. Ideally then, an ecosystem-based, holistic approach to conservation with the participation of local communities is needed whereby the focus should be on conserving and restoring systems of benefit to the conservation of various species at the same time. Due to the poor level of knowledge available for bats and the high costs of detailed research, management interventions to guarantee the future presence of this keystone group of important and yet vulnerable animals should focus on holistic ecosystem measures that simultaneously benefit multiple species, systems and the human communities that share the habitat with the bat fauna. As pointed out by Pedersen et al. (2013) and Simal et al. (2021), the conservation actions most needed for protection of bats are:

a. Find the key daytime shelter habitats for the bats of Saba and St. Eustatius, whether natural or manmade (as these remain largely unknown).
b. Protect caves and other shelter locations, such as abandoned cisterns or rock overhangs that are known to serve as bat shelters (e.g., implementation of the Bonaire Caves and Karst Nature Reserve). Protect roosts in houses and/or create special artificial shelters.
c. Improve forestation and forest diversity for a more ample, and stable supply of fruit and insect food.
d. Restore and protect hydrological systems, such as springs, ponds, and natural freshwater sources that bats will eagerly make use of.
e. Increase local awareness and create direct and indirect connections between local communities, bats and their roosts, through bat-related cultural and ecotourism activities as a source of profit. Improve knowledge on healthcare in combination with mosquitos and diseases brought by mosquitos.
f. For Saba, a longstanding bat habitat is the so-called sulphur mine. In recent years, several entrances to the mine have become obstructed, probably leading to unfavourable conditions inside the old mine shafts resulting in reduced bat usage of the mine. Restoring the openings could help restore the apparently degraded habitat quality in the mines.

Key Threats and Management Implications

The major threats to bats are principally fourfold:

a. The first and most immediate is further degradation and or lack of recovery of forest habitat. Grazing pressure by goats causes aridification, floral impoverishment and lack of forest recovery. This plays on all three islands.
b. The second is the pressure of increasing human disturbance or destruction of caves and other daytime roosting habitat. For Saba and St. Eustatius, with only two exceptions, roosting caves/mines are all but unknown. On the other hand, for Bonaire the main roosting caves are largely known. Although they are not inside managed or protected parklands, a conservation intervention led by WILDCONSCIENCE and IVIC (Venezuela) since 2019 to exclude humans from the main cave chambers used by bats, is gradually showing positive effects on the protection of cave-dwelling bats on this island (F. Simal and J.M. Nassar, unpublished data).

c. The assessment by Morgan (2001) shows that specialized or obligate cave-dwelling species are the most vulnerable to extinction and the role of past climate change appears to have been quite important. The threat of climate change whereby the expected warming and drying trend in the Caribbean will reduce and ultimately eliminate the rainforest and remnant elfin woodlands on the highest zones of these islands will endanger the food supply and daytime roost habitat for those species which roost in trees. Climate change will also affect temperatures within cave systems used by bats, and other vulnerable fauna (Mammola et al., 2019; Medina et al., 2023), possibly causing bats to lose certain roosting habitat. An important negative link between bat diversity and temperature in the Caribbean has been further suggested by Hoffman et al. (2019).
d. For the bats of Saba and St. Eustatius, hurricanes can cause heavy mortality rates, especially for those species or populations that shelter in trees or buildings as opposed to caves (Gannon and Willig, 2009).

Data Quality and Completeness

In spite of (and possibly because of) the fact that many bat species are endangered and have small population size and/or limited distribution, very little is known about the species-specific ecology of most Antillean species. This is a key knowledge gap that may seriously hamper their conservation, not only in the Caribbean Netherlands but in the Caribbean as a whole. As pointed out by Pedersen et al. (2009), quantitative insights into bat populations (population size) are extremely difficult, not only because many roosts remain unknown, but because bats will switch between roosts depending on their specific needs and because counting all bats in complex roost sites is almost impossible. Documenting population trends is therefore also very problematic. Both are key criteria for determining IUCN population Conservation State of a species (Le Breton et al., 2019). In addition, practically nothing is known about the ecology of three of the island endemic species of Saba and St. Eustatius (*A. nichollsi*, *M. plethodon*, *N. stramineus*) (Davalos & Rodriguez Duran, 2019; Davalos & Tejedor, 2016; Rodriguez Duran and Davalos, 2018). More is fortunately known about the fourth more-widely-spread island endemic *B. cavernarum* (Rodriguez Duran and Davalos, 2019). Further understanding of the ecology of all, but especially the range-restricted bats species (which represent the unique local contribution to biodiversity), is of great value.

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22 Conservation State of the Sea Turtles of the Caribbean Netherlands

Dogruer, G., Schut, K., Butler, E. Smulders, F.O.H, and Becking, L. 2024. From: State of Nature Report for the Caribbean Netherlands, 2024. WUR report C001/25.

Status

Sea turtles, resilient inhabitants of the oceans for 100 million years, embody both ecological significance and cultural value. With their highly migratory nature, these marine reptiles travel vast distances and utilize diverse habitats. Sea turtles play a pivotal role in maintaining the health of aquatic ecosystems, contributing significantly to preserving coral reefs, seagrass beds, and sandy beaches (K. A. Bjorndal & Jackson, 2002; Christianen et al., 2012; Goatley et al., 2012). Furthermore, sea turtle species are true ecosystem engineers, fulfilling multiple roles as consumers, prey, and competitors, hosts for parasites and pathogens, substrates for epibionts, nutrient transporters, and landscape modifiers (K. A. Bjorndal & Jackson, 2002; Lal et al., 2010). For instance, the green sea turtle, a keystone species for seagrass ecosystems, notably enhances sediment carbon and seagrass nutrient content (Christianen et al., 2023). These charismatic creatures also hold significant economic value, with global tourism-based revenue exceeding billions of dollars annually and ~50 million USD in the United States alone (McCrink-Goode, 2014)

The status and the need for conservation efforts to support the population recovery of these key species have sparked widespread interest from government agencies, non-governmental organisations (NGOs), and the public globally and locally in the BES islands (Bonaire, St. Eustatius, and Saba). However, the need for more data on turtles, human-turtle interactions, population status, threats, and the effectiveness of conservation measures often poses challenges to management actions. This report summarises the status of sea turtles from the BES islands and provides an overview of the threats, knowledge gaps, and recommendations for conservation and management actions.

The Netherlands follows a clear set of international and regional agreements that guide how it protects sea turtles and their habitats in the Dutch Caribbean. At the global level, the United Nations Convention on the Law of the Sea (UNCLOS) and its Straddling Stocks Agreement set out who controls different parts of the ocean, require countries to manage shared marine life carefully and forbid actions that harm the environment in territorial seas and Exclusive Economic Zones (EEZs), although sea turtles still receive little protection once they travel into the high seas. Building on these, three other global treaties each add a different layer of protection: the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) strictly controls all international trade in sea turtles; the Convention on Biological Diversity (CBD) forces countries to make national plans for protecting biodiversity; and the Convention on the Conservation of Migratory Species of Wild Animals (CMS) requires "range states" to work together to protect migratory animals like sea turtles.

Regionally, the Inter-American Convention for the Protection and Conservation of Sea Turtles (IAC) creates binding rules on catching, trading, and bycatch (including mandatory turtle excluder devices). The SPAW Protocol to the Cartagena Convention (SPAW Protocol) bans taking, trading, or disturbing turtles and calls for marine protected areas; the Ramsar Convention on Wetlands of International Importance (Ramsar Convention) safeguards key coastal wetlands; and the International Maritime Organization's Particularly Sensitive Sea Areas (PSSAs) designation protects areas like Saba Bank from harmful shipping activities. Together, these agreements form a strong network of research, monitoring, trade controls, and habitat safeguards, yet enforcing rules on the high seas and ensuring all Dutch and Caribbean territories apply them consistently remains a challenge.

Finally, several bodies and treaty annexes rank sea turtles by how much danger they face and trigger extra protections (Table 1). The International Union for Conservation of Nature (IUCN) lists species as Least Concern (LC), Vulnerable (V), Endangered (E), or Critically Endangered (CE) based on extinction risk. Under the SPAW Protocol, species in Annex II are treated as endangered or vulnerable and must receive strict protection. CMS Annex I covers species already in danger of extinction, while CMS Annex II flags those needing better conservation cooperation. CITES Appendix I bans most international trade in species at highest risk. These categories help focus conservation efforts where they're needed most. Overall, the alignment of these international frameworks highlights that none of the sea turtle species are currently in good conservation status, and all require strong legal protection and sustained conservation efforts. Therefore, it is highly warranted to continue and extend monitoring populations and actively manage human-induced stressors such as habitat loss, pollution, bycatch, and climate change to prevent further declines and promote long-term recovery.

Table 1. *Conservation state and legal protections for sea turtle species: IUCN: International Union for Conservation of Nature, LC = Least Concern, EN = Endangered, VU = Vulnerable, CE = Critically Endangered, SPAW Annex: Specially Protected Areas and Wildlife Protocol, CMS Annex: Convention on the Conservation of Migratory Species of Wild Animals, CITES Appendix: Convention on International Trade in Endangered Species of Wild Fauna and Flora.*

Name (Latin)	IUCN Status	SPAW (Caribbean)	CMS (Migratory) Annex	CITES (Trade) Appendix
Caretta caretta	VU	II	I, II	I
Chelonia mydas	EN	II	I, II	I
Dermochelys coriacea	VU	II	I, II	I
Eretmochelys imbricata	CE	II	I, II	I
Lepidochelys olivacea	VU	II	I, II	I

Characteristics and present distribution

Sea turtles are divided into two main subgroups with a distinct family: Dermochelyidae, which includes only one species, the leatherback (Figure 1), and the family Cheloniidae, which encompasses the six hard-shelled sea turtles. Many sea turtle species undergo dietary changes during different life stages. After hatching, sea turtles typically enter a cryptic life stage, during which little information about their early years is known. Promising new acoustic and satellite transmitters could help provide information on the early life stages of sea turtles, such as the finding that different species of juvenile turtles were actively swimming and not drifting only with the currents (Phillips et al., 2025). Many of these animals settle in shallow, often coastal waters to forage in their later juvenile and subadult stages. Sea turtles show high site fidelity to their foraging and, as adults, to their nesting grounds, which brings them close activities (Shimada et al., 2020). Sea turtles typically reach sexual maturity at older ages; green turtles, for example, reach sexual maturity between 25 and 35 years. Once sexually mature, they often migrate back to their nesting beaches to mate and reproduce (Limpus, 2008; Limpus et al., 1984; Limpus & Fien, 2009). Overall, they rely on the marine and terrestrial environments of the coastal zone for their survival.

On a global level, the loggerhead turtle (*Caretta caretta*) population consists of 10 subpopulations. Post-hatchlings transition to a pelagic stage, exhibiting low-energy swimming and feeding on floating material, especially Sargassum (Witherington et al., 2012). Juvenile loggerheads move between oceanic and neritic zones for several years before adulthood (Ramirez et al., 2015). Adult females exhibit reproductive longevity, with some nesting up to 25 years. Throughout their lifecycle, loggerheads primarily consume carnivorous diets with regional and ontogenetic variations (Bjorndal, 2017). For the loggerhead turtle, mortality due to bycatch has been identified as the most severe threat globally, followed by coastal development and hunting for meat and eggs (Casale & Tucker, 2017)

The green turtles (*Chelonia mydas*) in the Caribbean grow more slowly than hawksbills or loggerheads of similar size but faster than green turtles in the Pacific. Growth functions also differ between ocean basins, highlighting the need for region-specific management(Bjorndal et al., 2000; Bjorndal & Bolten, 1988). Young green turtles undergo an omnivorous pelagic phase for several years before potentially drifting with ocean currents and settling in neritic environments where they primarily feed a herbivorous. Thus, green turtles heavily rely on seagrass fields. However, seagrasses rapidly decline through human-induced stressors such as nutrient and chemical pollution runoff and coastal development (Dunic et al., 2021), causing green turtles to aggregate in shrinking foraging habitats (Gangal et al., 2021). In addition, the exotic seagrass species *Halophila stipulacea* is rapidly expanding in the Caribbean Sea (Winters et al., 2020). Selective feeding by green turtles on native seagrass facilitated this invasive seagrass on Bonaire (Christianen et al. 2019). However, evidence from Bonaire(Becking et al., 2014) and other Caribbean sites shows that turtles opportunistically feed on this exotic species (Siegwalt et al., 2022), possibly providing an opportunity for meeting the nutritional demands of green turtle populations. An important question remains if subtropical seagrass meadows have the carrying capacity to sustain increasing green sea turtle populations that are migrating northwards because of rising seawater temperatures (Campbell et al., 2024; Rodriguez & Heck, 2021). Care should be taken to regulate turtle-tourist interactions, as swimming and feeding turtles (which occurs on e.g. Curaçao) can result in unnatural aggressive behavior, and alternative food sources may pose a risk to turtle health(Smulders et al., 2021). Green turtles are long-lived and exhibit strong fidelity to relatively narrow foraging grounds, spending more than a decade in these areas before reaching sexual maturity(Limpus et al., 1992; Shimada et al., 2020). Because they remain in the same coastal habitats for so long, they steadily accumulate contaminants from runoff and other local sources. This makes them ideal bioindicators: by measuring pollutant loads in green turtles, researchers can assess the sublethal effects of poor water quality effects that often manifest as increased susceptibility to disease and other stressors rather than immediate mortality (Dogruer, 2022; Dogruer et al., 2021; Gallen et al., 2019; Gaus et al., 2019; Weltmeyer et al., 2021).

In the Northwest Atlantic, subpopulations of Leatherback have decreased since 1990, and declines are particularly severe in French Guiana (Eckert and Hart, 2021; Eckert and Eckert, 2019). Threats for nesting females include habitat loss and sargassum influx, and at sea threats include net fisheries, pollution, and entanglement (K. Eckert & Hart, 2021; Saba et al., 2008; Tröeng et al., 2004) . Key nesting beaches in the Caribbean include Grand Riviere and Fishing Pond in Trinidad, Armila in Panama, and the Gulf of Urabá, Colombia. This species primarily inhabits aquatic environments, demonstrating deep-diving behaviours and feeding on pelagic jellyfish and related mollusks. They visit the warm tropical waters of the Caribbean solely for nesting purposes.

Hawksbill turtles (*Eretmochelys imbricata*) exhibit genetic diversity among nesting populations, necessitating separate management units. Global declines of 84-87% in animal numbers over the last three turtle generations have occurred due to overexploitation of nesting females, egg collection, capture on foraging grounds, loss and degradation of nesting beaches, and bycatch in fisheries (Mortimer & Donnelly, 2008). Despite reduced trade in tortoise shells, which remains a severe threat, some protected populations have increased. Hawksbill turtles mature after 20 years or more, primarily feeding on sponges in the Caribbean. Within the Western Atlantic, hawksbill turtles migrate throughout the wider Caribbean Basin, and there is a need to protect the important corridors linking their high-use areas (Maurer & Eckert, 2024). Enhanced protection measures on nesting beaches and reduced exploitation in nearby foraging areas, especially in Cuba, have contributed to significant increases in the Caribbean (Campbell, 2014).Hawksbill turtles primarily inhabit coral reefs, feeding predominantly on sponges. Despite coral reef decline, there appears to be an adequate supply of sponges on relatively healthy reefs in Bonaire and surrounding areas like St. Eustatius and Saba. Therefore, food availability is not a limiting factor for this species in these regions (Debrot et al., 2014).

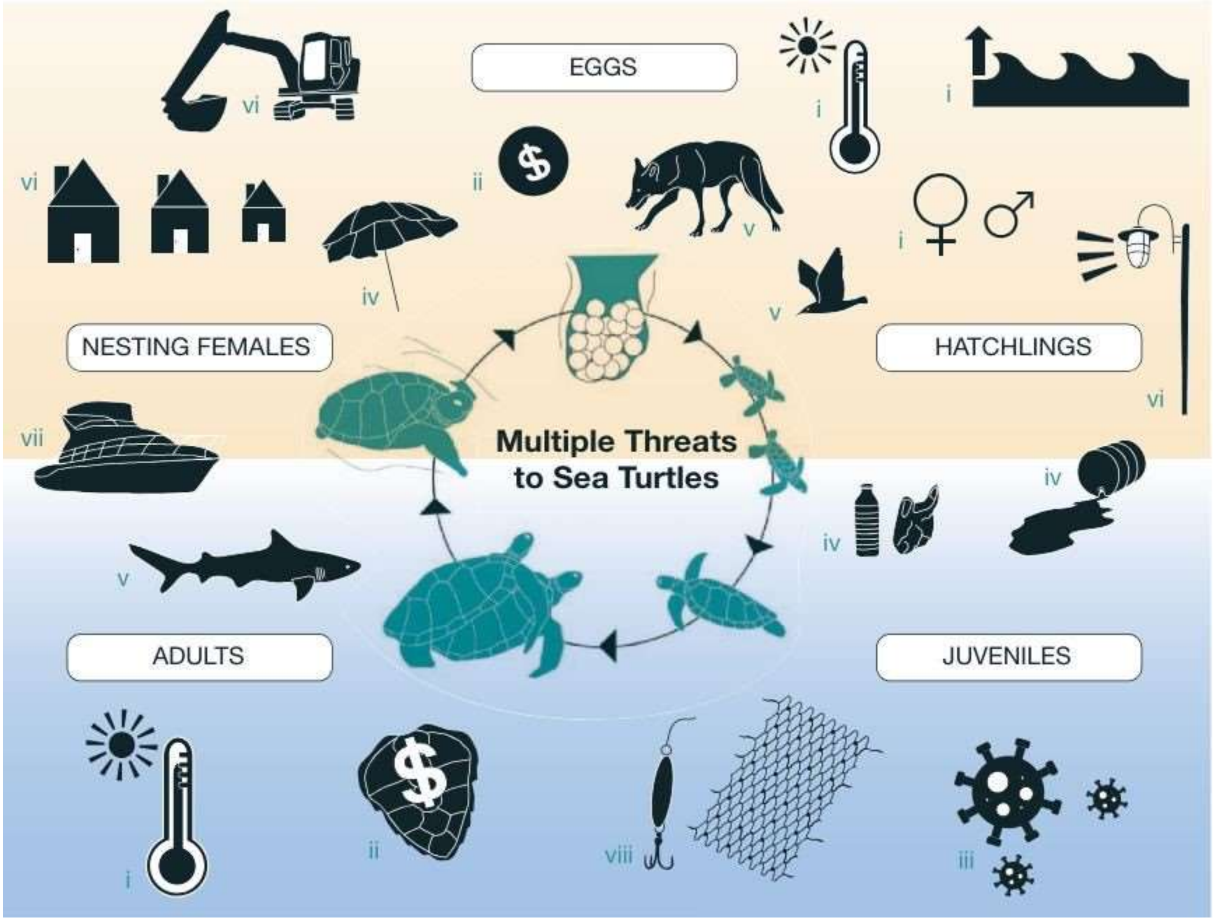

Figure 2. *Sea turtles face cumulative and synergistic threats across their life stages and habitats. They face threats on land and in the ocean, which may create conservation challenges. Depicted threats: (i) climate change; (ii) direct take; (iii) disease; (iv) pollution; (v) predation; (vi) coastal development;(vii) marine development; (viii) fisheries (ix). From Fuentes et al., 2023).*

Olive ridleys (*Lepidochelys olivacea*) face population declines due to slow growth and human impact across different life stages and habitats, including nesting beaches, migration routes, and foraging areas spanning a wide geographic range (Abreu-Grobois & Plotkin, 2008) . Coastal development poses a significant threat to nesting beaches, such as Eilanti in Suriname, which experienced a nearly complete loss of its breeding colony by 2005. Conversely, French Guiana saw significant colony increases during the same period (Abreu-Grobois & Plotkin, 2008). In the Dutch Caribbean, the olive ridley is notably absent from nesting sites, with rare sightings until 2008, when a specimen was found near Curaçao, followed by another in St. Eustatius (St. Eustatius Sea turtle Conservation Programme - Annual Report 2008). Recent observations of stranded turtles on Bonaire and Curaçao suggest an enhanced monitoring network or altered migration patterns. Like other sea turtles, the olive ridley has a complex life cycle, relying on distinct geographic areas and habitats, primarily living as oceanic carnivores and returning to the coast solely for reproduction (Abreu-Grobois & Plotkin, 2008).

A long-term study on growth rates of sea turtles from the West Atlantic highlights significant declines in growth rates among carnivorous sea turtle species, namely West Atlantic hawksbills and North Atlantic loggerheads, mirroring the patterns observed in green turtles (Bjorndal et al., 2017). Beginning around 1997 after peak growth rates, these declines are attributed to the same ecological regime shift phenomenon affecting the broader marine ecosystem. While the study emphasizes the role of thermal stressors in driving these declines, it underscores the compounding effects of multiple stressors, including anthropogenic degradation of foraging habitats.

Furthermore, rising temperatures have been shown to reduce the survival of hatchlings and increase the female-to-male ratio of emergent hatchlings because of temperature-dependent sex determination in sea turtles (Laloë et al., 2017). Hawksbills, closely associated with coral reefs, suffer from the extensive loss and degradation of reef habitats, while loggerheads face habitat destruction from trawl fisheries and accumulation of marine debris. The cumulative impacts of these stressors exacerbate the decline in growth rates across all sea turtle species, indicating a pressing need for comprehensive conservation measures to mitigate the threats posed by climate change and human activities to sea turtle populations worldwide. Fuentes et al. (2023) recently reviewed the key threats to sea turtle populations. They listed them as climate change, direct take, disease, pollution, predation, coastal and marine development, and fisheries (see Figure 2), underscoring that these marine reptiles are animals of high conservation concern. Many of these threats can lead to direct mortality (e.g., harvesting) or indirectly reduce the resilience or health of sea turtles (e.g., pollution).

In the Dutch Caribbean, five sea turtle species inhabit the waters, each with varying levels of protection and presence, as detailed in Tables 1 and 2.

The loggerhead (*Caretta caretta*), known locally as "Kawama," is present on Bonaire with both nesting and infrequent foraging activity, while it is absent from St. Eustatius and seen infrequently on Saba. The green turtle (*Chelonia mydas*), or "Turtuga Blanku," nests and forages on Bonaire and St. Eustatius and is also observed foraging infrequently on Saba. The leatherback (*Dermochelys coriacea*), locally known as "Drikil," nests infrequently on Bonaire, regularly on St. Eustatius, and infrequently sighted on Saba. The hawksbill (*Eretmochelys imbricata*), known as "Karet," nests and forages on Bonaire and St. Eustatius, while on Saba, it is only observed foraging. The olive ridley (*Lepidochelys olivacea*), locally called "Turtuga Bastardo," is sighted infrequently on Bonaire and absent from St. Eustatius and Saba.

Table 2. *An overview of the sea turtle species found in the waters of the Dutch Caribbean, their respective statuses in the IUCN category, and their presence on each island. Source:* (Eckert & Eckert, 2019)- *LC = Least Concern; EN = Endangered; VU = Vulnerable; CE = Critically Endangered; A = Absence; N = Nesting; F = Foraging; I = Infrequent (further detail unavailable); IN = Infrequent Nesting (following Eckert and Eckert, 2019).*

Name (Latin)	**Common Name (English)**	**Local Name**	**Dutch Name**	**Bonaire**	**St. Eustatius**	**Saba**
Caretta caretta	loggerhead	Kawama	Onechte Karetschildpad	N, IF	A	I
Chelonia mydas	green turtle	Turtuga Blanku	Soepschildpad/Groene Zeeschildpad	N, F	N, F	IN, F
Dermochelys coriacea	leatherback	Drikil	Lederschildpad	IN	N	I
Eretmochelys imbricata	hawksbill	Karet	Karetschildpad	N, F	N, F	F
Lepidochelys olivacea	olive ridley	Turtuga Bastardo	Warana	I	A	A

Assessment of National Conservation State

Nesting site fidelity, which refers to the tendency of individual adult female turtles to return to the same nesting areas within a limited geographical range, has been extensively studied in the literature. Traditionally, information on fidelity during movements between nesting events has been gathered through tag-recapture studies, as demonstrated by Limpus et al. (1992) and (Shimada et al., 2020). More recently, advances in satellite telemetry have further confirmed nesting site fidelity across various turtle species. Studies by Humber et al. (2014) and Whiting et al. (2021) have corroborated this behaviour in green turtles while Parker et al. (2009) and Walcott et al. (2012) have shown similar patterns in hawksbill turtles. Additionally, leatherback and loggerhead turtles have also exhibited nesting site fidelity, as evidenced by research conducted by e.g., Byrne et al. (2009) and Tucker (2010).

Establishing connectivity between rookeries and foraging habitats and determining phylogeography and broad-scale stock structure for most marine turtle species is important to increase the effectiveness and guidance of conservation measures. Recent genetic and migratory behaviour studies (e.g., Esteban et al. (2015); Becking et al. (2016)) show that the exchange between turtles from different nesting sites is sufficient to maintain genetic diversity in the Dutch Caribbean. Satellite tracking of sea turtles nesting at Bonaire and Klein Bonaire in the Caribbean Netherlands has provided valuable insights into their migration patterns. Becking et al. (2016) revealed migration distances ranging from 197 to 3135 km to foraging grounds across the Caribbean. These grounds include coastal waters where harvesting activities persist (García-Cruz et al., 2015; Humber et al., 2014; Lagueux et al., 2014) exposing young sea turtles to the anthropogenic threat. Both studies highlight that further research is required, particularly to unravel the migratory behaviour of male sea turtles. Both studies underscore the significance of international marine turtle conservation efforts, revealing extensive post-breeding migration routes.

The study of Esteban et al. concludes that green and hawksbill turtles nesting on St Eustatius and St Maarten in the eastern Caribbean demonstrate behavioural flexibility in their inter-nesting movements and post-nesting migration routes. While their nesting behaviour aligns with previous reports in the region, some turtles exhibited unconventional post-nesting migration behaviour, challenging the assumption of migratory behavior among adult female turtles in the Caribbean. The research also reveals varying nesting and post-nesting strategies among green and hawksbill turtles, with some individuals showing repeated nesting on the same beach and others nesting on beaches separated by significant distances. Satellite tracking data indicate that green turtles may nest on multiple islands nearby, suggesting a more comprehensive nesting range than previously thought. The study also reveals that some hawksbill turtles took indirect paths, travelling over 200 km to nest again before returning to foraging locations less than 50 km from their original nesting sites, a behaviour not previously documented.

Saba has no significant permanent beaches suitable for turtle nesting, with only two recorded instances. Nesting likely occurs only sporadically, but in 2015, a green turtle nest was found and successfully hatched on Saba. In scope of the report, we have also requested the Saba Conservation Foundation for up-to-date data, however, aside from sightings of foraging turtles, no nesting has been recorded on Saba in the past years (*personal communication*, Camille Tuijnman from SCF). The two islands with a significant nesting population of sea turtles are Bonaire and St. Eustatius.

Trends in the Caribbean Netherlands
Bonaire Nesting

We have collected data on the number of nests per location across the islands of Bonaire and St. Eustatius (Figure 3-5). The time-series data collected by STCB and STENAPA were aggregated at the island level. Only nests with confirmed species identifications were retained (nests of unknown species were excluded), all subregions were combined, and the resulting dataset was analyzed in RStudio using generalized linear models (GLMs) with a Poisson error distribution. Figures 3, 4, and 5 illustrate the nesting trends of three turtle species: hawksbill, loggerhead, and green turtle on Bonaire from 2003 to 2023.

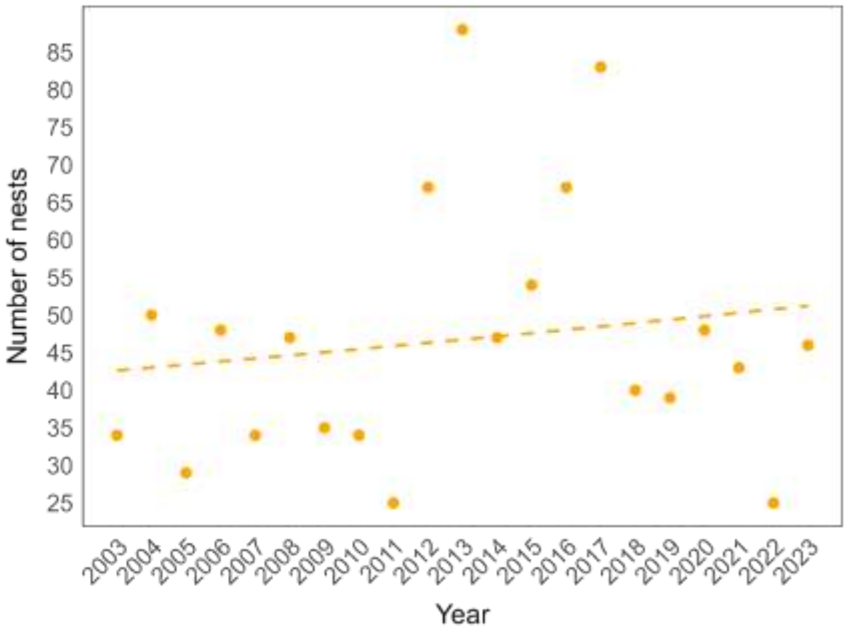

Figure 3. *GLM Analysis of hawksbill turtle nesting trends in Bonaire (2003–2023). Dots represent observed annual nesting counts, while the dashed line illustrates the trend estimated by the GLM.*

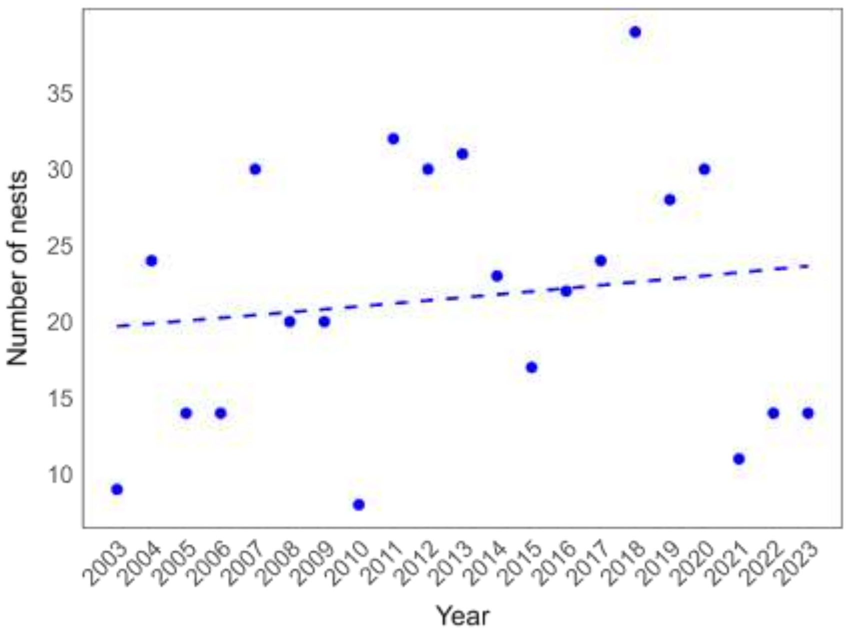

Figure 4. *GLM Analysis of loggerhead turtle nesting trends in Bonaire (2003–2023). Dots represent observed annual nesting counts, while the dashed line illustrates the trend estimated by the GLM.*

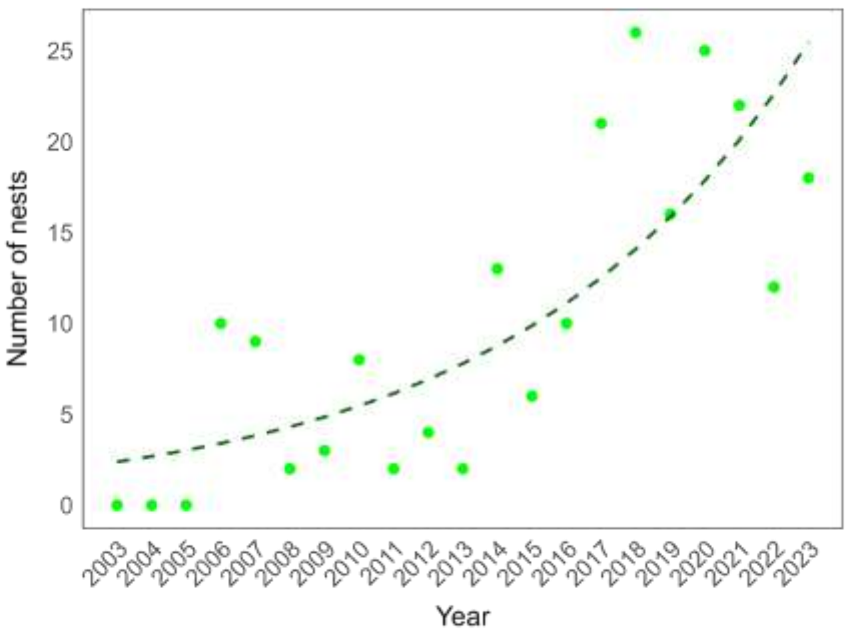

Figure 5. *GLM Analysis of green turtle nesting trends in Bonaire (2003–2023). Dots represent observed annual nesting counts, while the dashed line illustrates the trend estimated by the GLM.*

Key Findings

- loggerhead turtle: The analysis indicates no significant trend in nesting numbers over the years ($p = 0.127$). The AIC suggests a moderate model fit.
- hawksbill turtle: The model shows a non-significant trend ($p = 0.0616$), indicating a potential weak increase in nesting numbers.
- green turtle: This species exhibits a highly significant positive trend in nesting numbers ($p < 2e-16$), suggesting a substantial increase in nests over the years.

St. Eustatius Nesting

This analysis evaluates the nesting trends of three turtle species (green turtle, hawksbill, and leatherback) in St. Eustatius from the period 2003-2023 using Generalized Linear Models (GLMs) with a Poisson distribution (Figures 6, 7, and 8). Only nests with confirmed species identifications were retained (nests of unknown species were excluded), all subregions were combined, and the resulting dataset was analyzed in RStudio.

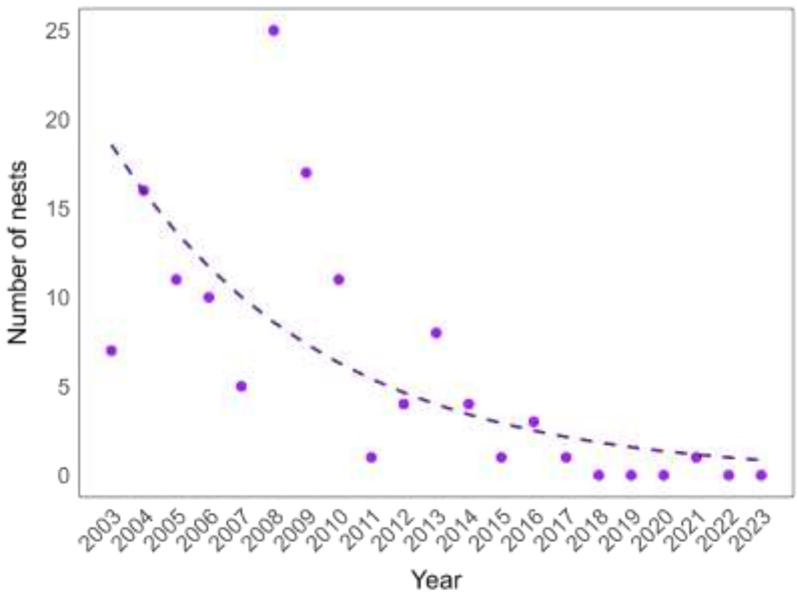

Figure 6. *GLM Analysis of leatherback turtle Nesting Trends in St. Eustatius (2003–2023). Dots represent observed annual nesting counts, while the dashed line illustrates the trend estimated by the GLM.*

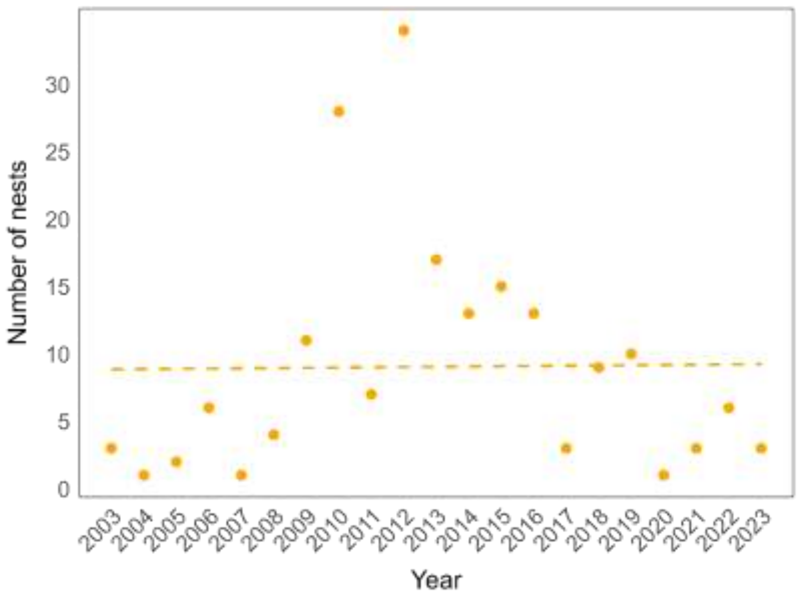

Figure 7. *GLM Analysis of hawksbill turtle Nesting Trends in St. Eustatius (2003–2023). Dots represent observed annual nesting counts, while the dashed line illustrates the trend estimated by the GLM.*

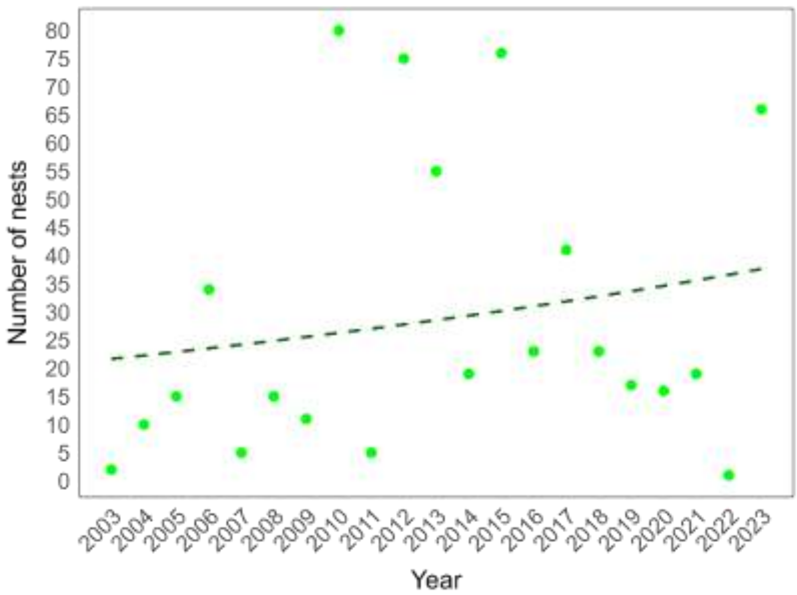

Figure 8. *GLM Analysis of green turtle Nesting Trends in St. Eustatius (2003–2023). Dots represent observed annual nesting counts, while the dashed line illustrates the trend estimated by the GLM.*

Key Findings

- green turtle: The analysis indicates a significant positive trend in nesting numbers (p = 0.000372), suggesting that nests increase over the years. The AIC of 578.62 suggests a moderate fit of the model.
- hawksbill turtle: The model shows no significant trend (p = 0.686), indicating that the number of nests does not significantly change over the years. The AIC value 235.18 suggests a poorer model fit than the green turtle.
- leatherback turtle: This species exhibits a highly significant negative trend in nesting numbers (p < 2e-16), indicating a substantial decrease in nests over the years. The AIC of 130.33 indicates the best fit among the three models analysed, as shown by the lower AIC value.

Overall, the green turtle nesting counts have significantly increased on St. Eustatius and Bonaire, indicating positive trends for this species, which is in line with global assessments (Hays et al., 2025) (Hays et al. 2025). However, such positive development does not mean that the sea turtles are not more susceptible to multiple stressors and remain threatened. This data may indicate that local and regional conservation efforts and the protective goals set by different regulations and directives, as mentioned above, lead to a positive development for this species. In contrast, hawksbill turtles exhibit a slight, though not statistically significant, increase in Bonaire, while no notable trend is observed in St. Eustatius. Similarly, no clear trends have been detected for loggerhead turtles on Bonaire. However, there is a significant and concerning decline in nesting numbers for leatherback turtles on St. Eustatius, suggesting a substantial population decrease for this species, in line with regional declines documented in the period from 2008-2017 in both the Northern and Western Caribbean (Eckert & Hart, 2021).

It must be kept in mind that the apparent rise in nest counts may accidentally reflect increased survey effort as more person-hours in the field, longer transect distances, and broader spatial coverage inevitably uncover more nests. To correct this, raw counts should be normalized by effort metrics (e.g. nests per person-hour or per km surveyed). Complementary capture–mark-recapture approaches—whether via flipper or PIT-tagging, photo-ID of individual scale patterns, or DNA-based genetic marks- can provide more robust estimates of nesting-female abundance. Although these methods require additional resources and logistical coordination, they yield critical parameters (population size, survival rates, remigration intervals) that are vital to accurate population assessments and informed management.

Nesting success

In St. Eustatius, the green turtle nesting success increased from 72.75% in 2020 to 92.50% in 2021, followed by a drastic decline to 31.3% in 2023. Bonaire's green turtles maintained robust success, ranging from 83% to 91% during the same period, indicating a more stable nesting environment based on the provided data. Hawksbill turtles in St. Eustatius had widely varying success rates, with no data available for 2019 and a low of 0.44% in 2023. In Bonaire, hawksbill's success declined from 81% in 2019 to 74% in 2023, yet it consistently surpassed figures from St. Eustatius. Loggerhead nesting success is reported only for Bonaire, which remained steady at around 80-90%. Data for leatherback turtles is limited to St. Eustatius, with a recorded % success rate of 24% in 2019, and no further data is available. In this analysis, sub-local variation in nesting success was incorporated. Therefore, this data does show locations with high human activity, which, in turn, may still negatively impact turtle populations, highlighting the continued need for protection of these critical habitats.

We conclude by pointing out that nesting success varies strongly between St. Eustatius and Bonaire. Bonaire shows consistently high hatch-success rates across multiple turtle species, whereas St. Eustatius data are more inconsistent. These differences indicate the need for sustained monitoring and targeted conservation to research the ecological drivers behind these patterns. However, uneven reporting due to a lack of personnel capacity may make distinguishing biological fluctuations from data collection gaps challenging. By contrast, Bonaire’s long-term datasets appear more complete to inform effective management strategies for sea turtle conservation.

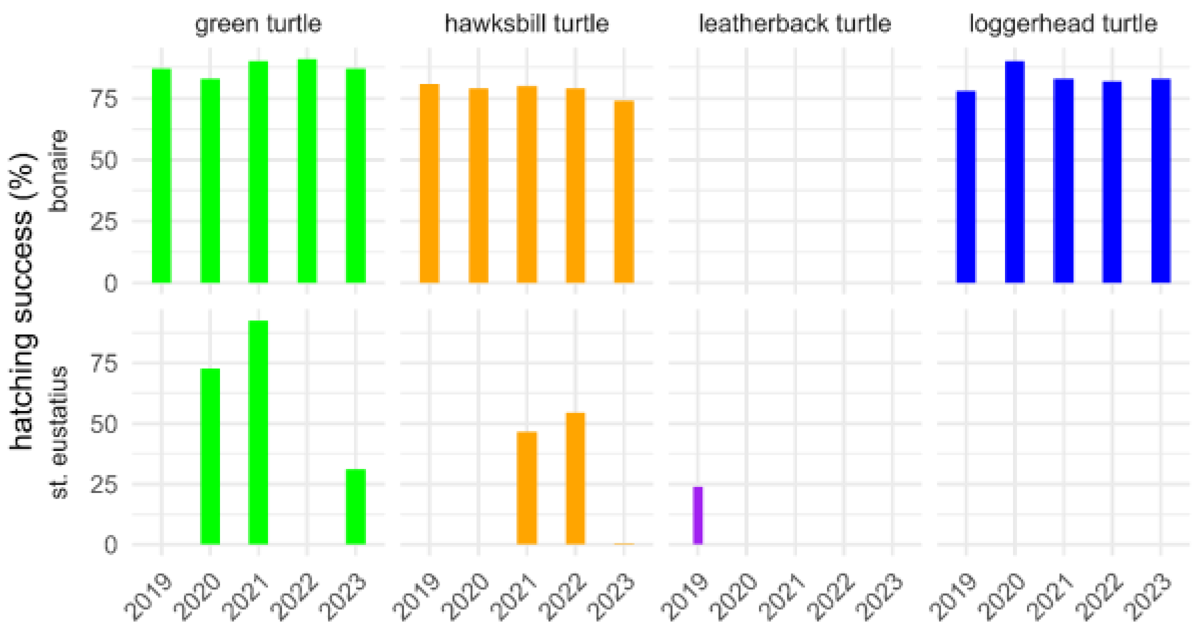

Figure 9. *Sea turtle Nesting Success in St. Eustatius and Bonaire. The figure consists of two sections: the upper section illustrates sea turtle nesting success data for Bonaire, while the lower section presents comparable data for Sint Eustatius.*

Foraging Grounds

The Dutch Caribbean islands are also rapidly becoming more critical as foraging areas (Debrot et al., 2005; Bjorndal, 2017). On Saba, the seagrass fields around the island are a fixed foraging area for subadult green turtles, and the hawksbill turtle regularly visits the island's reefs. There are also indications that the 2,200 km² Saba Bank is a foraging area for adult hawksbill turtles. The diversity of algae and sponges on the Saba Bank means ample food is available, especially for hawksbill turtles. Several adult hawksbill turtles were encountered during various dives on the Saba Bank (Lundvall, 2008). Also, a male hawksbill turtle equipped with a satellite transmitter in 2004 on Bonaire was tracked heading towards the Saba Bank until the transmitter's signal was lost prematurely. Leatherback turtles and loggerhead turtles have also been sighted on the Saba Bank. Little is known about the foraging areas of St. Eustatius, but more about the nesting beaches (as described above).

In Bonaire, hawksbill and green turtles are found island-wide in the shallow foraging areas of the coral reefs and seagrass fields. The densities of green turtles were higher everywhere compared to the hawksbill turtle. The highest concentrations of green turtles are found at Lac Bay on the east coast of the island, where the most significant food source for this species is located. In the last two decades, there has been a surge in available data from the BES islands, particularly from Bonaire, where systematic monitoring and tagging over twenty years has revealed that the total abundance of green turtles remained consistent throughout the recent years 2019 to 2022, showing no significant deviation when compared to both the survey-based estimates from 2003 to 2018 and the model-based predictions for 2019 to 2030 (Rivera-Milán et al., 2019). The forecast for hawksbill turtles from 2019 to 2022 also closely resembled those from 2003 to 2018 and the predictions for 2019 to 2030 (Rivera-Milán et al. 2019). In western Bonaire and Klein Bonaire, hawksbill turtles were less detectable from January to March compared to April. Detectability remained relatively stable for green turtles between these periods. Furthermore, genetic and demographic analyses reveal an increase in the proportion of juvenile green turtles from recovering rookeries in the northwestern Caribbean. This increase suggests a potential positive impact of sea turtle conservation measures in the region on juvenile abundance at feeding grounds. However, juvenile recruitment from the eastern Caribbean and southern Atlantic has decreased, signalling a concerning decline in reproductive output in those areas (Becking et al., 2016).

Recent analyses of marine sediments from Bonaire’s 2022 heavy-rainfall and runoff event show that metal and organic contaminant concentrations exceed during this period European guideline limits, posing a significant threat to aquatic life (Dogruer et al., 2025).Because many of these contaminants bioaccumulate in organisms, sediment-based monitoring offers valuable first insights into the quality of both foraging and nesting habitats (Leusch et al., 2021). To date, however, no biological contaminant surveys have been conducted on Caribbean aquatic species, including green turtles, which feed on seagrass and are particularly vulnerable when trace metals concentrate in sediments (Talavera-Saenz et al., 2007; Thomas et al., 2020). In Queensland, Australia, studies have linked persistent organic pollutant (POP) levels in coastal sediments with internal POP burdens in green turtles, demonstrating sediment-to-seagrass bioaccumulation pathways (Hermanussen et al., 2004, 2008; Weltmeyer et al., 2021). Assessing water-column pollutant concentrations alongside sediment and biota sampling is therefore essential to fully understand—and mitigate—the health risks that elevated chemical levels pose to sea turtle habitats(Gaus et al., 2019; Leusch et al., 2021).

The Bonaire case study reports cadmium concentrations that exceed sea-turtle–specific protective thresholds (Dogruer et al., 2021). The risk quotient indicates how much the measured level exceeds (or falls below) those benchmarks; values above one signal an early warning of toxicity (Figure 10).

In Lagun, cadmium levels are markedly higher than in more urbanized and industrialized foraging grounds in Japan and Australia (Figure 10), countering the usual west-to-east decline in turtle tissue burdens. Elemental analyses of *Thalassia testudinum* leaves collected at three sites in Lagun (March 2022) revealed a mean cadmium concentration of 3.27 µg/g dry weight (Ouwersloot, 2022). Ecological studies have linked cadmium at these levels to reduced rhizome density and diminished seagrass resilience (Fraser & Kendrick, 2017), suggesting that the same protective threshold may safeguard turtle health and habitat integrity. Moreover, copper and zinc in Lagun’s seagrass also exceed species-specific toxicity thresholds (Ouwersloot, 2022). Copper exposure has been implicated in fibropapillomatosis (FP), a tumour-forming disease affecting sea turtles (da Silva et al., 2016)—, and high FP rates have been reported locally (STCB, pers. comm.). These observations underline the need to assess how metal contamination influences sea turtle health.

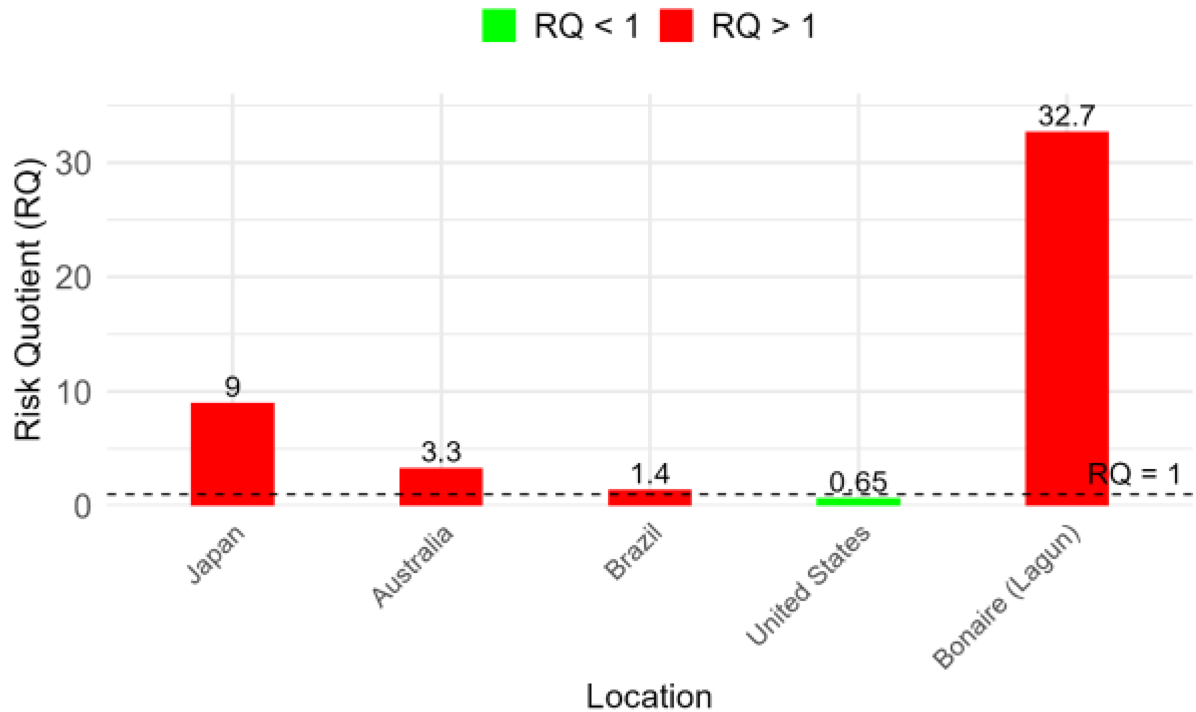

Figure 10. *The risk-quotient(RQ)-based risk evaluation, as described in Dogruer et al. 2021. Seagrass concentrations are evaluated against the species-specific threshold for green turtles in Australia, the United States, Brazil, Japan and Bonaire (Lagun). The concentration in seagrass was normalized to the proposed threshold values. An RQ > 1 typically signals a potential risk and the need for further investigation or mitigation.*

Although these contaminant levels may not cause immediate mortality, chronic sublethal exposure erodes turtles' resilience, making them more vulnerable to disease and environmental stressors. With cadmium's biological half-life extending beyond 30 years, the next logical step is to measure internal cadmium burdens in turtle organs and compare them to harmful thresholds. The elevated external concentrations suggest toxicological risk, highlighting the necessity for ongoing monitoring and targeted research on bioaccumulation and long-term health impacts in these endangered species.

Based on all the above, Tables 3 and 4 give an overall assessment of the nesting turtles of the Caribbean Netherlands. In general, distribution and population size are judged as "favourable" or "unfavourable-inadequate". Based on the criteria used, the overall CS for all species distribution is "favourable" except the leatherback turtle population of St.Eustatius is considered "unfavourable-inadequate", while habitat is considered an "unfavourable-bad" factor for all except the loggerhead turtle, which forages mainly offshore. The population size is only for the green turtles categorised as "favourable".

Table 3. *Diagnostic scores for the four different State of Nature criteria for the three breeding sea turtles of Bonaire and an overall conservation assessment for the year 2024.*

Aspect of sea turtles Bonaire	**loggerhead turtle**	**hawksbill turtle**	**green turtle**
Distribution	Favourable	Favourable	Favourable
Population size	Unfavourable-inadequate	Unfavourable-inadequate	Favourable
Habitat	Unfavourable-inadequate	Unfavourable-bad	Unfavourable-bad
Future prospects	Unfavourable-inadequate	Unfavourable-inadequate	Unfavourable-inadequate
Overall Assessment of Conservation State	**Unfavourable-inadequate**	**Unfavourable-inadequate**	**Unfavourable-inadequate**

Table 4. *Diagnostic scores for the four different State of Nature criteria for the three breeding sea turtles of St. Eustatius and an overall conservation assessment for the year 2024.*

Aspect of sea turtles St. Eustatius	**leatherback turtle**	**hawksbill turtle**	**green turtle**
Distribution	Unfavourable-inadequate	Favourable	Favourable
Population size	Unfavourable-bad	Unfavourable-inadequate	Favourable
Habitat	Unfavourable-bad	Unfavourable-bad	Unfavourable-bad
Future prospects	Unfavourable-inadequate	Unfavourable-inadequate	Unfavourable-inadequate
Overall Assessment of Conservation State	**Unfavourable-bad**	**Unfavourable-inadequate**	**Unfavourable-inadequate**

Comparison to the 2018 State of Nature Report

Overall, compared to the 2018 assessment, several changes can be detected in the CS of different sea turtle populations of the Caribbean Netherlands. Most encouraging are the long-term increases in the nesting populations of the green turtle on Bonaire and St. Eustatius. In contrast, the substantial declines in leatherback turtle nesting on St. Eustatius are worrisome.

Recommendations for National Conservation Objectives

Habitat Protection and Restoration:

Nesting Beaches:

- Enforce regulations to limit coastal development and sand excavation (and enforcement in general, for example, also cats & dogs on beaches)
- Implement measures to control light pollution.
- Protect and restore natural vegetation on beaches to provide shade and cooling for nests.

Foraging Grounds:

- Improve water quality by controlling run-off.
- Protect seagrass beds and coral reefs from pollution and physical damage.
- Manage Sargassum influxes through environmentally sensitive removal methods.
- Enforcement (e.g. speed limits for boats, regulation for wind/kite/foil surfers)

Research and Data Collection:

Regular Monitoring:

- Conduct regular surveys to monitor the health and population dynamics of sea turtles and their habitats.
- Ensure staff capacity to ensure consistency of monitoring on all islands to avoid disperse data

Chemical pollution and Disease Surveillance:

- Implement programs to monitor the prevalence of diseases such as fibropapillomatosis and the correlation with contaminants.

Migration Patterns:

- Use satellite tracking and/or genetic analyses to gather data on migration patterns and identify critical habitats for protection.
- Understanding the drivers in nesting success fluctuations

Public Awareness and Education:

Community Involvement:

- Engage local communities in conservation efforts through education and involvement in monitoring programs.

Tourism Management:

- Educate tourists about the importance of sea turtle conservation, the protection of their foraging (seagrass) habitat and encourage responsible behaviour on nesting beaches.

Legislation and Enforcement:

Enforce Regulations:

Implement and enforce existing regulations to reduce bycatch and illegal fishing.

Protected Areas:

Establish and enforce marine protected areas to safeguard critical nesting and foraging habitats.

Further recommendations:

Develop a new joint Sea turtle Recovery Action Plan (STRAP), as the current ones (Sybesma 1992 and Barmes et al., 1993) are highly outdated. The new STRAP should be part of the cooperative management of the marine biodiversity and fisheries of the Caribbean islands within the Kingdom of the Netherlands (EEZ MoA and Management Plan) and should include:

- Management and interventions to maintain and/or enhance nesting beach quality (in terms of beach size and setback to allow for encroaching sea level rise, sand depth, and quality natural vegetation, as well as control on disturbance of beaches and nests);
- Improve knowledge of foraging sea turtle populations around the islands and their ecosystem effects and dependence.
- Qualify, quantify, and address threats due to climate change.
- Expand regional international collaboration, especially towards countries such as Nicaragua, where the best data available shows that Dutch Caribbean nesting turtles spend part of their life cycle.
- Institutionalize essential monitoring to accurately follow population and nesting trends to evaluate strategies and interventions for adaptive management purposes.

Key Threats and Management Implications

Table 5. *Listing of different threat categories to sea turtle species, their predominant cause and the ensuing management implications.*

Category	Threat	Description	Status: high concern/moderate/ low/unkown
Nesting Beaches	**Coastal development**	Urbanization and infrastructure projects cause habitat loss and fragmentation, reducing available nesting areas.	**Bonaire: High** **St. Eustatius: Moderate**
	Sand excavation	Sand removal reduces nesting habitat, especially on St. Eustatius, where thin sand layers hinder egg-laying and hatching success.	**Bonaire: Low** **St. Eustatius: Moderate**

	Erosion	Overgrazing causes cliff erosion, increasing the vulnerability of nesting beaches on St. Eustatius.	**Bonaire: Low** **St. Eustatius: High**
	Plastic pollution	Litter on beaches and plastic in the ocean threaten adult and newborn turtles by blocking hatchlings' emergence and causing harm at sea.	**Bonaire: Moderate** **St. Eustatius: Moderate**
	Light pollution	Disorients hatchlings and egg-laying females, leading them inland, where they may die from dehydration.	**Bonaire: High** **St. Eustatius: Moderate**
	Oil spills	Risk of oil spills from cargo ships near St. Eustatius threatens nesting and foraging turtles.	**Bonaire: Moderate** **St. Eustatius: Moderate** **Saba: Moderate**
	Higher temperatures	Alters nesting site conditions, affecting sex ratios and hatchling survival rates. Higher temperatures can skew the sex ratio towards females, especially on beaches with less sand and vegetation (Laloë et al., 2016). For example, on St. Eustatius, where the sand is black and absorbs more heat, 85.9–93.5% of the young turtles are female.	**Bonaire: High** **St. Eustatius: High**
	Recreational activities	Increasing tourism on beaches, particularly in Bonaire, causes trampling of nests, necessitating protective measures like marking and fencing.	**Bonaire: High** **St. Eustatius: Low** **Saba: Low**
Foraging Grounds	**Chemical Pollution**	Land-based activities degrade water quality. Elevated chemical pollution in sediment and seagrass tissues exceeds safety thresholds, causing potential health concerns.	**Bonaire: High** **St. Eustatius: unknown** **Saba: unknown**
	Coral reef degradation	Human activities and climate change destroy coral reefs' essential turtle habitats and reduce food availability.	**Bonaire: High** **St. Eustatius: High** **Saba: High**
	Seagrass habitat degradation	Over the past three decades, Bonaire's seagrass habitat has decreased by more than 2 hectares yearly (Hylkema et al., 2014).	**Bonaire: High** **St. Eustatius: unknown** **Saba: unknown**
	Sargassum blooms	Large mats of Sargassum seaweed hinder hatchlings, and its decomposition destroys seagrass beds, creating anaerobic conditions harmful to marine life.	**Bonaire: High** **St. Eustatius: Moderate** **Saba: Low**
	Recreational activities	Aquatic tourism activities (e.g., boating, kitesurfing) can collide with foraging sea turtles in seagrass and coral reef areas)	**Bonaire: High** **St. Eustatius: Moderate** **Saba: Low**
Migrating Species	**Bycatch**	Turtles are unintentionally caught in fishing gear, leading to injuries or death.	**Bonaire: High** **St. Eustatius: unknown** **Saba: Low**
	Poaching (regionally)	Becking et al. (2016) revealed migration distances ranging from 197 to 3135 km to foraging grounds across the Caribbean. These grounds include coastal waters where harvesting activities persist.	**Bonaire: Low** **St. Eustatius: Low** **Saba: Low**

Data Quality and Completeness

Data quality for assessing local trends in nesting frequency and hatching success varies markedly among the Dutch Caribbean islands. Bonaire benefits from long-term, robust monitoring programs, whereas St. Eustatius suffers from fragmented turtle surveys. Green turtle monitoring is absent on Saba, although its expansive seagrass beds are an important foraging habitat. Because sea turtles are highly migratory, insights gained from one location may reflect regional dynamics, yet such broader datasets are often unavailable or non-standardized. Consequently, it remains unclear whether observed fluctuations in nesting are driven by local conservation measures or climate-driven shifts in migratory behaviour. With “casually”-collected data, it must be kept in mind that nest counts may reflect increased survey effort as more person-hours in the field, longer transect distances, and broader spatial coverage inevitably uncover more nests. To correct this, raw counts should be normalized by effort metrics (e.g. nests per person-hour or per km surveyed), as pointed out above. Complementary capture–mark-recapture approaches—whether via flipper or PIT-tagging, photo-ID of individual scale patterns, or DNA-based genetic marks- can provide more robust estimates of nesting-female abundance. Although these methods require additional resources and logistical coordination, they yield critical parameters (population size, survival rates, remigration intervals) that are critical for accurate population assessments and informed management. Finally, very little is known about how coastal pollution and changing beach parameters (e.g., sand temperature and moisture regimes) affect turtle health and reproductive success. Addressing these critical gaps will require harmonized, multi-island monitoring protocols and targeted studies on contaminant burdens and climate impacts at nesting beaches.

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Status

Sharks and rays (i.e., elasmobranchs) are often considered a vital part of marine ecosystems (Heithaus et al., 2022). Their presence can be used as an indicator for ecosystem health, and this species group can have a large variety of roles in marine food webs depending on species and life stage (Dedman et al., 2024; Flowers et al., 2021). Although their ecological roles are important in a variety of marine ecosystems, their global status has deteriorated over the past decades (Dulvy et al., 2021; Simpfendorfer et al. 2023; Sherman et al., 2023). Currently, over 32% of all shark and ray species are threatened with extinction (Dulvy et al., 2021). Oceanic shark and ray species have declined by an estimated 71% over the past five decades (Pacoureau et al., 2021). Recent estimates show that on 20% of coral reef ecosystems around the world, sharks are completely absent (MacNeil et al., 2020). The waters of the Dutch Caribbean islands (Aruba, Bonaire, Curaçao, Saba, Sint-Maarten, and Sint-Eustatius) overlap with the distribution of 95 species (61 shark and 34 ray species), of which 47% is currently threatened with extinction (or extirpated in case of the large-tooth sawfish, *Pristis pristis*; IUCN, 2022). Based on these distributions the waters of the BES-islands (i.e., Bonaire, Sint-Eustatius and Saba) may potentially host 87 species for which presence of 41% (n = 36 species) is confirmed. This amounts to 8 species more than the 28 species confirmed in the first inventory by van Beek et al. (2012). However, the presence of the majority of the species in Table 1, has yet to be confirmed around these islands. For the three BES-islands, most species were confirmed for the EEZ of Saba (n = 25 species, 49% of expected species). Table 1 provides an updated species list for the six Dutch Caribbean islands and their exclusive economic zones and is based on the species list presented by van Beek et al. (2012). This resulted in 52 new island records for the six islands in total. New records were based on published literature, online video/photographic material or observations shared with the authors by local researchers, divers and (recreational) fishers. The waters of the Dutch Caribbean are data deficient for many shark and ray species, as the presence of 60% (n = 58) of species has not been confirmed (54% of shark and 82% of ray species listed in Table 1 can be expected to be present) (Figure 1). The main reason for this lack of records is that research on this group for the Dutch Caribbean waters has been limited, especially species using pelagic or deep sea habitats, for which 35% and 79% of the species are data deficient respectively.

Characteristics

Description
Only few studies and reports mention the diversity of sharks and rays in the waters of the Dutch Caribbean. Although the windward and leeward islands are over 850 km apart, these island groups share similar species composition (Table 1). Especially pelagic species have a wide distribution and range throughout the entire Caribbean Sea region. In terms of coastal and reef-associated species, the islands share common species like the Caribbean reef shark (*Carcharhinus perezi*), nurse shark (*Ginglymostoma cirratum*), Southern stingray (*Hypanus americanus*), and spotted eagle ray (*Aetobatus narinari*). The movement and habitat use of Caribbean reef sharks and nurse sharks have been studied in the waters of the windward islands over the past years, concluding that these species reside in the waters of their respective islands for prolonged time periods and often show high fidelity to the same reef locality

(Winter et al., 2018). Young individuals of these species use the shallow waters of these islands as potential nursery area (Stoffers et al., 2021). Adult tiger sharks (*Galeocerdo cuvier*) move between the waters of the windward islands and likely use the Saba Bank as a mating and/or parturition site (unpublished observations of aggregations of adult tiger sharks with fresh bite marks, Winter). The long-distance movements of species like tiger shark between different EEZs in the region and the continued shark fisheries within some waters, support the need for a regional network of MPAs to protect large shark species (Gallagher et al., 2020).

The waters around the windward islands (Sint-Eustatius and Saba, incl. Saba Bank) differ in elasmobranch species composition from the waters around the leeward islands (Bonaire), especially in their (expected) diversity in ray species. For the Leeward islands, 18 ray species are occurring or expected to occur, whereas for the windwards islands there are 31 species of rays potentially present (Table 1).

Based on curated distribution maps per species (i.e., IUCN Shark Specialist Group and Ebert et al. 2021), many species of sharks and rays are expected to be present in the waters of the Dutch Caribbean but so far their presence has not been confirmed yet (Figure 1).

For the Windward Islands species that have not been confirmed yet, but are expected to be present, predominantly include pelagic and deep-sea species (Table 1). This includes deep-sea skates (Rajidae), lanternsharks (*Etmopterus spp*.), gulper sharks (*Centrophorus spp.*), deep-sea catsharks (*Galeus spp., Apristurus spp.*), and pelagic species such as blue shark (Prionace glauca), smooth hammerhead shark (*Sphyrna zygaena*), and shortfin mako (*Isurus oxyrinchus*). For the Leeward islands species that can be expected to be present, but have not been confirmed, also mostly include deep-sea species, such as lanternsharks, gulper sharks, catsharks, and skates (Table 1). However, some demersal and reef-associated species that can be expected in these waters and are yet to be confirmed are the cownose rays (Rhinoptera spp.), the yellow stingray (*Urobatis jamaicensis*), chuparee stingray (*Styracura schmardae*), and reef-associated sharks such as the blacknose shark (*Carcharhinus acronotus*).

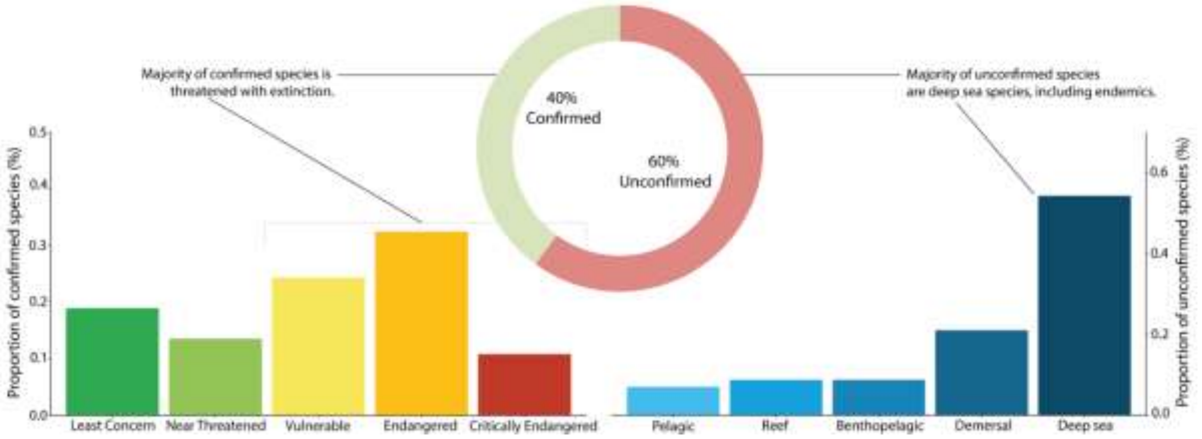

Figure 12. *Status of sharks and rays in the Dutch Caribbean. For 40% of the species the presence has been confirmed, of which the majority is currently threatened with extinction. The presence of the majority of species remains unconfirmed, with mostly species in the deep sea for which the presence is currently unknown.*

Of the 12 range-restricted (i.e., endemic; species with a distribution range of < 100,000 km2) species included in the species list for the Dutch Caribbean, the presence of only two species was confirmed: the boa catshark (*Scyliorhinus boa*, Saba) and the white-saddled catshark (*Scyliorhinus hesperius*, Bonaire) (Table 1). The large majority (83% of endemic species) of these species represent deep sea species, which complicates determining their presence and highlights the need for future research on deep-sea sharks and rays in the Dutch Caribbean. In the waters of the BES-islands, most of these endemic species are expected to occur in the waters of Saba (n = 7), followed by Sint-Eustatius (n = 6), and Bonaire (n = 3). The only endemic non-deep-sea species that are supposed to be indigenous to the BES-island waters are the Venezuelan dwarf smoothhound (*Mustelus minicanis*, unconfirmed yet) and the Venezuelan dwarf numbfish (*Diplobatus guamachensis*, unconfirmed yet). Given the data deficiency of the deep-sea shark and ray community within the Caribbean Sea, it is likely that the number of indigenous species will increase when more research is focused to these species in deep sea habitats.

Concluding remarks:

- The presence of many sharks and rays was confirmed for the Dutch Caribbean islands and we provide a new, updated species list here for this vulnerable and data deficient species group.
- The overall (expected) diversity of sharks and rays appears to be higher in the waters of the Leeward islands.
- Deep sea species are data deficient for all waters and basic information is still missing.
- The limited data availability and expected species occurrence shows that endemic species richness is higher in the Windward islands compared to the Leeward islands.
- Once common species such as the now highly endangered Oceanic whitetip shark could be confirmed for some islands by a limited number of observations over the past decades. Reflecting the limited knowledge and worrying Conservation State of large pelagic shark species in the Dutch Caribbean waters.

Relative Importance Within the Caribbean

The waters surrounding the islands of the Dutch Caribbean provide a range of habitats (e.g., mangroves, coral reefs, deep shelfs, pelagic zone) that are important to different shark and ray species or specific life stages (e.g., mangroves are important nursery areas for lemon sharks *Negaprion brevirostris*). The Saba Bank in particular is a large and important reef system hosting reef-associated elasmobranchs with sharks moving from and to the Saba Bank from different regions around it (Saba, St Eustatius, St Maarten and the US Virgin islands: Winter et al., 2019, and unpublished telemetry data), indicating of being of regional importance for tiger sharks and functions as a nursery area for the endangered silky shark (*Carcharhinus falciformis*). The Dutch Caribbean likely hosts some range-restricted species of deep-sea sharks and rays, especially in the waters of the Windward islands. Although the Caribbean Sea is not a hotspot for shark and ray endemism, the adjacent northern coasts of Colombia and Venezuela have a high richness of endemic shark and ray species (Stein et al., 2018). The occurrence of range-restricted species is limited compared to other regions. However, the importance of the Dutch Caribbean waters cannot be assessed properly for the majority of shark and ray species at this stage, as basic information for their status assessment is lacking, especially for pelagic and deep-sea species.

Ecological Aspects

Habitat: coral reefs, sandflats, pelagic zone and deep sea.

Sharks and rays occur in a variety of habitats in the Dutch Caribbean, including coral reefs and adjacent sandflats, seagrass fields, the pelagic zone and the deep sea (Winter and de Graaf, 2019). Previous research has shown that Caribbean reef sharks and nurse sharks may use the coral reefs surrounding these islands throughout their lifecycle, with young individuals using more shallower reefs (Stoffers et al., 2021). Mangroves, adjacent sandflats and seagrass beds are important to early life stages of stingrays (Dasyatidae), lemon sharks and requiem sharks (Carcharhinidae; Knip et al., 2010). However, coastal reefs, mangrove and seagrass systems in the Dutch Caribbean are under pressure of coastal development for the tourism sector or for new ports. This is in addition to the long-term stressors that these ecosystems face, such as climate-change induced coral bleaching, increased nutrient runoff and the spread of coral diseases (Bouchon et al., 2008).

Table 1. *Confirmed and expected occurrence of sharks and rays in the Caribbean Netherlands (Bonaire, Saba and Sint Eustatius), and the waters of Aruba, Curaçao and Sint Maarten. Species highlighted in blue are regionally endemic species (i.e., range is <100,000km2). Occurrence is based on documented and validated observations (X, green; references below the table), unconfirmed information (U, orange), or occurrence is expected (E) based on species distribution maps (IUCN Red List). The IUCN Red List status classes: Least concern (LC), Near Threatened (NT), Vulnerable (VU), Endangered (EN) and Critically Endangered (CR) and status of a species on appendices of relevant conventions is also shown.*

			Confirmed and expected occurrence						Status			
Habitat	**Scientific name**	**Common name**	**Aruba**	**Curaçao**	**Bonaire**	**Saba**	**St. Maarten**	**St. Eustatius**	**IUCN**	**CITES**	**CMS**	**SPAW**
	Ray species											
Bentho-pelagic	*Aetobatus narinari*	Whitespotted eagle ray	X13	X5, 22	X5,22	X13	X13	X13	EN			
	Myliobatis freminvillei	Bullnose eagle ray	E	E	E				VU			
	Myliobatis goodei	Southern eagle ray	E	E	E				VU			
Reef	*Mobula birostris*	Oceanic manta ray	X27	X16,22	X9	X24	E	E	EN	II	I/II	III
	Mobula cf. birostris	Mobula cf. birostris	X27	X27	X27	X27	X27	X27	-			
Demersal	*Diplobatis guamachensis*	Venezuelan dwarf numbfish	E	E					VU			
	Gymnura micrura	Smooth butterfly ray	E	E	E				NT			
	Hypanus americanus	Southern stingray	X13	X5, 22	X5,22	X13	X13	X13	NT			
	Hypanus guttatus	Longnose stingray	E	E	E	E			NT			
	Hypanus say	Bluntnose stingray				E	E	E	NT			
	Narcine bancroftii	Caribbean numbfish	E	E	E	E	E	E	LC			
	Pristis pristis	Largetooth sawfish	E	X4, 6					CR	I	I/II	II
	Pseudobatos percellens	Chola guitarfish	E	E	E	E	E	E	EN	II		
	Rhinoptera bonasus	American cownose ray	E	E	E				VU			
	Rhinoptera brasiliensis	Brazilian cownose ray	E	E	E				VU			
	Styracura schmardae	Atlantic chupare	E	E	E	X17			EN			
	Urobatis jamaicensis	Yellow stingray	E	E	E	E			LC			
Deep sea	*Benthobatis marcida*	Caribbean blind numbfish	E	E					LC			
	Breviraja nigriventralis	Blackbelly skate	E						LC			
	Cruriraja rugosa	Rough pygmy skate	E			E	E	X24	LC			
	Dactylobatus clarkii	Hook skate	E	E	E	E	E	E	LC			
	Dipturus bullisi	Tortugas skate	E	E	E				LC			
	Dipturus garricki	San Blas skate	E	E	E				LC			
	Dipturus teevani	Caribbean skate	E	E	E	E	E	E	LC			
	Fenestraja sinusmexicanus	Gulf pygmy skate	E						LC			
	Gurgesiella atlantica	Atlantic finless skate	E	E	E				LC			
	Pseudoraja fischeri	Fanfin skate				E	E	E	LC			
	Rajella fuliginea	Sooty skate	E	E	E	E	E	E	LC			
	Rostroraja cervigoni	Venezuela skate	E	E	E				NT			
	Schroederobatis americana	American legskate	E	E	E				LC			
	Springeria longirostris	Longnose legskate				E	E	E	LC			
Pelagic	*Mobula hypostoma*	Atlantic pygmy devil ray	E	E	X27	U27			EN	II	I/II	

	Mobula tarapacana	Sicklefin devil ray	X27	X27	X27				EN	II	I/II
	Pteroplatytrygon violacea	Pelagic stingray	E	E	E	E	E	E	LC		
	Shark species										
Bentho-pelagic	*Carcharhinus signatus*	Night shark	E	E	E	E	E	E	EN	II	
	Galeocerdo cuvier	Tiger shark	X2,22	X16	X12	X7,8	X9,22	X10	NT		
	Mustelus minicanis	Venezuelan dwarf smoothhound	E						EN		
	Sphyrna tudes	Smalleye hammerhead	E	E	E				CR	II	
Reef	*Carcharhinus acronotus*	Blacknose shark	E	E	E	E			EN	II	
	Carcharhinus altimus	Bignose shark	E	E	E				NT	II	
	Carcharhinus galapagensis	Galapagos shark				E			LC	II	
	Carcharhinus leucas	Bull shark	X2,22	X16	X2	X8,18	X9	X10	VU	II	
	Carcharhinus limbatus	Blacktip shark	X2	X16,22	E	X8,18	X9	X22	VU	II	
	Carcharhinus obscurus	Dusky shark	X2	E	U17				EN	II	II
	Carcharhinus perezi	Caribbean reef shark	X2	X22	X2	X7,8	X9	X10	EN	II	
	Carcharhinus porosus	Smalltail shark	E	E	E				CR	II	
	Ginglymostoma cirratum	Atlantic nurse shark	X2	X4	X3	X7,8	X9	X10	VU		
	Negaprion brevirostris	Lemon shark	X2,22	X24	X25	U26	X9,17	X21	VU	II	
	Rhizoprionodon lalandii	Brazilian sharpnose shark	E	E	E				VU	II	
	Rhizoprionodon porosus	Caribbean sharpnose shark	X17,22	E	E	X17	E	E	VU	II	
	Sphyrna tiburo	Bonnethead shark	X2	X5	E				EN	II	
Demersal	*Mustelus canis*	Dusky smoothhound	E	E	X17	E	E	E	NT		
	Mustelus higmani	Smalleye smoothhound	E	E	E				EN		
	Mustelus norrisi	Narrowfin smoothhound	E	E	E				NT		
	Sphyrna media	Scoophead shark	E	E	E				CR	II	
Deep sea	*Apristurus canutus*	Hoary catshark		X5		X24	E	X24	LC		
	Apristurus riveri	Broadgill catshark	E	E	E				LC		
	Centrophorus granulosus	Gulper shark				U26	E	E	EN		
	Centrophorus squamosus	Leafscale gulper shark		E	E				EN		
	Centrophorus uyato	Little gulper shark	E	E	E	E	E	E	EN		
	Etmopterus bigelowi	Blurred lanternshark	E	E	E	E	E	E	LC		
	Etmopterus bullisi	Lined lanternshark				X7	E	E	LC		
	Etmopterus gracilispinis	Broadbanded lanternshark	E	E	E				LC		
	Etmopterus hillianus	Caribbean lanternshark				E	E	E	LC		
	Etmopterus robinsi	West Indian lanternshark				E	E	E	LC		
	Etmopterus schultzi	Fringefin lanternshark	E	E	E				LC		
	Etmopterus virens	Green lanternshark	E	E	E				LC		
	Euprotomicrus bispinatus	Pygmy shark	E	E	E	E	E	E	LC		
	Galeus antillensis	Antilles catshark				E	E	E	LC		
	Galeus springeri	Springer's sawtail catshark				E	E	E	LC		
	Heptranchias perlo	Sharpnose sevengill shark	E	E	E				NT		
	Hexanchus griseus	Bluntnose sixgill shark	E	X13	E				NT		
	Hexanchus vitulus	Atlantic sixgill shark	E	X4,5	E	X8			LC		

	Oxynotus caribbaeus	Caribbean roughshark	E	E	E				LC			
	Scyliorhinus boa	Boa catshark	E	E	E	X17	E	E	LC			
	Scyliorhinus hesperius	White-saddled catshark	X13	X17	X13				LC			
	Squalus clarkae	Genie's dogfish	E		E	E	E	E	LC			
	Squalus cubensis	Cuban dogfish	E	X4	X17	X7	E	E	LC			
	Squatina david	David's angelshark	E	E	E				NT			
	Zameus squamulosus	Velvet dogfish		E	E				LC			
Pelagic	*Alopias superciliosus*	Bigeye thresher	X2,17	E	E				VU	II	II	
	Alopias vulpinus	Common thresher	X2	E	E				VU	II	II	
	Carcharhinus falciformis	Silky shark	X25	X4, 16	X17	X19	E	X19	VU	II	II	III
	Carcharhinus longimanus	Oceanic whitetip shark	E	X4, 14	X17	X17,26	E	E	CR	II	I	III
	Carcharodon carcharias	White shark				E	E	E	VU	II	I/II	
	Cetorhinus maximus	Basking shark	X23	E	E				EN	II	I/II	
	Isurus oxyrinchus	Shortfin mako	X2,25	X16	X17	E	E	E	EN	II	II	
	Isurus paucus	Longfin mako	E	E	E	E	E	E	EN	II	II	
	Megachasma pelagios	Megamouth shark	E	E	E	E	E	E	LC			
	Prionace glauca	Blue shark	E	X4	X25	U20	E	E	NT	II	II	
	Rhincodon typus	Whale shark	X1,2,28	X1, 28	X1,3,28	X1,28	X1,9,28	X1,28	EN	II	I/II	III
	Sphyrna lewini	Scalloped hammerhead	X2	X11	X13	X17			CR	II	II	III
	Sphyrna mokarran	Great hammerhead	X2	X15,22	X22	X17	X9	E	CR	II	II	III
	Sphyrna zygaena	Smooth hammerhead	X2			E	E	E	VU	II	II	III

References. 1-13: previous species list by Dolfi et al. (2013); 14: ongoing research by Wageningen Marine Research (Clements pers. obs.); 15: youtube video; 16: shark study by Hübner and Leurs (in prep.); 17: Fishers and divers shared observations with authors; 18: pictures published by the Dutch Caribbean Coast Guard; 19: ongoing research by Leurs et al. (in prep.); 20: observed around humpback whale carcass; 21: Leurs et al. (2018); Winter & de Graaf (2019); 23: Geelhoed et al. (2016); 24: observations from the Global Biodiversity Information Facility; 25: online published media by fishing charters; 26: Saba Bank Management Unit/A. Kuramae Izioka; 27: Database curated by the Manta Trust/Caribbean Island Manta Conservation Program; 28: Debrot et al. (2013).

This has significantly impacted coral reefs in the wider Caribbean region (Bouchon et al., 2008; Jackson et al., 2014), including the Dutch Caribbean (Sommer et al., 2011; Meesters, 2010). These stressors have impacted the extent and quality of coral reefs, mangroves and seagrass beds as base of Caribbean food webs and habitat for sharks and rays. The introduction of invasive predators such as the lionfish may also impact reef-associated food webs (e.g., reduction of herbivores) of which sharks and rays are a part.

Our knowledge on the pelagic and deep sea as a habitat for sharks and rays is limited since hardly any research has been performed in these respective domains. The pelagic (oceanic) zone is important for species that were once amongst the most common elasmobranch species in the world's oceans: the oceanic whitetip shark, mako shark and silky shark. These long-distance migrants occur in the waters of the Dutch Caribbean, but their movements, ecology and status remain poorly known. Given that these widely roaming pelagic shark species are amongst the most threatened species worldwide (Pacoureau et al., 2021), it is likely that the status of these species in the Dutch Caribbean is 'endangered' as well. For the deep-sea habitats in the Dutch Caribbean waters also hardly any data on occurrence of rays and sharks exist, but unlike the pelagic zone, the deep sea is far less affected by human stressors at present. It is therefore likely that, even though data-deficient, the current status of deep-sea ray and shark species in the Dutch Caribbean is much more favourable than for pelagic and reef habitats.

Minimum viable population size: a minimum viable population (MVP) means a 95% probability of survival over the next 100 years (Frankham et al., 2014; Traill et al., 2007). The MVPs for sharks and rays are currently unknown. Many of their populations, especially fished populations, are assessed based on stock statistics (e.g., biomass required for maximum sustainable yield; Simpfendorfer and Dulvy, 2017). However, for the Caribbean Netherlands or the wider region, no assessment of either the MVPs or any indicators for stock status are known. Also underlying ranges, connectivity between (sub)populations and population structures are still unknown. Points to consider about population sizes and indicators:

1. Some reef-associated species show high site fidelity and small home ranges (e.g., Caribbean reef sharks). Individual sharks may therefore be protected by local conservation measures/zones/MPAs. However, for the part of the population that moves over longer distances or resides outside protected zones, a network of protected zones or large protected areas are needed to sustain viable populations.
2. The presence of many pelagic and deep-sea species is currently unknown for the islands. Although some basic information of the more common, reef-associated and pelagic shark species is present, for the majority of shark and ray species there is no information about their absolute abundance or population sizes.

Present Distribution and Reference Values

The windward and leeward islands share the majority of shark and ray species, but based on distribution maps the diversity of rays in the waters of the leeward islands is expected to be higher (Table 1). This may be due to their proximity to South American countries and the influence of more coastal and estuarine ecosystems (e.g. large soft-bottom flat habitats). Many of these species have a distribution that ranges throughout the Caribbean Sea, or even a more global distribution (e.g., oceanic manta or hammerhead sharks). Only 13% (n = 12) of species included in Table 1 can be considered regionally endemic, mostly to the region of the windward islands and these are predominantly deep-sea species. However, it should be noted that this endemism is based on the distribution of these deep-sea species, rather than actual observations in these largely unstudied waters.

The waters surrounding the Dutch Caribbean islands offer a variety of coastal reef, coral, deep-sea and pelagic habitats. However, current knowledge and lack of data for most of the elasmobranch species in the area will not allow us to determine present distribution or establish reference values. This is also hampered by the lack of knowledge on historical baselines for occurrence before most of the anthropogenic impact started, although current population levels of sharks in the Caribbean Sea are considered only a fraction of what was present under former more natural conditions (Ward-Paige et al., 2010). This applies especially to reef-associated and pelagic elasmobranch species.

Assessment of National Conservation State

Trends in the Caribbean Netherlands (and Dutch Caribbean)
No temporal data on abundance exists for shark and ray species, except for temporal information over the past decade on Caribbean reef sharks and nurse sharks, which show an increase in observations (Leurs et al., in prep.). To enable population status assessments and the collection of both fisheries-dependent and -independent shark/ray data are needed. Based on the deteriorating global status of coastal and pelagic shark and ray species (Dulvy et al., 2021; Sherman et al., 2023; Pacoureau et al., 2021), the (shark) fisheries over the past decades in Dutch Caribbean waters (e.g., van Beek et al., 2012), and the continued degradation of their vital habitats, it can be assumed that shark and ray abundance and diversity is low compared to historical baselines (Ward-Paige et al., 2010).

Recent developments:

- Following an inventory on knowledge regarding elasmobranchs for areas of relevance to the Netherlands in 2012 as commissioned by the ministry of LNV (Overzee et al., 2012), in 2014, Wageningen Marine Research (then still named IMARES) was commissioned to prepare a Caribbean Netherlands shark protection plan (Beek et al., 2014). This was followed the next year by establishment of the Yarari Marine Mammal and Shark Sanctuary by ministerial decree. Prior to this
  Saba, Bonaire and Sint-Eustatius each had their entire or part of their shallow coastal waters protected as multi-use marine parks but, apart from Bonaire, lacking any special protection for sharks. In light of the ecosystem value of sharks and their value as charismatic marine creatures of importance to dive tourism, Bonaire led the way for shark protection by establishing sharks as protected species in their territorial waters as of 2008 (Island Ordinance Nature Management Bonaire (AB 2008, No. 23). Since 2015 the waters of Saba (including the Saba Bank) and Bonaire are part of the Yarari Marine Mammal and Shark Sanctuary. This sanctuary was extended to the entire waters of Sint-Eustatius in 2018. The goal of the sanctuary is a stricter protection of marine mammals, sharks and rays. The prohibition of the catch, transport and landing of sharks and rays in these waters has been included in the Fishing Decree BES since 2023 (Dutch Ministry of Agriculture, Fisheries, Food Security and Nature; DCNA, 2019). However, in 2008 in Bonaire, all shark species, the spotted eagle ray (Aetobatus narinari), Southern stingray (Hypanus americanus), and the oceanic manta ray (Mobula birostris) already gained protection under the Eilandsbesluit Natuurbeheer Bonaire.

- Fisheries. Anecdotal information suggests that pelagic fisheries with the use of fish aggregating devices (FADs) is increasing and that more FADs are deployed without legal embedding or management (Debrot et al., 2022). This increased pelagic fishery may increase the interactions of pelagic shark species with fisheries.

- Sharks and rays on international conventions.
  Multiple rays and sharks occurring in the Dutch Caribbean are listed under international and regional conventions to safeguard their conservation and sustainable trade (Table 1). The Convention on the Conservation of Migratory Species (CMS) has so far listed species like mobulid rays (2012), oceanic whitetip shark (2020), silky shark (2015), shortfin mako shark (2009), and hammerhead sharks (2015). Range states should ensure the conservation of these species and recognize their cross-boundary, long-distance movements. The SPAW (Specially Protected Areas and Wildlife) protocol from 1990 is a regional agreement between Caribbean nations for the protection and sustainable use of coastal and marine biodiversity. Shark and ray species listed include the large-tooth sawfish (2019), silky shark (2019), whale shark (2017), hammerhead sharks (2017), and manta rays (2017). More recently many shark and ray species have been listed on Appendix II of the Convention on International Trade in Endangered Species (CITES), which requires parties to control the trade of listed species. For sharks and rays this was mainly aimed at urging countries to regulate the trade of shark and ray meat and/or fins. To prevent misidentification issues, recently in 2023 all Carcharhinidae species were listed on Appendix II in addition to species like hammerhead sharks (2013), mobulid rays (2013), and thresher sharks (2017) that were already listed. This means that any trade in these species originating from the BES-island waters needs to be regulated and requires appropriate CITES documents.

Assessment aspects of natural area of distribution: Unknown (coastal, pelagic, deep sea)
There is currently no temporal information on range restriction and/or loss for any of the shark and ray species within the EEZ of the BES-islands. However, indications on coastal developments and the status of coral reefs under climate change indicate a deteriorating habitat quality for reef-associated species or species that associate with mangroves during part of their lifecycle. Historically all shark and ray species were likely (much) more common in these waters, including species which have now largely disappeared from Dutch Caribbean waters (e.g., oceanic whitetip shark).

Assessment aspect population: Unfavourable-inadequate (coastal), Unfavourable-bad (pelagic), Unknown (deep sea)

Coastal: shark and ray populations may benefit from the presence of MPAs and zones for recreational dive tourism, which can offer a degree of protection for resident (non-migratory) coastal species (Leurs et al., in prep.). However, they remain vulnerable to habitat degradation, particularly the loss of mangroves due to coastal developments and the loss of coral reefs due to developments and climate-triggered bleaching events. While international conservation assessments for these species remain concerning, the MPAs in the Dutch Caribbean could provide essential protection for (the non-migratory life stages of) these species, assuming that key habitats are preserved. In some of the Dutch Caribbean islands reef-associated sharks such as Caribbean reef shark or nurse shark may occasionally be taken in local fisheries, but particularly large numbers of juvenile nurse sharks (1,712 – 2,499 per year; between three and six sharks per trip) are taken as by-catch in Saba Bank lobster traps Graaf et al. (2017). Intermittent yet considerable effort has been dedicated to developing possible exclusion devices on the traps but so far these efforts have remained inconclusive (Debrot et al., 2022).

Pelagic: sharks and rays are threatened by (commercial and recreational) fishing along their migratory routes. Global population estimates of pelagic shark and ray species indicate serious declines (up to 71% since 1970s, Pacoureau et al., 2021). Data on pelagic species and their population size within the Dutch Caribbean is lacking, but anecdotal evidence suggests that these species are now less abundant compared to historical levels and are occasionally taken in pelagic fisheries near fish aggregating devices (e.g., silky sharks in Curaçao).

Deep sea: the population status of deep-sea sharks and rays is largely unknown, as all basic information for correct assessments of these species is lacking both globally and in Dutch waters. While many deep-sea species are shielded from threats such as large-scale fisheries and degradation of habitat, certain smaller scale fisheries (commercial and recreational) may catch shark species as bycatch. Overall, deep-sea species are classified as data deficient for the Dutch Caribbean as their presence has not been confirmed, their life history is unknown and so is their population status. Given the relatively low impact of anthropogenic activities in deep sea habitats at present, their status is likely to be favourable at present.

Assessment aspect habitat: Unfavourable-bad (coastal), Unfavourable-inadequate (pelagic), Favourable (deep sea)

Coastal: although coastal sharks may find refuge in coastal MPAs, their habitat may be impacted by coastal development and climate change. With a growing tourism sector and the reliance of island states on this sector, and the continued impact of rising seawater temperatures on coral reefs, it is expected that the habitat quality for these species will further deteriorate.

Pelagic: although pelagic sharks are not (or less) impacted by coastal development, their habitat is expected to be impacted by warming sea surface temperatures and associated increase of oxygen minimum zones. In addition, the increase of (legal and illegal) fish aggregating devices in the Dutch Caribbean waters may increase the interactions sharks and fishers, increasing the vulnerability of these species to fishing-related mortality.

Deep sea: sharks and rays in the deep sea within the Dutch Caribbean are only impacted by deep sea fishing, but both commercial and recreational deep sea fishing is not widespread and sharks and rays are not taken as target species in these fisheries. Other threats to deep sea habitats such as mining are non-existent. Deep sea habitats in the Dutch Caribbean are not well known and so is their importance to elasmobranchs, and their value for conservation.

Assessment aspect future prospects: Unfavourable-bad (coastal, pelagic), Unknown (deep sea)

Many aspects of shark and ray populations and their ecology remain unknown or in unfavourable conditions. Coastal sharks and rays are directly impacted by ongoing coastal developments and associated degradation or destruction of their habitat (e.g., coral reefs, mangrove nursery areas). Fishing may continue on these species in some of the islands and their global Conservation State has deteriorated over the past decade (Dulvy et al., 2021). The main threats for pelagic species come from the increasing use of fish aggregating devices, which may increase fisher-shark interactions. These species move over long distances and are therefore also negatively impacted by threats (e.g., fisheries, climate-related changes) along their migratory routes. The global Conservation State has also deteriorated significantly over the past decades. The threats for deep sea species are less known. Their presence in Dutch Caribbean waters are often unconfirmed or unknown, and the status of their critical habitat is also unknown, making an assessment of this species impossible until more information from Dutch Caribbean waters is available.

Table 2. *Diagnostic scores for the four different State of Nature criteria for coastal, pelagic and deep-sea sharks and rays, as well as well as an overall conservation assessment for the year 2024.*

	2024		
Aspect	**Coastal (reefs/demersal/benthopelagic)**	**Pelagic**	**Deep sea**
Distribution	Unknown	Unknown	Unknown
Population	Unfavourable-inadequate	Unfavourable-bad	Unknown
Habitat	Unfavourable-bad	Unfavourable-inadequate	Favourable
Future prospects	Unfavourable-bad	Unfavourable-bad	Unknown
Overall Assessment of Conservation State	**Unfavourable-bad**	**Unfavourable-bad**	Unknown

Comparison to the 2018 State of Nature Report

This is the first CS assessment made for the sharks and rays of the Caribbean Netherlands and hence no comparison can be made to any earlier report.

Recommendations for National Conservation Objectives

a. Improve data collection on sharks and rays to effectively assess their Conservation State, especially temporal abundance information, and information on pelagic and deep-sea species. This includes information on their current abundance, movements within the wider Caribbean region and their role in safeguarding marine ecosystem resilience as predators.
b. Determine crucial habitats for sharks and rays for different phases of their life cycle, e.g. pupping and nursery areas, mating grounds, feeding habitats, and safeguard their extent and quality.
c. Further develop conservation measures for sharks and rays under the Yarari Sanctuary and determine if existing legislation (e.g., shark fishing bans) are effective. This could include monitoring of fisheries catches or mark-recapture programs for elasmobranchs captured in commercial/recreational fisheries.
d. Collect more local (historical) ecological knowledge on shark and ray abundance and distribution from the fishing communities, as well as addressing the possible socio-economic importance of sharks and rays to the island communities, e.g. importance to dive tourism.

e. Determine the risk of fishing mortality for coastal, pelagic and deep-sea shark and ray species, especially under changing fisheries and the possibly increasing use of fish aggregating devices that may increase fisher-shark interactions (e.g., depredation by sharks).

Key Threats and Management Implications

The major threats to sharks and rays.

a. Loss or degradation of habitat: coastal development, climate change and nutrient runoff have caused essential shark and ray habitats to degrade. Degradation of (deep-sea) habitats needs to be halted or prevented and their importance to different life stages of shark and ray species should be further studied for an adequate and effective science-based management.
b. Interaction with fisheries throughout their range: although targeted shark and ray fisheries within the Dutch Caribbean may be limited, the extent and risk of fishing related mortality such as bycatch or illegal fisheries remains unclear. Depredation of catches by sharks may increase negative interactions and increase in fish aggregating devices may increase the risk of fishing mortality for pelagic shark species, and the long-distance movements of many elasmobranch species means these species may risk fishing mortality elsewhere along their migratory routes (e.g., outside MPAs or Dutch Caribbean waters). For this, international cooperation and establish measures on large scale, e.g. in the pelagic realms, or within networks of protected zones, e.g. reef and mangrove systems.

To address these threats, we advocate to: invest in reducing data deficiency: currently information for the status assessment is still largely missing for shark and ray species; create more awareness on the vital role sharks and rays play in the waters of the Dutch Caribbean, both in ecological and potentially also in socio-economic ways; increase international cooperation between the Caribbean countries and join efforts in conservation or mitigation of anthropogenic impacts, e.g. in fisheries management, preventing illegal fisheries, further strengthen networks of MPAs, enlarging local resilience to global climate change.

Data Quality and Completeness

Data on shark and ray populations and threats and status are missing for all shark and ray species in the Dutch Caribbean waters, which hampers the assessment of their status. Although for the BES-islands relatively more is known about sharks and rays compared to Aruba, Sint-Maarten and Curaçao, vital information is missing from all these islands. This is especially the case for pelagic and deep-sea shark and ray species, as for many basic information such as their occurrence and impact of threats are insufficiently known.

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24 Conservation State of the Deep-water Fish Fauna of the Dutch ABC-islands

Debrot, A. O., Robertson, D. R., Baldwin, C., van der Wal, J. T. and Vermeij, M. J. A. 2025. From: State of Nature Report for the Caribbean Netherlands, 2024. WUR report C001/25.

Status

With the term "deepwater" fish we here refer to fish species that have been documented at depths of 80 m or more. The literature providing documented deepwater fish records for the leeward Dutch islands (Aruba, Bonaire, Curaçao; ABC islands) is quite scant. Most studies are fairly recent and have been directed towards the description of the many small, new and long-overlooked species that the area is rich in. Table 1 provides a list of documented and expert-verified deep-water fish species records, for which island(s) they have been documented, their local documented depth (range), and their geographical distribution. We use six terms to classify the fish according to their geographical distribution. In the literature these terms are often used loosely by the many authors but for our assessment we more narrowly define the following seven-tiered geographic distributional terms as follows:

1) **Inner Caribbean**: those parts of the inner Caribbean close to continental South America or Central America.
2) **Greater Caribbean:** including the Gulf of Mexico, Bermuda the Bahamas and extending to the Guianas and Florida. N the literature this area is typically referred to as the West Central Atlantic.
3) **Western Atlantic:** extending from the Greater Caribbean either northward beyond Florida and/or southwards from the Guyana's.
4) **Eastern and Western Atlantic**
5) **Circumtropical:** worldwide tropics and subtropical
6) **Circumglobal:** worldwide temperate and tropical but also extending into colder and polar latitudes
7) **Unknown:** too few samples to say anything

Many species of principally deepwater fish can occasionally occur in shallow water or may use shallow waters during parts of their life cycle (such as the deepwater snapper *Lutjanus vivanus* of which juveniles have been documented in 2 m of depth along the beach in Curaçao (Pors and Debrot, unpubl. data). Also, many shallow water species may occasionally be seen at greater depths. In this inventory, all records of fish confirmed at depths of 80 m or more were included, regardless of their known or presumed depth preferences.

Based on this effort, we here present a list of 144 confirmed species and 16 unique yet unidentified species-level taxa for depths of >80m based on 4,642 expert sightings or studied specimens (i.e., 160 species-level taxa). This adds 89 taxa to the list of 71 species-level taxa provided by Baldwin et al. (2018a), and of which 67 were described species and four were unique species-level taxa yet undescribed. In contrast to Baldwin et al. (2018a), we did not include any species that were not recorded at 80 m or deeper which corresponds to the upper boundary of the lower mesophotic zone (Baldwin et al., 2018a). Therefore, four species included by Baldwin et al. (2018a), were not included in the present list (*Chromis cyanea, C. multilineata, Haemulon flavolineatum* and *Stegastus partitus*).

This list is based on four principal sources. The first is the fishes caught by local fishermen throughout the years and brought to our attention for identification. Most of these identifications were done by resident biologists (A. Debrot, I. Kristensen, I. Nagelkerken, L. Pors and M. Vermeij) of the Carmabi Foundation and relying principally on FAO fish identification sheets (Fischer, 1978). The second was a list of deepwater fishes observed and videoed (but otherwise unpublished) from a series of 24 submersible dives made in May 2000 by Harbor Branch, FLA, USA, down to depths of 900 m, off Curaçao, Bonaire and Aruba (Reed and Pomponi, 2000). The third principal source was the collection of dives by the Smithsonian Institution falling under the "DROP" project using the Curasub submarine to explore the reefs of Curaçao and Bonaire to maximum depths of about 300 m (Baldwin et al., 2018a). Finally, the fourth key source of records was the scientific literature, most publications of which were led by the Smithsonian Institution and pertain to species identifications coming forth from the Smithsonian "DROP" project. All difficult fish records were submitted to various specialists for identification so far as they deemed was possible.

Previously, Baldwin et al. (2018a) compiled a most extensive inventory of deepwater fish records for the island of Curaçao (and the southern Caribbean). They documented fish occurrence records for that island from depths of 40 meters and deeper and yielded a list of 71 species-level taxa (which then included four yet-unnamed species) based on 4,436 depth observations (Baldwin et al., 2018a). We here build on that initiative by compiling and listing 208 additional old as well as new previously published and unpublished deepwater fish records (ie: 4,642 records total) for an additional confirmation of 89 species-level taxa, of which 35 are added to the deepwater fish species list of records for Curaçao (and in some cases Bonaire or Aruba as well). However, 54 of these are, so far, exclusively listed as deepwater records for Bonaire and/or Aruba. Additional unconfirmed records suggest the presence of many more species but that would require better photographs or even specimens for detailed study and description.

Characteristics

Description:
Whereas a few papers assess the abyssal fish fauna (> 2000 m) of the Caribbean and adjacent tropical Atlantic (Anderson et al., 1987; Merrett & Fasham, 1998), fishes of the deep coral reef and upper continental and island slopes remain among the poorest-known marine faunas of the Caribbean (Williams & Williams, 2004; Garcia-Saes, 2010, Pinheiro et al., 2016; Baldwin et al., 2018a). Several authors have recently stressed the likely ecological and faunistic value of deep continental and island-slope habitats. Such deep habitats may function as key refugia for fish populations (e.g., Lesser et al. 2009), may possess unique coral assemblages (e.g. Meesters et al., 2013; Vermeij et al., 2003; Rocha et al., 2018) and many as yet undescribed species (e.g. Baldwin & Robertson, 2013; Baldwin et al., 2018b). Shelf-edge reefs have been highlighted as a priority for conservation of reef fish diversity in the tropical Atlantic (Olavo et al., 2011) but very few marine protected areas include such habitat.

For the description of new species, it is essential to be able to collect specimens and the ability to efficiently collect targeted specimens has been limited to the smallest species. Consequently, most new knowledge of the deepwater fishes has been developed for small collectable species (particularly gobies and serranids). The larger more vagile species, which include the commercially interesting species like deepwater snappers, groupers, tilefish and several others (like pomfrets and the swordfish) have hardly been documented in the literature even though they are either commonly caught by fishermen and are either casually or else with great likelihood known to occur

Table 1. *Fish occurrence records for Aruba (A), Bonaire (B) and Curaçao (C) documented from depths of 80 meters and deeper, for 144 confirmed species and 16 unique yet unidentified species-level taxa based on 4,642 expert sightings or studied specimens. For all but the most-recently described or reassigned species we used the ECoF (Fricke et al., 2025) list of accepted scientific names and American Fisheries Society list of accepted common names (Page et al., 2023).*

Species name	Common name (English / Papiamento)	Documented range	Island occurrence Aruba (A), Bonaire (B) Curaçao (C)	Depths (m.)	Number of records
ACROPOMATIDAE					
Synagrops bellus	Blackmouth Bass	Atlantic & Western Pacific	B, C	100-422	3
Verilus sordidus	(Stèlchi di hundu)	Western Atlantic	B, C	100	2
ANTHIADIDAE					
Anthias asperilinguis		Western Atlantic	B, C	96-297	20
Baldwinella cf vivanus		Unknown	B, C	125-232	62
Bathyanthias sp.		Unknown	B	213-216	2
Choranthias tenuis	Threadnose Bass	Western Atlantic	B, C	90-301	25
Hemanthias leptus	Longtail Bass	Western Atlantic	B, C	114-183	4
Plectranthias garrupellus	Apricot Bass	Greater Caribbean	B	126	1
Pronotogrammus martinicensis	Roughtoungue Bass	Western Atlantic	A, B, C	85-293	571
APOGONIDAE					
Apogon gouldi	Deepwater Cardinalfish	Greater Caribbean	B, C	102-157	20
Apogon pillionatus	Broadsaddle Cardinalfish	Western Atlantic	B	122-140	1
Apogon pseudomaculatus	Twospot Cardinalfish	Western Atlantic	B	114	2
Paroncheilus affinis	Bigtooth Cardinalfish	Eastern & Western Central Atlantic	B, C	90-154	9
AULOPIDAE					
Aulopus filamentosus	Yellowfin Aulopus	Eastern & Western Atl.	C	100-150	1
Saurida normani	Shortjaw Lizardfish	Western Atlantic	B	200-250	1
BALISTIDAE					
Xanthichthys ringens	Sargassum triggerfish (Pishi porko shinishi)	Western Atlantic	B, C	49-105	17
BATHYCLUPEIDAE					
Neobathyclupea cf. argentea	Silver Deep-sea Herring	Western Atlantic	C	633-665	2
BRAMIDAE					
Eumegistus brevorti	Tropical Pomfret (Pamper di hundu)	Western Atlantic	B, C	300-650	4
BYTHIDAE					
Stygnobrotula latebricola	Black Brotula	Western Atlantic	B	114-137	1
CALLIONYMIDAE					
Synchiropus agassizii	Spotfin Dragonet	Western Atlantic	C	236-304	7
CAPROIDAE					
Antigonia capros	Deepbody Boarfish	Circumtropical	B, C	100-299	84
Antigonia sp.	Boarfish sp.	Greater Caribbean	C	NA	1
CARANGIDAE					
Caranx bartholomaei	Yellow Jack (Sareu)	Western Atlantic	B	93	1
Caranx lugubris	Black Jack (Korkobá pretu)	Circumtropical	A, C	<144	3
Seriola rivoliana	Almaco Jack (Kabijou)	Circumglobal	B, C	115-190	6
CETORHINIDAE					
Cetorhinus maximus	Basking Shark	Cosmopolitan	A	0	2
CHAETODONTIDAE					
Chaetodon capistratus	Foureye Butterflyfish	Western Atlantic	C	<40-101	17
Chaetodon sedentarius	Reef Butterflyfish	Western Atlantic	B	85-112	3
Chaetodon striatus	Banded Butterflyfish (Makamba marinir)	Western Atlantic	B	84	1
Prognathodes aculeatus	Longsnout Butterflyfish	Greater Caribbean	B, C	88-149	5

Prognathodes guyanensis	Guyana Butterflyfish	Western Atlantic	B, C	88-216	32
CHAUNACIDAE					
Chaunax cf suttkusi	Pale-cavity Gaper	Eastern & Western Atl.	C	610	1
Chaunax pictus	Uniform Gaper	Eastern & Western Atl.	C	247-307	6
CONGRIDAE					
Conger triporiceps	Manytooth conger (Conglá)	Western Atlantic	C	120-150	0
DALATIIDAE					
Scyliorhinus hesperius	White-saddled Cat Shark	Greater Caribbean	B, C	200-220	4
EPIGONIDAE					
Epigonus gemma		Southern Caribbean	C	156-309	5
Epigonus hexacanthus		Southern Caribbean	C	156-297	8
Epigonus sp.	Deepwater cardinalfish sp.		C	357-369	3
Sphyraenops bairdianus	Triplespine Deepwater Cardinalfish	Worldwide temperate & tropical	C	265-280	6
EPINEPHELIDAE					
Cephalopholis cruentata	Grasby (Purunchi)	Western Atlantic	B, C	<40-135	37
Epinephelus morio	Red Grouper (Meru)	Western Atlantic	A	3-140	1
Gonioplectrus hispanus	Spanish Flag (Bandera spañó)	Western Atlantic	B, C	101-224	58
Hyporthodus niveatus	Snowy Grouper (Djampou)	Western Atlantic	B	175-300	5
Mycteroperca bonaci	Black Grouper (Djampou)	Western Atlantic	C, B	91	2
Paranthias furcifer	Atlantic Creolefish (Mahawa, Stèlchi)	Eastern & Western Atl.	C, B	<40-110	248
FISTULARIIDAE					
Fistularia petimba	Red Cornetfish	Worldwide temperate & tropical	C	263	1
Fistularia sp.	Cornetfish sp.		B	175-178	2
GEMPYLIDAE					
Epinnula magistralis	Domine	Greater Caribbean	A	NA	1
Gempylus serpens	Snake Mackerel	Worldwide subtropical & tropical	C	8	1
Nesiarchus nasutus	Black Gemfish	Worldwide subtropical & tropical	C	NA	1
Promethichthys prometheus	Roudi Escolar (Kaka leu leu)	Worldwide tropical & subtropical	C	120-150	1
Ruvettus pretiosus	Oilfish (Kaka sin sinti)	Worldwide tropical & subtropical	B, C	100-150	3
GOBIIDAE					
Antilligobius nikkiae	Sabre Goby	Caribbean	B, C	82-205	259
Bollmannia eigenmannorum	Shelf Goby	Greater Caribbean	B	84	1
Coryphopterus curasub	Yellow-spotted Sand-goby	Southern Caribbean	B, C	70-163	7
Palatogobius incendius	Ember Goby	Greater Caribbean	B, C	114-205	326
Priolepis hipoliti	Rusty Goby	Greater Caribbean	B	115-117	2
Psilotris laurae	Thin-barred Goby	Southern Caribbean	B	121-162	3
Psilotris vantasselli	Clementine Split-fin Goby	Greater Caribbean	B	149-159	1
Ptereleotris helenae	Hovering Dartfish	Western Atlantic	B, C	<40-127	29
Robinsichthys nigrimarginatus	Black-margined Goby	Southern Caribbean	C	229	0
Undescribed goby	Goby sp.	Southern Caribbean	B	142-158	3
Varicus cephalocellatus	Ocellated Split-fin Goby	Southern Caribbean	B	121-305	4
Varicus decorum	Decorated Split-fin Goby	Southern Caribbean	B, C	159-164	4
Varicus lacerta	Godzilla Goby	Southern Caribbean	C	129-143	6
Varicus veliguttatus	Spotted sail-Goby	Greater Caribbean	C	152-225	1
GRAMMATIDAE					

Lipogramma barrettorum	Blue-spotted Basslet	Southern Caribbean	C	123-161	6
Lipogramma evides	Banded Basslet	Greater Caribbean	B, C	124-265	59
Lipogramma haberorum	Yellow-banded Basslet	Southern Caribbean	C	152-233	3
Lipogramma klayi	Bicolor Basslet	Greater Caribbean	B, C	79-114	42
Lipogramma levinsoni	Hourglass Basslet	Greater Caribbean	B, C	111-153	13
Lipogramma schrieri	Maori Basslet	Southern Caribbean	C	173-207	6
GRAMMICOLEPIDIDAE					
Grammicolepis brachiusculus	Thorny Tinselfish	Worldwide subtropical & tropical	B	463-650	2
GRAMMISTIDAE					
Jeboehlkia gladifer	Bladefin Basslet	Western Caribbean	B, C	125-203	57
Rypticus cf. randalli	Plain Soapfish	Western Atlantic	B	84	1
Rypticus saponaceus	Greater Soapfish, Habon	Eastern & Western Atl.	B	114	1
HAEMULIDAE					
Haemulon striatum	Striped Grunt	Western Atlantic	B, C	53-107	34
Haemulon vittatum	Boga, Traki traki	Western Atlantic	C	<40-105	17
HEXANCHIDAE					
Hexanchus griseus	Bluntnose Sixgill Shark	Eastern & Western Atl.	B, C	>100	2
Hexanchus vitulus	Atlantic Sixgill Shark (Tribon brabu)	Circumglobal	B, C	100-350	3
HOLOCENTRIDAE					
Corniger spinosus	Spineycheek Soldierfish	Western Atlantic	C	137-238	11
Myripristis jacobus	Blackbar Soldierfish (Barí di klabu)	Eastern & Western Atl.	C	<40-83	13
Neoniphon marianus	Longjaw Squirrelfish (Korá kandèl)	Greater Caribbean	C	53-101	37
Ostichthys trachypoma	Bigeye Soldierfish	Western Atlantic	B, C	100-287	105
IPNOPIDAE					
Bathypterois spec.	Deep-sea tripod fish sp.		B	916	2
LABRIDAE					
Bodianus parrae	Creole Wrasse	Greater Caribbean	C	<40-95	287
Decodon puellaris	Red Hogfish (Pewchi di hundu)	Western Atlantic	B, C	87-231	12
Decodon sp2			B	116-161	3
Halichoeres bathyphilus	Greenband Wrasse	Western Atlantic	B	116	1
Polylepion gilmorei	Red-barred Wrasse	Western Atlantic	C	219-272	4
LABRISOMIDAE					
Haptoclinus dropi	Four-fin Blenny	Southern Caribbean	C	157-167	2
Starksia sp.			B	84-226	1
LATILIDAE					
Caulolatilus cyanops	Blackline Tilefish (Tumba)	Western Atlantic	B	200-350	3
Caulolatilus dooleyi	Bankslope Tilefish	Greater Caribbean	B, C	200-350	3
Caulolatilus guppyi	Reticulated Tilefish	Southern Caribbean	A	90	1
Caulolatilus williamsi	Yellowbar Tilefish (Donseo)	Greater Caribbean	B, C	200-350	4
LIOPROPOMATIDAE					
Liopropoma aberrans	Eyestripe Basslet	Greater Caribbean	B, C	98-241	39
Liopropoma carmabi	Candy Basslet (Carmabivis)	Greater Caribbean	B	84	1
Liopropoma mowbrayi	Cave Basslet	Greater Caribbean	B, C	56-133	112
Liopropoma olneyi	Yellow-spotted Basslet	Greater Caribbean	B, C	110-229	83
Liopropoma santi	Spot-tail golden Basslet	Southern Caribbean	C	182-209	3
LUTJANIDAE					
Etelis oculatus	Queen Snapper (Sabonèchi)	Western Atlantic	B, C	200-500	6
Lutjanus apodus	Schoolmaster (Bèrs)	Western Atlantic	B	146	1

Lutjanus buccanella	Blackfin Snapper (Korá hala pretu)	Western Atlantic	B, C	91-163	4
Lutjanus purpureus	Caribbean Red Snapper (Korá)	Western Atlantic	C	>100	1
Lutjanus vivanus	Silk Snapper (Shiriki)	Western Atlantic	C	>100	1
Ocyurus chrysurus	Yellowtail Snapper (Grastèlchi di piedra)	Western Atlantic	B	95-116	1
Pristipomoides aquilonaris	Wrenchman (Rondu)	Western Atlantic	B	216	1
Pristipomoides freemani	Slender Wrenchman	Western Atlantic	B	197	1
Pristipomoides macrophthalmus	Cardinal Snapper	Western Atlantic	C	210-290	1
MACROURIDAE					
Coryphaenoides sp.	Grenadier sp.		C	711	1
Macrouridae sp.	Grenadier sp.		B	765	1
Malacocephalus cf. laevis	Velvet Grenadier	Worldwide subtropical & tropical	C	740	1
Nezumia cf. aequalis	Atlantic Blacktip Grenadier	Eastern & Western Atl.	C	608	1
MORIDAE					
Physiculus fulvus	Metallic Codling	Western Atlantic	A, B	151-297	4
MURAENIDAE					
Gymnothorax maderensis	Sharktooth Moray (Kolebra)	Eastern & Western Atl.	A, B, C	109-302	5
Gymnothorax moringa	Spotted Moray (Kolebra)	Western Atlantic	B	113	1
Gymnothorax ocellatus	Ocellated Moray (Kolebra)	Western Atlantic	B	187-194	1
OGCOCEPHALIDAE					
Ogcocephalus parvus	Roughback Batfish (Palomba di awa)	Western Atlantic	B, C	100-161	2
PARALICHTHYIDAE					
Citharichthys dinoceros	Spined Whiff	Western Atlantic	B	185	1
PENTANCHIDAE					
Apristurus sp.	Deepwater Cat Shark sp.		B	100-919	3
Parmaturus sp.	Cat Shark	Western Atlantic	B	NA	1
PERCOPHIDAE					
Chrionema squamentum	Scalychin Flathead	Greater Caribbean	B, C	162-306	186
PERISTEDIIDAE					
Peristedion brevirostre	Flathead Armored Searobin (Sabu sèiskiel)	Greater Caribbean	B	180-300	4
Peristedion cf. imberbe	Tropical Armored Slender Searobin	Western Atlantic	C	NA	1
POLYMIXIIDAE					
Polymixia sp.	Beardfish sp.		B, C	300-380	3
POMACANTHIDAE					
Centropyge argi	Cherubfish	Greater Caribbean	B, C	63-99	42
Holacanthus ciliaris	Queen Angelfish	Western Atlantic	B	84	1
Pomacanthus paru	French Angelfish (Sheu)	Eastern & Western Atl.	B	91	1
POMACENTRIDAE					
Chromis cf. scotti	Purple Reeffish	Western Atlantic	B, C	49-101	85
Chromis insolata	Sunshinefish	Greater Caribbean	B, C	40-110	215
Chromis vanbebberae	Whitetail Reeffish	Western Atlantic	C	49-178	7
PRIACANTHIDAE					
Pristigenys alta	Short Bigeye	Western Atlantic	B	73-183	14
REGALECIDAE					
Regalecus glesne	Oarfish	Circumglobal	C	0	1
SCIAENIDAE					

Eques lanceolatus	Jackknife-fish (Rei di lamán)	Western Atlantic	B	100-160	7
SCOMBROPIDAE					
Scombrops oculatus	Gnome Fish	Greater Caribbean	C	NA	1
SCORPAENIDAE					
Pontinus castor	Longsnout Scorpionfish	Greater Caribbean	B, C	87-243	25
Pontinus longispinis	Longspine Scorpionfish	Western Atlantic	B, C	237-343	3
Pontinus nematophthalmus	Spinythroat Scorpionfish	Western Atlantic	B, C	132-302	24
Pterois volitans	Lionfish	Pacific Ocean	B	85-182	9
Scorpaena agassizii	Longfin Scorpionfish	Western Atlantic	B	174-204	2
Scorpaenodes barrybrowni	Stellate Scorpionfish	Greater Caribbean	B, C	114-154	8
SERRANIDAE					
Bullisichthys caribbaeus	Pugnose Bass	Greater Caribbean	B, C	81-134	263
Serranus atrobranchus	Blackear Bass	Western Atlantic	B	122-140	1
Serranus chionaraia	Snow Bass	Greater Caribbean	B	83-95	3
Serranus fusculus	Twospot Seabass	Western Atlantic	C	48-245	13
Serranus luciopercanus	Crosshatch Bass	Greater Caribbean	B, C	61-129	72
Serranus notospilus	Saddle Bass	Greater Caribbean	B, C	107-234	111
Serranus phoebe	Tattler (Vrumoe)	Western Atlantic	B, C	83-186	35
Serranus tortugarum	Chalk Bass	Greater Caribbean	B	82	1
SQUALIDAE					
Squalus cubensis	Cuban Dogfish	Western Atlantic	C	293	2
STOMIIDAE					
Chauliodus sloani	Manylight Viperfish	Worldwide subtropical & tropical	C	>100	2
SYMPHSANODONTIDAE					
Symphysanodon berryi	Slope Bass	Eastern & Western Atl.c	C	126-296	117
Symphysanodon octoactinus	Insular Bunquelovely	Greater Caribbean	B, C	133-167	155
SYNODONTIDAE					
Synodontidae sp.	Lizardfish sp.		b	150-331	2
TETRAODONTIDAE					
Canthigaster jamestyleri	Goldface Toby	Western Atlantic	B, C	70-152	117
Canthigaster rostrata	Sharpnose Puffer	Western Atlantic	B	116	1
Sphoeroides pachygaster	Blunthead Puffer (Fugu)	Worldwide subtropical & tropical	B, C	200	2
TRACHICHTYIDAE					
Gephyroberyx darwinii	Big Roughy	Cosmopolitan	B, C	266-562	4
Hoplostethus mediterraneus	Silver Roughy	Cosmopolitan	C	360	1
Hoplostethus sp.	Roughy sp.		B, C	114-319	3
TRICANTHODIDAE					
Hollardia meadi	Spotted Spikefish	Western Atlantic	B	176-305	1
TRICHIURIDAE					
Evoxymetopon taeniatus	Channel Scabbardfish (Machete)	Worldwide subtropical & tropical	B	0	1
Trichiurus lepturus	Atlantic Cutlassfish (Cachicang, Guepi baraháns)	Worldwide subtropical & tropical	C	0	1
TRIGLIDAE					
Bellator brachychir	Shortfin Searobin	Western Atlantic	B, C	119-245	14
Bellator egretta	Streamer Searobin	Western Atlantic	B, C	176-232	8
Bellator sp.	Searobin sp.		B	150-220	1
XIPHIIDAE					

Xiphias gladius	Swordfish (Balaú salmou)	Worldwide subtropical & tropical	B	650	1

Isl.occ. = Island occurrence; n.recs = number of records.

The southern Caribbean has been demonstrated to be a separate biogeographic province for different animal groups among which are molluscs (Diaz, 1995) and fishes (Spalding, 2007; Robertson & Cramer, 2014). Consequently, many described species are principally or strictly limited to the southern and or western Caribbean. These species include both long-described shallow and recently described deepwater species. The fauna also may include several unique and possibly range-restricted elasmobranchs (e.g. *Apristurus* spp., Table 1) but obtaining actual specimens for detailed study is difficult. Consequently, this interesting component of the deep-water fish fauna remains very poorly known.

Of all taxa distinguished, something can be said about their known geographical distribution for 156 species (Fig. 1). Of these, 16 species (10%) appear to be endemic to a small inner portion of the Caribbean Sea, 36 (23%) appear endemic to the Greater Caribbean, while 67 (43%) are limited to the Western Atlantic. More widely distributed species, extending across the Atlantic and beyond amounted to 35 species (23%). Overall, the results show the strong regional affinities of the fish fauna and a high proportion of species with a highly range-restricted (i.e.. "endemic") distribution.

The geographic position of the Dutch leeward islands off the northeast coast of south America has remained unchanged since the upper Miocene (7-9 Ma) (Itturalde-Vinent, 2006). This placed these islands in a strategic upstream location with respect to current flows within the Caribbean and likely allowed them to play an exceptional role for dispersal of marine life throughout the region, even long before the isthmus of Panama fully closed during the Pliocene-Holocene epochs (3.7-0 Ma) (Iturralde-Vinent, 2006). Their location outside the main Western Atlantic hurricane zone, further reduces exposure to annual hurricane risk while their separation from the mainland of South America provides protection from continental stress factors of freshwater and sediment loads that can strongly limit reef development (Weil, 2003).

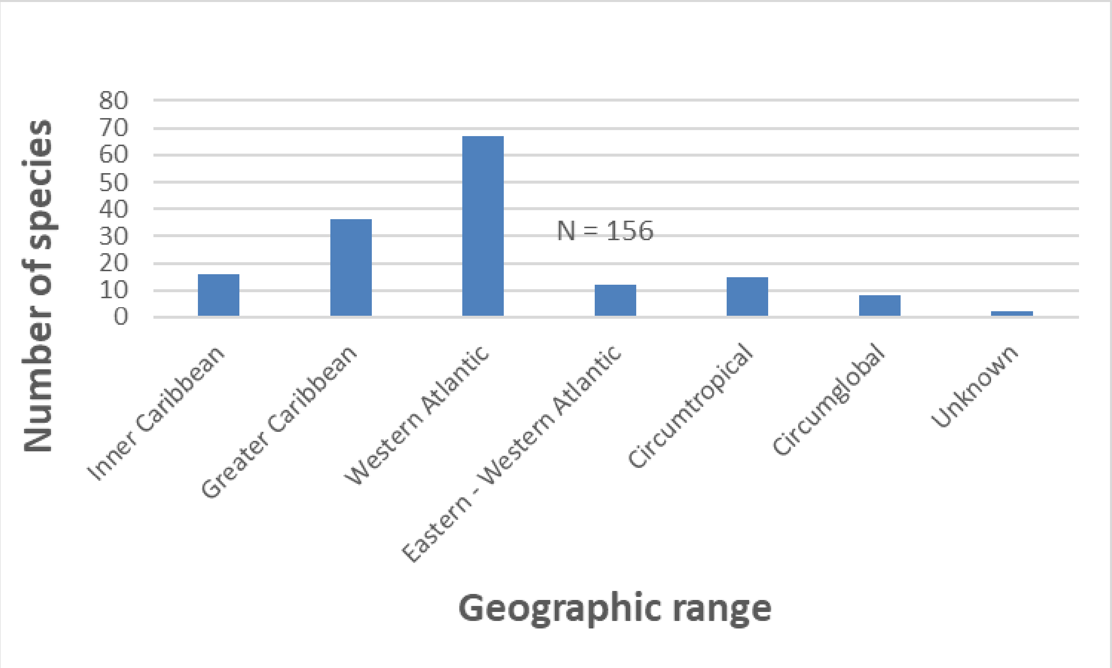

Figure 1. *Known geographical distribution of the fish species documented (to verified or almost certain species level "cf") at depths of 80 m or more around the leeward Dutch Caribbean islands of Aruba, Bonaire and Curaçao. Relative Importance within the Caribbean: High.*

Finally, their location close to the main upwelling zone of the Caribbean also appears to explain their unique protection from the worst impacts of regional sea-surface warming (Eakin et al., 2010), The long-lasting favourable conditions for reef growth likely underpin their role as a storehouse of coral reef-associated biodiversity that continues to be documented. Not surprisingly then, the southern

Caribbean is very interesting from a biogeographic perspective as it has been recognized as a distinct marine biogeographic province for fish diversity (Spalding 2007; Robertson & Cramer 2014) and more than a dozen new species have recently been described for these islands (Baldwin et al., 2016a; 2016b; 2018b; 2023; Baldwin & Robertson, 2013; 2014; 2015; Carvalho-Filho et al., 2023; McFarland et al., 2020; Okamoto et al., 2024; Tornabene and Baldwin, 2017; 2019; Tornabene et al., 2016a; 2016b; 2023).

Ecological aspects

Habitat: Baldwin et al. (2018a) distinguish large differences in deepwater fish faunas of the upper mesophotic (40-79 m), the lower mesophotic (80-129) and the upper rariphotic (130-189 m) and lower rariphotic (190-309m) zones. The strongest partition between these four different faunas occurs between the mesophotic and rariphotic zones and between the upper and lower rariphotic zones (Baldwin et al., 2018a). The main differences between these zones are defined by temperature (e.g.: thermocline), light (which influences photosynthesis) and food and energy availability, as at depths of beyond 200 m the input of food becomes limited (Woolley et al., 2016). Consequently, many mesopelagic fish and plankton species will migrate vertically on a diurnal basis to feed in the shallower food-rich waters typically at night (Benoit-Bird and Moline, 2021).

Food: Herbivorous fish species are absent in the deep mesophotic zone or deeper (80 m and beyond). Almost all species are either piscivores, benthic invertivores or planktivores, this includes the four deep-reef pomacentrids of the genus *Chromis* which are all planktivorous.

Minimum viable population size: Unknown
For fish species, minimum viable population sizes have not yet been meaningfully explored. This concept is best applied to terrestrial species but even for those there are many caveats and limitations (Traill et al., 2007; Ottburg and van Swaay, 2014).

Present Distribution and Reference Values

The distribution of fish species is here recorded in terms of their: a) documented depth ranges for these islands; b) documented island occurrences and also: c) the documented geographical range for each species. The documented depth ranges for many species for the islands is based on only few confirmed records. Therefore, many species can be expected to have broader depth ranges than this limited data set would suggest (and as is also likely from the depth distributions known for many of the same species from elsewhere). The most reliable local depth ranges are provided for several species (Baldwin et al., 2018a) based on much larger sample sizes. At present, the island occurrence of different species is seriously under-represented due to the low level of sampling, especially for Aruba. With more extensive and judicious observation, probably most species will eventually be able to be confirmed for all three islands. Finally, the most reliable information on distribution is the data on geographic distribution. The only exception would be the many newly described and possibly range-restricted small species (mainly gobies and serranids), which may be ultimately found to have a wider distribution once deepwater surveys from different parts of the Caribbean become more widely available.

Assessment of National Conservation State

Aside from the scant species records for most species, little is known about the local or international Conservation State of most species. However, four of the deepwater sharks documented from the ABC-islands have a IUCN Red List threatened classification. Thes are *Cetorhinus maximus* (Endangered), *Hexanchus griseus* and *H. vitulus* (Near Threatened) and *Parmaturus angelae* (Vulnerable). *Scylorhinus hesperius* and *Squalus cubensis* are classified as “Least Concern”.

In addition, the heavily fished swordfish, *Xiphius gladius*, and grouper *Mycteroperca bonaci* have a Near Threatened listing while the groupers *Epinephelus morio* and *Hyporthodus niveatus* are listed as Vulnerable by the IUCN. The status of all other species is either Least Concern or Unknown due to lack of information.

Trends in the Caribbean Netherlands: Unfavourable-inadequate
Trends for deepwater species in the Caribbean Netherlands are unknown, but for most of the small deepwater species no serious immediate threats are really known but this surely must be ascribed to the lack of research and not to the lack of serious yet unknown threats.

Recent developments: Description of many new deepwater species
Many new and possibly unique species have been recently described (Baldwin et al., 2016a, 2016b, 2018b; Baldwin and Robertson, 2013; 2014; 2015; McFarland et al., 2020; Tornabene and Baldwin, 2017; 2019; Tornabene et al., 2016a; 2016b; Okamoto et al., 2024). It remains to be seen how these species are distributed in the southern Caribbean or rest of the region. These mostly involve a whole number of small collectable serranids and gobies. However, there is considerable untapped potential for possibly unique range-restricted deep-water sharks as well (*Apristurus* spp., *Isistius* spp.) (Debrot et al., 2014a).

Assessment of distribution: Favourable
Most species records are for Curaçao and Bonaire while the least are for Aruba. This is likely largely based on the much greater sampling effort expended around Curaçao and Bonaire than around Aruba (or for that matter most elsewhere in the southern Caribbean). The distribution of individual species is likely more extensive than based on this limited sampling effort, and thus favourable.

Assessment of population: Favourable
Quantitative assessments at community or population levels are very problematic because quantitative fish surveys at depth are difficult to obtain. Such assessments will only become available once directed deepwater ROV transects or drop data become available. At present Wageningen Marine Science is working with drones to obtain quantitative insights into the population density, size-structure and relative distribution of commercially targeted fish stocks of the Saba Bank but such work has yet to commence for the fish stocks of Bonaire. The population sizes of most species around the leeward Dutch Caribbean are likely to be favourable as the habitats are intact and fishing pressure is likely low or zero on most species.

Assessment of habitat: Favourable
Some insight is available regarding the local depth distribution of each species. For most species little local knowledge is available on other aspects of habitat use or dependence.

Table 2. *Summary overview of the status of the deep-water fish fauna of the leeward Caribbean Netherlands in terms of different conservations aspects.*

Aspect deep water fish fauna	**2024**
Distribution	Favourable
Population size	Favourable
Habitat	Favourable
Future prospects	Unfavourable-inadequate
Overall Assessment of Conservation State	**Unfavourable-inadequate**

Comparison to the 2018 State of Nature report

This is the first CS assessment made for the deepwater fishes of the Leeward Caribbean Netherlands and hence no comparison can be made to any earlier report.

Goals for the national conservation objective

The goals as outlined in the management plan for the Exclusive Economic Zone of the Dutch Caribbean (Meesters et al., 2010) remain pertinent. These were to: a) assemble, review and assess existing literature and data on the Dutch Caribbean deep sea and adjacent areas; b) encourage the organization of a deep-sea expedition to collect, describe and document the biodiversity, possibly coupled with preliminary bioprospecting. Several initial steps in this have thus been taken. As part of action point b, in 2013 the Ministry of Economic Affairs financed a 2-day deep-reef submersible exploration in the coastal waters of Bonaire during which about 15 species likely new to science (none of which were fish) were collected (Becking & Meesters, 2014). In addition, some preliminary deep-water exploration has been done on the Saba Bank (van Duyl & Meesters, 2020; Humphreys et al., 2022).

Key threats and management implications

Unknown but likely worse than until recently expected
Loiseau et al. (2024) have most recently discovered that a much larger proportion of marine fish than expected (about 25%) should be regarded as seriously threatened. For the Caribbean Netherlands this remains unknown. While so far only for one endemic deep reef fish species have authors raised conservation concerns due to predation by an invasive species (Tornabene and Baldwin, 2017) the situation could well be alarming for many species. However, this remains unknown due to the lack of research. The mesophotic and rariphotic deepwater habitats in question do suffer high levels of litter pollution (Debrot et al., 2014b) but the impacts of litter pollution on deep-water fishes also have not been assessed.
For the larger commercially interesting species, several have been internationally overfished to the point at which their Conservation State has become of concern (particularly deepwater snappers, groupers and sharks). Several of these species may be overfished around the ABC-islands particularly due to the generally limited surface area of habitat available which means that overfishing can quickly occur. While overfishing of the shallower reef-associated fish stocks around these islands is well documented (Debrot and Criens, 2005; Debrot and de Graaf, 2018; Vermeij et al., 2019) the status of deepwater fish stocks is very poorly documented. More work is recommended to assess the potential of these stocks to support sustainable fisheries (Debrot et al., 2019). More generally it is known that deepwater fish species often are long-lived, with slow growth rates and can only support limited fishing pressure (Norse et al., 2012). Finally, the potential impact of climate change that may possibly affect water temperatures and oxygen concentrations at depth, as well as current patterns, must be kept in mind but are yet unknown.

Data quality and completeness

Apart from the seminal work by Metzelaar (1922), there are very few comprehensive treatments of the fish fauna of the Dutch ABC islands. The first major treatment of the groupers and snappers for the Dutch Caribbean is given by Nagelkerken (1981). In it he presents information on a total of 14 groupers and 12 snappers, most of which are shallow water species. Baldwin et al. (2018a) recently compiled an extensive inventory of deepwater fish records for the island of Curaçao (and the southern Caribbean) and Robertson et al. (2022) point out the crucial role that submersibles have and will need to play in developing more and better information on deepwater fish faunas.

Nevertheless, in general, very little is known about the local occurrence and ecology of deepwater fishes in the southern Caribbean and most deepwater species listed elsewhere for the Dutch Caribbean are not based on actual documented records but based on interpolation. The ABC-islands get included as “range” islands based on the combination of two main arguments. These are that: a) if a species is listed for location A and B, it is logical to infer that it also occurs at the points (or countries) in between. so long as b) suitable habitat is also present. Our purpose here was hence to compile documented and expert-verified records of deepwater fish occurrences for the ABC-islands as a first step towards a better understanding of the represented biodiversity and fisheries potential.

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25 Conservation State of the Fish Stocks of the Caribbean Netherlands

Debrot, A. O., Brunel, T. and Clements, O. N. 2025. From: State of Nature Report for the Caribbean Netherlands, 2024. WUR report C001/25.

Current status

Fishing has always been a tradition on the islands of the Caribbean Netherlands, and up until about fifty years ago, this fishery made a significant contribution to the protein supply of the island population. Since then, the import of fish and meat from abroad has increased, while local fisheries have declined. This decline is partly due to decreasing fish stocks. The primary measure of fish stock health is usually the scale of the fishery and the catches. It is based on the fact that catches and their development often reflect the fish stocks themselves, which are usually much more difficult to measure. Not only the total catch (C) is considered, but also the effort (E) required to achieve a certain catch. In fishery jargon, this is called Catch Per Unit Effort (CPUE). CPUE is an important measure of fish abundance. The total catch (C) is then CPUE x E. In most situations where abundance (CPUE) increases, fishermen will be inclined to fish more (increased effort). Typically, an increase in catches from a particular area is indicative not only of an increase in abundance but also of an increase in effort. Therefore, in the analysis of fish catches, it is crucial to measure both underlying factors and not just look at the realized catches.

Fish catches are, as it were, the "pulse" of the fish stock, the total of the fished stocks. Only since 2012 have catch data been regularly collected for the Saba Bank and St. Eustatius, and since then several analyses have become available (de Graaf et al., 2014; 2015; 2016; 2017; Brunel et al. 2018, 2020, 2021; Amelot et al., 2021; Domingues et al., 2024 in prep.). For Bonaire, only a one year-long inventory of the fishery with an estimate of the catches has been made so far (de Graaf et al., 2016).

In addition to fishery-dependent indicators of fish stocks, there are also fishery-independent indicators. Fisheries dependent and independent data serve different yet complementary purposes but typically fisheries independent data, collected by controlled research design is of better quality. For shallow coral reefs in clear water, this usually involves visually counting the number of fish in a certain area. This provides a measure of fish density per unit area. For the Caribbean Netherlands, there are several such studies that provide snapshots of the abundance status of mainly diurnal fish in parts of the shallow coral reef (Hawkins et al., 2007; Hylkema et al., 2014; Klomp and Kooistra, 2003; Steneck and Arnold 2013, Steneck et al. 2011; 2013; 2015; Steneck and McClanahan, 2005; Kuik et al., 2014; Looiengoed, 2013; Luckhurst and Luckhurst, 1978; McLellan, 2009; Meesters et al., 1996; Nagelkerken et al. 2002; Pattengill-Semmens 2002; Roberts 1995; Roberts and Hawkins, 1995; Sybesma et al., 1993; Sandin et al., 2008; Toller et al., 2010; Vlugt, 2016). In such visual counts, nocturnal fish (those that hide during the day), which are otherwise caught in traps and nets or during night fishing with hand lines, are often largely overlooked.

Conservation State: Despite the current importance of the reef fish stock for both dive tourism and fisheries, large predatory fish species have drastically declined in number over the past decades to the point where they are hardly seen by divers and now form a negligible part of catches. Many of these species (see Table 1) are currently listed by the IUCN as vulnerable and/or endangered. This also applies to many of the sharks in the Caribbean Netherlands, which are extensively discussed elsewhere (but not in this report) (Beek et al., 2012; 2014; Overzee et al., 2012). All species in the table below can be considered important "target species" for consumption except for the Tarpon, Bonefish, and Rainbow Parrotfish. With the disappearance of important food fish, these latter species are increasingly being targeted for consumption. As a result of fishing pressure, the fish stock of the reefs (local fishing) and the pelagic zone (international fishing elsewhere) of the Caribbean Netherlands is largely characterized by a low density of large predatory fish species (Sandin et al., 2008; Pattengill-Semmens, 2002; van Kuik et al., 2015).

Table 1. *Overview of fish species of the Caribbean Netherlands attributed a vulnerable Conservation State by IUCN. Only three of these species (*) are not target species for consumption.*

	Scientific	Common	Local	IUCN category
Pelagic	*Makaira nigricans*	Blue marlin	Balaú pretu	VU
	Kajikia albida	White marlin	Balaú blanku	VU
	Thunnus obesus	Bigeye tuna	Buní	VU
	Thunnus alalunga	Albacore	Buní	NT
	Albula vulpes	Bonefish*	Warashi	NT
	Megalops atlanticus	Tarpon*	Sábalo	VU
Deep water	*Rhomboplites aurorubens*	Vermillion snapper		VU
	Hyporthodus niveatus	Snowy grouper	Djampou	NT
Reef	*Lutjanus cyanopterus*	Cubera snapper	Caraña	VU
	Lutjanus analis	Mutton snapper	Kapitán	VU
	Lutjanus synagris	Lane snapper	Kora spañó	NT
	Dermatolepis inermis	Marbled grouper	Olitu	NT
	Epinephelus itajara	Goliath grouper	Djùkfis	CR
	Epinephelus morio	Red grouper	Djampou	NT
	Epinephelus striatus	Nassau grouper	Jakupeper	EN
	Mycteroperca interstitialis	Yellowmouth grouper	Patachi	VU
	Scarus guacamaia	Rainbow parrotfish*	Gutu kedebe	NT
	Lachnolaimus maximus	Hogfish	Hogfès	VU
	Balistes vetula	Queen triggerfish	Pishiporko rab'i gai	NT

Fishery status: In addition, to international Conservation State as above (Table 1), Wageningen Marine Research also provides FAO annually with (expert) assessments of the fishery status of many species for the islands (FAO, 2022). The assessments of most species that are assessed as being overfished are based on their known prior historical abundance and current near absence. For some species recent data allow a more quantitative expert judgement (Table 2). These species are being assessed as either a) likely not overfished, b) certainly overfished or c) of uncertain fishing status for Bonaire, Saba and or St. Eustatius.

Table 2. Fishery status for selected commercial reef associated fisheries resources of Bonaire, Saba and St. Eustatius: underfished, maximum yield, overfished, possibly overfished, unknown. Source: FAO, 2022.

Island/Scientific name	Local name	Fishery status	Assessment approach	Reference year	Assessment year	Assessment Availability
Bonaire						
Balistes vetula	Pishiporko rabi'gai	Overfished	historical length-frequency	2003	2005	Yes
Cittarium pica	Kiwa	Overfished	expert opinion	1982	1987	Yes
Epinephelus guttatus	Gatu kora	Overfished	historical length-frequency	2003	2005	Yes
Epinephelus itajara	Djukfes	Overfished	historical length-frequency	2003	2005	Yes
Epinephelus striatus	Jakupeper	Overfished	historical length-frequency	2003	2005	Yes
Lobatus gigas	Karko	Overfished	density and size-frequency surveys	2000	2000	Yes
Lutjanus cyanopterus	Caranja	Overfished	historical length-frequency	2003	2005	Yes
Lutjanus jocu	Baster bers	Overfished	historical length-frequency	2003	2005	Yes
Melichtys niger	Doro	Overfished	historical length-frequency	2003	2005	Yes
Mycteroperca interstitialis	Patachi	Overfished	historical length-frequency	2003	2005	Yes
Mycteroperca tigrinus	Gramel	Overfished	historical length-frequency	2003	2005	Yes
Mycteroperca venenosa	Djampou	Overfished	historical length-frequency	2003	2005	Yes
Panulirus argus	Kref	Overfished	old reports	2003	1985	Yes
Sphyraena barracuda	Piku/ Snuk	Possibly Overfished	expert opinion		2023	No
Saba Bank						
Balistes vetula	Moonfish	Maximum	CPUE and catch trends	2020	2021	Yes
Epinephelus guttatus	Red hind	Maximum	CPUE and catch trends	2018	2021	Yes
Epinephelus striatus	Nassau grouper	Overfished	CPUE and catch trends	2020	2021	Yes
Lobatus gigas	Conch	Underfished	density and size-frequancy data	2014	2015	Yes
Lutjanus buccanella	Blackfin snapper	Unknown	Catch trends	2020	2021	Yes
Lutjanus cyanopterus	Cubera snapper	Overfished	CPUE and catch trends	2020	2021	Yes
Lutjanus jocu	Dog snapper	Overfished	CPUE and catch trends	2020	2021	Yes
Lutjanus vivanus	Yellow-eye snapper	Maximum	CPUE and catch trends	2020	2021	Yes
Melichtys niger	Black triggerfish	Unknown				
Mycteroperca interstitialis	Yellowmouth grouper	Overfished	CPUE and catch trends	2020	2021	Yes
Mycteroperca tigrinus	Tiger grouper	Overfished	CPUE and catch trends	2020	2021	Yes
Mycteroperca venenosa	Yellowfin grouper	Overfished	CPUE and catch trends	2020	2021	Yes
Panulirus argus	Lobster	Maximum	CPUE and catch trends	2020	2021	Yes
Rhomboplites aurorubens	Wenchman	Unknown	Catch trends	2020	2021	Yes
Sphyraena barracuda	Barracuda	Underfished	CPUE and catch trends	2020	2021	Yes
Saba Island						
Cittarium pica	Wilk	Unknown	expert opinion	2020	2020	No
Sint Eustatius						
Balistes vetula	Moonfish	Maximum	CPUE and catch trends	2020	2021	Yes
Cittarium pica	Wilk	Maximum	expert opinion	2020	2020	No
Epinephelus guttatus	Red hind	Unknown	CPUE and catch trends	2020	2021	Yes
Epinephelus striatus	Nassau grouper	Overfished	CPUE and catch trends	2020	2021	Yes
Lobatus gigas	Conch	Underfished	density and size-frequency data	2014	2015	Yes
Lutjanus cyanopterus	Cubera snapper	Overfished	CPUE and catch trends	2020	2021	Yes
Lutjanus jocu	Dog snapper	Overfished	CPUE and catch trends	2020	2021	Yes
Melichtys niger	Black triggerfish	Unknown	CPUE and catch trends	2020	2021	Yes
Mycteroperca interstitialis	Yellowmouth grouper	Overfished	CPUE and catch trends	2020	2021	Yes
Mycteroperca tigrinus	Tiger grouper	Overfished	CPUE and catch trends	2020	2021	Yes
Mycteroperca venenosa	Yellowfin grouper	Overfished	CPUE and catch trends	2020	2021	Yes
Panulirus argus	Lobster	Unknown	CPUE and catch trends	2020	2021	Yes
Sphyraena barracuda	Barracuda	Underfished	CPUE and catch trends	2020	2021	Yes

Basic overview

Fish Stock of Bonaire: Data on the fish stock for Bonaire are limited to reef fish in shallow waters on the protected west coast. The fish stock has been studied mainly in small areas of the reef (Steneck and Arnold, 2013; Steneck et al. 2011; 2013; 2015; Steneck and McClanahan, 2005; Hawkins et al., 2007; Hylkema et al., 2014; Luckhurst and Luckhurst, 1978; Nagelkerken et al., 2002; Pattengill-Semmens, 2002; Sandin et al., 2008).

The key findings from these studies indicate that Bonaire, compared to many other reefs in the region, has high densities of small fish. The most abundant fish are plankton and plant eaters (Sandin et al., 2008). The larger predatory fish, such as groupers, have become rare on the west coast (de Graaf et al., 2016). A comparison of spear fishing catches between the 1950s and 1960s and the 1990s shows that large reef fish, especially large groupers, were previously abundant (Debrot and Criens, 2005, Vermeij et al., 2019). This decline is supported by past counts (Fig. 1) and a current assessment of 14 distinct reef-bound demersal species (of fish and shellfish) for Bonaire indicate overfishing for all but one of the 14 stocks (Table 2). However, Meesters et al. (this report) have been assessing fish densities as part of the so-called Reef Health Index for several years now (see the chapter on coral reefs, Table 3) and conclude a slight positive trend in commercial fish densities (groupers, snappers, grunts and other predatory piscivores).

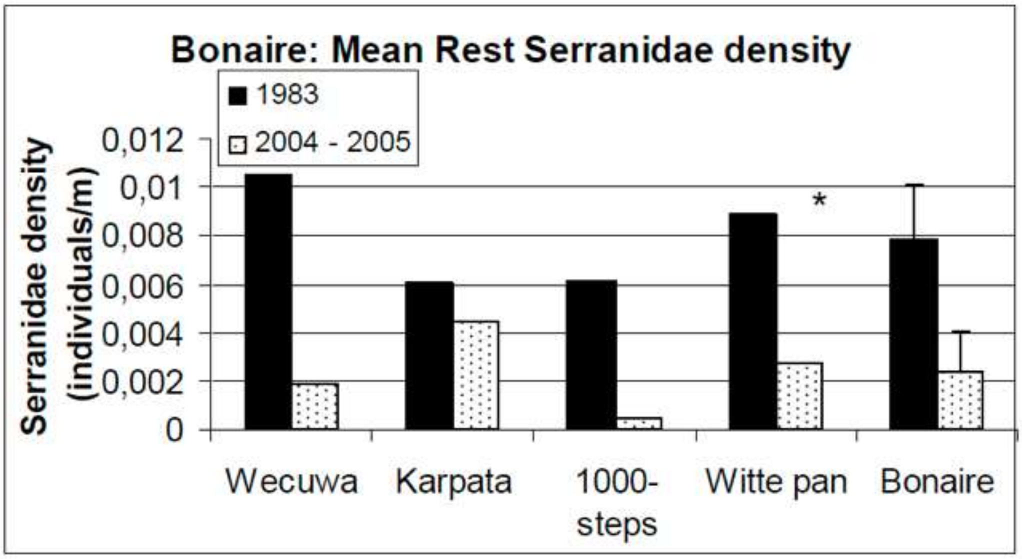

Figure 1. *The drastic decline of the large grouper species around Bonaire ("Rest" are all groupers except the grasby Epinephelus cruentatus and cony E. fulvus). Source: Debrot and Nagelkerken, unpublished data from the thesis by G. Atsma and J. Bosveld, 2006.*

The current annual fish catches for Bonaire (excluding schooling baitfish but including pelagic species) amount to a total of 103 tons per year (de Graaf et al., 2016). Fishing, in any case, remains a marginal activity. The average daily catch for a "large boat" with two crew members over an average time span of 9.5 hours is 28.1 kg (Graaf et al., 2016). This equates to an average of 1.5 kg (approximately US$15 per fisherman per hour). After deducting all fuel and other boat costs, even "commercial" fishing remains a marginal activity. For coral reef fishing alone, the "commercial" coral reef fish catches (including coastal pelagics) have an annual value of approximately US$400,000, while the sport fishing (particularly by those who do not fish as a profession but fish for consumption) is estimated at nearly US$700,000 per year (Schep et al., 2012).

With suspected declines in catches of pelagic transboundary species (like tunas, wahoo and dorado) around Bonaire since around 2010, a worrisome development is the increased targeting of the Greater barracuda (*Sphyraena barracuda*) by trolling charter boats (van Slobbe, Bonaire Agriculture, Livestock and Fisheries Service, pers. comm.). The species may be suffering from overfishing and is also vulnerable as the seagrass beds and lagoonal habitat of Lac Bay, which serve as its major nursery habitat for Bonaire, are also in decline (Christianen et al., 2018; Debrot et al., 2019).

Fish Stocks of Saba: The shallow coral reef fish stocks of Saba are described and compared in several small studies (Hawkins et al., 2007; Klomp and Kooistra, 2003; Polunin and Roberts, 1993; Looiengoed, 2013; Roberts, 1995; Roberts and Hawkins, 1995; Vlugt, 2016). The most recent studies indicate that the densities of large predatory fish around Saba remain much higher than in the rest of the Caribbean (Vlugt, 2016). This is attributed to the low fishing pressure around Saba. Data from 2007 for the coral reefs of the Saba Bank also show that the densities of large predatory fish (groupers and snappers) and sharks were high compared to the region (Toller et al., 2010). As of 2013, the densities of large predatory fish appear to have declined, while shark populations have remained abundant (Stoffers, 2014). Additionally, the fish community of the Saba Bank is characterized by the absence of species dependent on seagrass beds and mangroves (Toller et al., 2010; Stoffers, 2014).

The Saban fishery is almost entirely conducted on the Saba Bank, where lobster fishing is by far the most important activity (Lundvall, 2008). The annual value of the combined fishery, estimated based on catches from 2007-2012, amounts to approximately US$1.3 million (Toller and Lundvall, 2008). Additionally, the red snapper fishery is the second most important, with about 36.5 tons of "redfish" being caught (Boonstra, 2014). The fishery of Saba operates with 10 licensed holders/boats and provides employment for about 30 people (Boonstra, 2014).

Fish Stocks of St. Eustatius: The shallow coral reef fish stocks of St. Eustatius is discussed in five studies (Klomp and Kooistra, 2003; Kuik et al., 2014; McLellan, 2009; Sybesma et al., 1993; de Graaf et al., 2015). These studies indicate that large predatory fish have disappeared (Sybesma et al., 1993; McLellan, 2009; de Graaf et al., 2015), herbivore density is higher in areas where predatory fish are fished, and the fish communities of the island are characterized by the absence of species dependent on mangroves and an apparently higher density of sharks (Kuik et al., 2014). The total annual fish catch is estimated at 18 tons per year (11 tons of Caribbean spiny lobster, 4 tons of reef fish, 2 tons of queen conch, and 1 ton of pelagic fish) (de Graaf et al., 2015). According to ECORYS (2010), about 15 people make their living from fishing, with catches estimated at an annual value of approximately US$190,000 (Lely et al., 2014), or a gross annual catch of US$12,600 per fisherman and an average of US$380 gross per fishing day.

Description of Fisheries: Since visual fish counts performed with SCUBA diving provide insight into only a small portion of the fish stock of the Caribbean Netherlands, the fishery itself remains the most important indicator of the state of the fish stock in coastal waters. This section therefore briefly discusses the fishery sector of each island and what this tells us about the fish stock.

Fisheries of Bonaire:
The current state of the Bonairean fishery is largely traditional with little innovation. Aside from motorized propulsion instead of sails and the use of nylon fishing lines and nets instead of cotton braided lines and nets, virtually the same types of boats and techniques are used as a century ago. The fleet consists of approximately 84 small outboard motorboats and 26 larger boats over 7 meters, with cabins and largely diesel-powered. The total annual fish catches for Bonaire, about 103 tons (excluding small schooling pelagic species), are divided among fishing from shore with casting lines (± 12 tons), small gasoline-powered fishing boats without cabins (± 30 tons), and larger, primarily diesel-powered fishing boats over 7 meters in length (± 60 tons) (de Graaf et al., 2016; Tichelaar, 2015).

Noteworthy is the absence in the catches of species that were historically important, such as large groupers from the reef and highly migratory pelagic species like Mahi Mahi (*Coryphaena hippurus*), Rainbow Runner (*Elagatis bipinnulata*), and Yellowfin Tuna (*Thunnus albacares*). Due to the lack of historical data collection, long-term trends in the development of Bonaire's fish stocks are unknown. However, comparisons of sport fish catches between the 1950s and 60s and the 1990s, and based on fish counts, show that large reef fish (especially large groupers) were previously abundant (Debrot and Criens, 2005; Debrot and Nagelkerken, unpublished data; de Graaf et al., 2016; Vermeij et al., 2019). According to Erik Meesters, the average size of Yellow-tail Snappers, *Ocyurus chrysurus*, on the reefs has also declined. (Meesters, unpublished data) which would be a prime indicator of overfishing.

Fisheries of Saba:
General: Fishing in Saba targets the Caribbean spiny lobster (*Panulirus argus*) and the so-called "redfish" (*Lutjanus vivanus, Lutjanus buccanella, Rhomboplites aurorubens, Lutjanus synagris*, deepwater Lutjanidae) on the Saba Bank. The pelagic fishery is negligible. While 60% of the commercial fishery focuses on the lobster, 40% of the fishing activity targets "redfish" (Boonstra, 2014). In 2006, Saba imposed a moratorium on issuing new commercial fishing licenses for the Saba Bank due to declining catches. Since then, only existing licenses have been renewed (and in a few cases transferred to another fisherman), with no new licenses issued, even by the Fisheries Commission of the Netherlands Antilles, which advised on EEZ waters.

Caribbean Spiny Lobster: This fishery started with the rise of tourism on St. Maarten in the 1980s. The lobster is fished on the Saba Bank to depths of approximately 45 m using traps, meaning that 84% of the Saba Bank possibly has suitable habitat for this fishery (Toller and Lundvall, 2008). However, while the habitats of the Saba Bank have been mapped (Meesters et al., 2024), their specific roles for different species yet remain unknown. In 2012, around 1780 traps were used in this fishery (222 traps per fisherman and 8 fishermen). The number of fishermen has since increased to 10, and the number of traps per fisherman has increased to between 250 and 300 traps (de Graaf et al., 2017). The number of "trap sets" increased from approximately 48,000 sets/year in 2012 to about 73,000 sets/year in 2015 (de Graaf et al., 2017). The traps are emptied on average every two weeks. Most of the catch is exported to St. Maarten (Dilrosun, 2000). The total annual lobster catch is estimated at 62 tons, 92 tons, and 38 tons in 1999, 2007, and 2012, respectively (Fig. 2). The lower catch in 2012 compared to 1999 is largely due to a much lower CPUE (de Graaf et al., 2017). This suggests a likely decline in lobster populations, similar to the regionally observed decline during the same period (van Gerwen, 2013; FAO, 2011). Since then, fishing effort and realized catches on the Bank have steadily increased again, with a total annual catch of approximately 77 tons in 2015. After this date, fishing effort has decreased constantly and was in 2023 about half of the effort in 2025. Catches have also declined after 2015 to reach very low values in 2018 (around 25t) but have partially recovered afterwards and have been stable around 40t in the recent years. Based on models, the CPUE seems to have increased, first quickly from a low point in 2011 until 2013 and continued to increase steadily since then to reach in 2023 values like those recorded in 2007. The recent increases in total catches are therefore partly attributable to higher abundance (CPUE) but despite the somewhat lower effort. After all, the total catch is simply the product of effort multiplied by catch per unit effort. The annual catch development on the Saba Bank since 2000 appears to follow the regional catch development, likely driven by regional recruitment patterns (de Graaf et al., 2017). A healthy sign of the fishery is that the average size remains high compared to most lobster fisheries in the region (de Graaf et al., 2017; Brunel et al., 2018; 2020; Domingues et al., 2024 in prep.).

Bycatch in the Lobster Fishery:

Coral Reef Fish Bycatch: The lobster fishery has a significant bycatch of coral reef fish, with approximately 33% being discarded at sea. The three main species landed are the Queen Triggerfish (*Balistes vetula*), the White Grunt (*Haemulon plumierii*), and the Red Hind (*Epinephelus guttatus*), which together account for more than 50% of the weight of the landed bycatch. The annual landed coral fish bycatch increased from 6.6 tons to 13.6 tons between 2012 and 2015 due to increased fishing pressure on lobsters and has been variable in this range since then. The total catches on the Bank range between 0.025 and 0.10 tons/km²/year, which is very low compared to the rest of the region. This could be due to the generally low fish densities on large parts of the Bank and/or relatively low fishing pressure (with only 10 small active fishing vessels). Research shows that coral fish bycatch can be easily reduced without adversely affecting lobster catches by using smaller fish traps and installing escape slots. As of 2024, the implementation of the required escape slots has not really been pursued.

Shark Bycatch: Lobster traps frequently catch young Nurse Sharks, with an annual bycatch of between 1,700 and 2,400 individuals, almost all of which are (likely) released unharmed. Research into potential modifications to lobster traps to reduce Nurse Shark bycatch has significant benefits for reducing financial damage to fishing gear and catches, as well as for the protection of the species.

Ghost Traps: Saban fishers lost an average of 0.6 lobster traps per trip between 2012 and 2015, corresponding to an annual loss of 400-600 traps. These lost traps are known as ghost traps because they continue to fish without new bait. Experiments show that ghost traps can kill an average of 2.7 - 7 lobsters and 2.7 - 3.9 kg of coral fish annually. For the total number of lost traps, considering how long they continue fishing, this results in an annual loss of US$ 23,000 - US$ 51,000 in coral fish value and US$ 46,000 - US$ 176,000 in lobster value. Research indicates that the use of biodegradable panels can drastically mitigate this problem without being detrimental to the fishery (de Graaf et al., 2017).

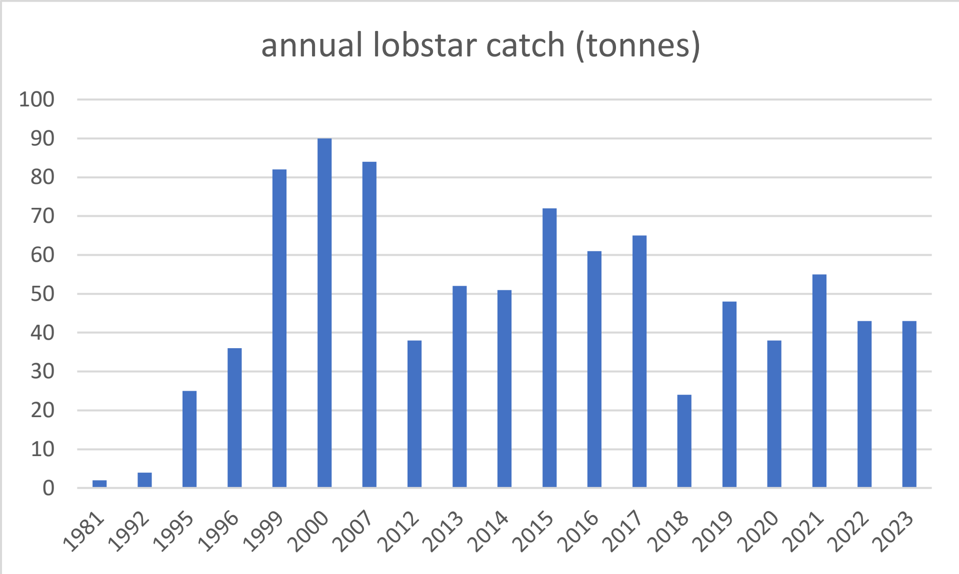

Figure 2. *Total weight of annually landed Caribbean Spiny Lobster from the Saba Bank (data 1981-2000 from Dilrosun, 2000, data 2007 from Toller and Lundvall, 2008, data 2012-2015 from de Graaf et al. 2017 and data from 2018-2023 from Brunel et al. 2018, 2021, Domingues et al. 2024, in prep.).*

Redfish: "Redfish" includes a selection of (red coloured) deepwater snappers (Lutjanidae: Silk Snapper, Blackfin Snapper, Lane Snapper, and Vermillion Snapper), which are nowadays primarily caught with fish traps at depths between 50 and 250 meters. In the 1970s, redfish were almost exclusively caught with lines, but gradually more and more traps were used. Around 2000, the fishermen temporarily implemented a voluntary moratorium on the use of traps. In 2007/8, the situation had changed to primarily a trap fishery alongside a limited line fishery; and by 2012, there was practically no line fishing for redfish anymore, and all fishermen used traps. This shift in fishing gear and water depth was accompanied by a shift in the size of the caught fish, from large fish of half a meter or more (Dilrosun, 2000) to (sub-adult) fish around 30 cm ('plate' size). This shift also led to a strong increase in the landings, from just above 10t in 2000 to over 37t in 2007 (Fig. 3). This increase continued in subsequent years to reach a record in 2019 at 54t. In 2017, 2022 and 2023, the fishers have implemented seasonal closures of the fishery which led to much lower landings in these years (around 20t). Due to the depths at which these fish live, relatively little is known about the status of these snapper species stocks. Both Dilrosun (2000) and Toller and Lundvall (2008) have pointed out the relatively high catches of sub-adult Silk Snappers (*Lutjanus vivanus*) in this fishery. Since 2000, the CPUE for redfish has fluctuated between 2.5-5 kg of fish per retrieved fish trap. This is 75% lower than in the 1970s, indicating a significantly reduced fish stock. However, the reliability of the data from the seventies is questionable, as they were extrapolated from catches by one fisherman in a short period. Changes in total catches since 2000 seem primarily due to differences in fishing effort, but without a decrease in densities, otherwise CPUE would also have decreased, which is not the case. These most recent findings therefore do not suggest an overly concerning development in this fishery. There is a small but now growing fishery for redfish with deepwater lines (longlines). This takes place in deeper water (on average 260 m instead of 50-115 m). The composition of the catch is different, and the dominant fish species is the Wrenchman (*Pristipomoides aquilonaris*), followed by the Queen Snapper (*Etelis oculatus*; sabonèchi) (de Graaf et al., 2017; Brunel et al., 2018; 2021; Domingues et al., 2024 in prep.).

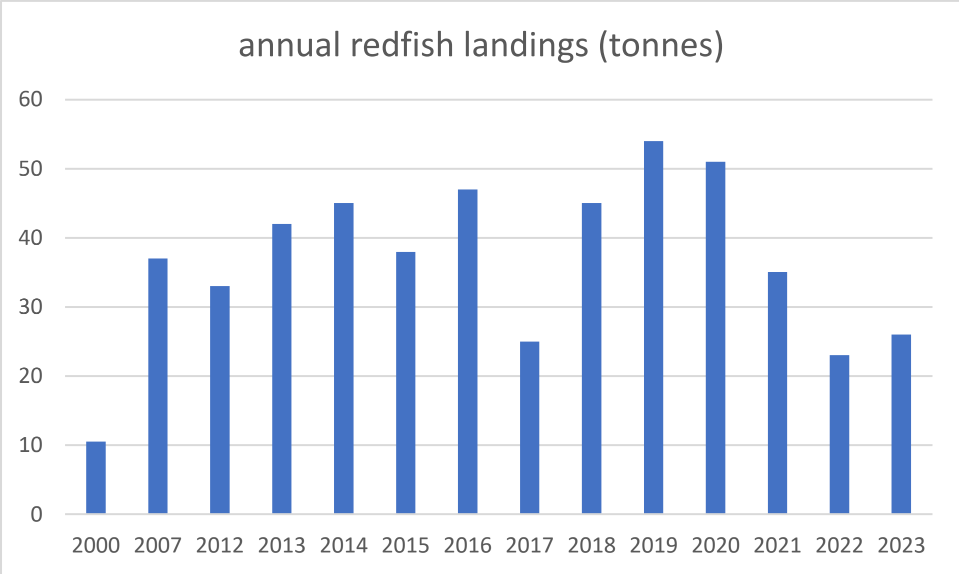

Figure 3. *Development in annual landings of redfish from the Saba Bank - Source: de Graaf et al. 2017, Brunel et al. 2018, 2021, Domingues et al, 2024 in prep.*

Fisheries of St. Eustatius

General: Over the past 15 years, the fishing sector in St. Eustatius was made up of about 5 active fishermen and a fleet of 15-20 small (<10m) wooden boats. The total fishing effort amounts to approximately 500 boat-days per year. The fishery primarily targets the Caribbean Spiny Lobster. In 2000, there were still some boats from St. Eustatius larger than 12m, with fishing permits for the Saba Bank, but this was reduced to zero by 2010. To protect fish stocks, two fishing reserves were established around the island with the creation of the St. Eustatius Marine Park in 1996. Since then, three studies have been conducted to determine whether these reserves have had a demonstrable positive effect on fish stocks (White et al., 2006; McLellan, 2009; van Kuijk, 2014). White et al. (2006) compared fish stocks at fixed locations between 2004 and 1992 and concluded that the fish population had increased by a factor of 4.9. In contrast, McLellan (2009) and van Kuijk (2013) compared fish stocks between locations inside and outside the reserves and could not demonstrate a significant difference in fish densities between areas inside and outside the reserves. This raises doubts about the effectiveness of these reserves, but there are clearly contrasting results and various explanations possible. Fig. 4 shows the recent developments in fisheries catches for both the lobster and reef fish catches for St. Eustatius.

Caribbean Spiny Lobster: Forty percent of landed lobsters are smaller than the minimum allowed size of 95mm CL (carapace length). This is most likely due to the St. Eustatius lobster regulation from the 1960s, which prescribed a minimum size of 86mm. Formally, this smaller minimum size has not been in effect since the higher hierarchy national fishing ordinance, now the BES Fisheries Act, came into force, but this was never enforced on St. Eustatius, so fishermen continued to use the smaller size. The average size of the lobster appears to have decreased from 110mm CL in 2003 to 99mm CL in the period 2012-2015 and has remained around 100mm CL since then (Amelot et al., 2021). Additionally, the catch per unit area (500 kg/km²) seems very high compared to the rest of the region and what is sustainable in the long term. For these reasons, the fishery appears to be heading towards overfishing, and it is doubtful whether the maximum economic yield can be achieved (Graaf et al., 2015). Current information indicates a potentially unfavourable development in the status of this species.

Reef Fish: Small grouper species (Serranidae) and ecologically essential herbivores such as surgeonfish (Acanthuridae) make up about 50% of the reef fish catch composition. The numerically most important species were the Blue Tang (25%), Squirrelfish (10%), Honeycomb Cowfish (10%), and the Doctorfish (9%). These species were considered worthless bycatch until a few decades ago but are now marketed due to the disappearance of the more valuable large consumable fish. This is an example of the phenomenon known as "fishing down the food chain," where successively less valuable fish are caught. The total annual catches (~0.2 t/km²/y) seem limited compared to the catches possible for coral reef ecosystems (0.2-27 t/km²/y). The current low catches of small fish are likely indicative of catches typical for degraded reefs and long-term chronic overfishing. Research has shown that in the past the reefs of St. Eustatius were in much better shape (Sybesma et al., 1993; Klomp and Kooistra, 2003). The status of the reef fish stock therefore appears unfavourable. This is likely less due to the limited catches currently being achieved than to the overall degradation of the reefs that has taken place over the past decades (Debrot et al., 2014). The modelled CPUE for this group is species has generally been decreasing between 2012 and 2020, indicating a general deterioration of the biomass of these species (Graaf et al., 2015; Brunel et al., 2020; Amelot et al., 2021).

Upon request by St. Eustatius, and as advised by Wageningen Marine Research (Debrot and de Graaf, 2018) two permits have been authorized for St. Eustatius boats to fish on the Saba Bank. However, so far no use of these permits has been made because the boats used in the St. Eustatius fishery are unsuitable for fishing on the Bank (K. Kitson-Walters, pers. comm.).

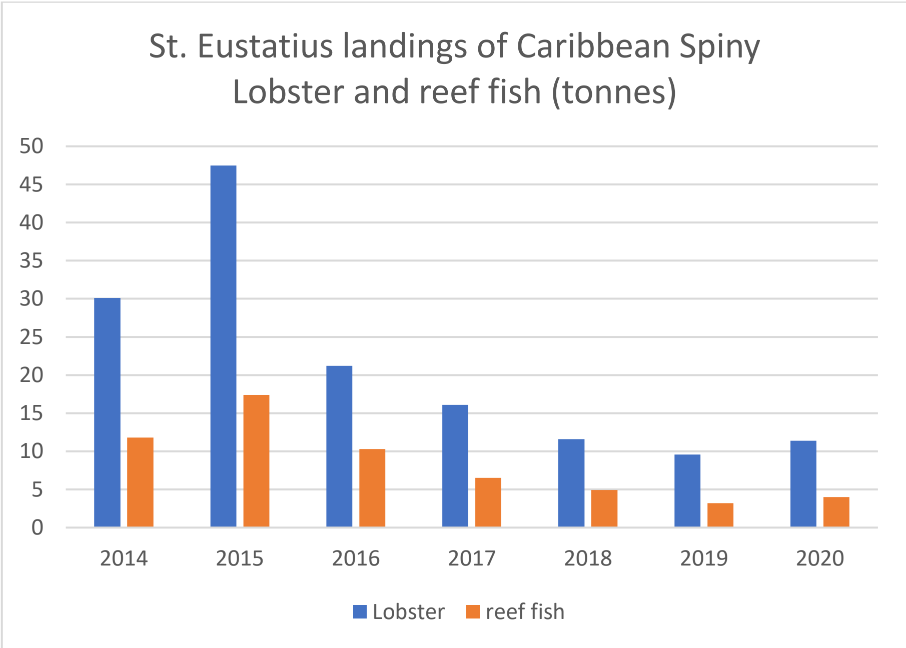

Figure 4. *Development in annual landings of Caribbean Spiny Lobster and reef fish from the Sint Eustatius - Source: Amelot et al. 2021.*

Queen Conch: The Queen Conch (*Lobatus gigas*; Strombidae; Gastropoda) is a large, long-living sea snail widely found throughout the Caribbean. The conch forms the basis of an important fishery and is heavily fished throughout the region for consumption. Due to large-scale overfishing across its range, trade in this species is regulated through the CITES convention. Around St. Eustatius, the species is mainly found at depths of 16 to 30 meters in areas with coral rubble and seagrass. In 2013, approximately 5000 adult Queen Conchs were landed for local consumption, which amounts to about 3% of the adult population (Meijer zu Schlochtern, 2014; de Graaf et al., 2014). The favourable development for this species around St. Eustatius indicates an increasing population of sexually mature adults, and limited export seems possible (de Graaf et al. 2014). Conch remains abundant around the island (Debrot and Clements, pers. obs. 2023).

Pelagic Fish: The fishery for larger migratory pelagic fish species is very underdeveloped in St. Eustatius, and there are no data series. Therefore, little is known about the status of the relevant fish stocks. The Department of Agriculture, Livestock, and Fisheries (LVV) has installed a Fish Attracting Device (FAD) to attract pelagic fish and make it more attractive for local fishermen to target pelagic species (many of which are not yet overfished), thereby reducing fishing pressure on the coral reef.

Conclusions: The fishing pressure on many reef fish around St. Eustatius is too high, with adverse consequences for the fish stocks. In contrast, the fishing pressure on Queen Conch and certain small pelagic fish stocks (like blackfin tuna, rainbow runner, flying fishes, but not the large tunas) is very limited, and a shift of fishing pressure to less-fished species may offer opportunities for the recovery of overfished stocks. For a list of 13 individual species the fishery status of six are deemed as certainly overfished while two may be considered underfished while the fishery status of lobster is uncertain (Table 2).

Ecological Characteristics

Fish form a very important and diverse part of the biodiversity found in the waters of the Caribbean Netherlands and play very important roles in the marine ecosystem. Herbivorous fish, for example, are important for maintaining the coral reef by preventing it from being overgrown by algae. Planktivorous fish capture the scarce food from the clear water, thereby maintaining nutrients in the ecosystem. Predatory fish, on the other hand, are important for keeping the populations of herbivorous and planktivorous fish healthy and balanced and provide an important ecosystem service to humans in the form of edible fish. The most studied component of tropical fish stocks are the medium-sized fish of the shallow reef that are active during the day (e.g., Pattengill-Semmens, 2002; Sandin et al., 2008; Toller et al., 2010; Steneck and Arnold, 2013; van Kuijk et al., 2015; Vlugt, 2016). Much less is known about nocturnal fish, large predatory fish, deepwater fish, small cryptic fish, and pelagic fish. Tropical fish stocks are known for their high diversity, which is also partly reflected in the many species caught. The management of such tropical "multi-species" fish stocks is complex compared to "single species" fish stocks typical of temperate seas. The coral reef fish stocks of the Caribbean Netherlands are typical for oceanic islands, where many fish families and species commonly found on continental reefs are absent (Sandin et al., 2008). However, there are also significant differences between the islands themselves. For example, there are large differences in species composition between Bonaire and the islands of Saba and St. Eustatius, which are clearly due to the absence of mangroves on the latter two islands (van Kuijk et al., 2015). The vast pelagic habitat of the open sea is of great importance to all fish. Not only as a habitat for large pelagic predators but also for the hundreds of coral reef fish species that spend a significant part of their larval stage there. Johns et al. (2014) have found that the Caribbean sometimes experiences large plumes of Amazon River water and that this is associated with lower larval densities of coral reef fishes. Hence, this factor may also prove to be a critical factor possibly affecting coral reef fish recruitment in the region and is deserving of further study. Due to dispersion by ocean currents during the larval stage, fish stocks over large areas are effectively connected and therefore need to be managed collectively. While adult coral reef fish species are usually highly site-specific, large pelagic predators are typically highly migratory, moving in large schools through the Caribbean each year. These so-called "transboundary" species are fished by different countries during their annual migration through the region and therefore need to be managed collectively.

Assessment of National Trends

Developments within the Caribbean Netherlands:

Bonaire
In former times, fish catches were high, and the export of fish from Bonaire was economically very important (in 1956, it accounted for 44% of the total export) (Hartog, 1957). For example, in 1956, the total catch was approximately 140 tons (Zaneveld, 1961). Estimates of total catches for the years 1978 and 1979 were similarly high (160 tons) (Palm, 1985). At that time, pelagic species accounted for 80% of the catches. These were mainly: Scombridae - tuna, *Acanthocybium solandri* - Wahoo, *Coryphaena hippurus* - Dolphin Fish/Mahi-Mahi, Xiphiidae & Istiophoridae - swordfish and marlins, *Elagatis bipinnulata* - Hawaiian Salmon, and Exocoetidae – flying fish. In interviews conducted by Johnson (2011), a fisherman stated that groupers were exported by airplane. The reef fish that could previously be spearfished while snorkelling are hardly present anymore nowadays (Fig. 5; Debrot and Criens, 2005). Catches have significantly declined over the years. It can be inferred from this that fish stocks must have similarly declined. Handline fishing during snorkelling has become popular in recent years on Bonaire, and due to the efficiency of this method, it poses a new threat to the already impoverished fish stocks.

A recent development affecting not only Bonaire but also the fish stocks of other areas is the population explosion of the Lionfish. This non-native fish is not recognized as a predatory fish and is therefore capable of depopulating reefs of young coral reef fishes. It is not known whether the species poses a long-term threat or not, and various studies from different areas seem to show different results. For the Saba bank in any case a major decline in density has been documented (debrot et al. 2023) but whether this is the case elsewhere remains unknow.

Figure 5. *Example of fish catches for an afternoon of snorkelling with two spearfishermen on the east coast of Bonaire in the 1960s. (Photo J. Streder, A. Debrot collection).*

Saba Bank

Spiny Lobster: There are hardly any historical data available on the spiny lobster fishery and stock of the Saba Bank. According to fishermen, this fishery only started with the rise of tourism on St. Maarten in the 1980s. The average size of the spiny lobster seems to fluctuate between 108 cm CL and 117 cm CL since 2000. No decrease in average size has been observed. However, the percentage of landed spiny lobsters smaller than the legally allowed minimum size of 95 cm has decreased from 28% in 2012 to 4% in 2015, which indicates better compliance to the legislation. Only a limited part of the bank is fished, and most fishing pressure is clearly concentrated in the northern and eastern sectors closest to Saba (Gerwen, 2013).

Redfish: No significant historical catch data are available for the redfish fishery before 2000. While an average of 28 redfish traps were retrieved per day in 2007, this was 37 per day in 2012. Fishing effort thus has increased, but annual catches seem to have decreased from 41.3 tons in 2007 to 34.6 tons in 2012. After 2012, total catches increased to 51 tons in 2014, followed by a decrease in catches again (42 tons in 2015). Wolf & Chislett (1974) concluded in their study that during the years 1966-1971, the Saba Bank was the area in the Caribbean with the highest observed CPUE for redfish, and population densities were among the highest in the entire Caribbean region (because there was no fishing yet). In 2016, Saba decided in consultation with fishermen to introduce some restrictions. These include restrictions on the mesh size of traps to a minimum of 3.8 cm, a maximum number of traps per fisherman (25), and the installation of biodegradable panels so that lost traps do not continue to fish for a long time.

Coral Reef Fish: The large groupers that used to be abundant on the Saba Bank (Meesters et al., 1996) are now hardly present anymore (Toller et al., 2010). The only medium-sized grouper that was still present in significant numbers until 2007 is the Red Hind, *Epinephelus guttatus* (Toller et al., 2010), but the situation seems to have changed unfavourably since then (Stoffers, 2014). Since 2013 the Red Hind has been protected by a seasonal closure of the presumed key spawning grounds on the Saba Bank. Preliminary analysis of the effectiveness of the spawning grounds closure was examined based on CPUE and mean size trends in the landings of Red Hind and the internationally Near Threatened Queen triggerfish, *Balistes vetula* (Table 1). However, the results gave no indication of any improvement in CPUE or mean size caught for either of the two species since the seasonal closure was initiated in 2013 (Debrot et al., 2020). Results even suggest a small but significant decrease in the size of Red Hinds caught as by catch in the lobster pot fishery. This means that, based on the port sampling method used, no significant positive effect on the Red Hind and Queen Triggerfish populations of the 5-year closure can yet be demonstrated and further research using fisheries independent sampling for a more precise assessment of effectivity of the spawning closure is currently underway.

Apart from the fact that current catches of the Caribbean spiny lobster and deep-water redfish are lower than before, neither fishery has shown a continuation of worrying developments in recent years. The situation seems stable. In addition, research has suggested adjustments for traps that can reduce the adverse and/or potentially adverse effects of bycatch of coral fish and sharks and the problem of ghost fishing.

St. Eustatius
For St. Eustatius, the availability of good data is extremely limited. Current results do not show convincingly that fishing pressure on the spiny lobster around St. Eustatius would be too high but groupers and snappers are overfished (Table 2). However, fishing pressure on the Queen Conch and pelagic fish stocks is limited, and a shift in fishing pressure to less-fished species may offer opportunities for the recovery of overfished stocks.

It seems that the composition of reported catches has changed enormously over time (de Graaf et al., 2015), even though the size of the fishermen's "fleet" has hardly changed in the past 100 years. While around 1906 only 5% of the catch consisted of small herbivorous surgeonfish and trunkfish (Acanthuridae and Ostracidae), these two families accounted for almost 50% of the catch in 2012-2015. Over the same period, the representation of economically significant groupers (Serranidae), grunt fish (Haemulidae), jacks (Carangidae), and triggerfish (Balistidae) has almost halved compared to the early 1900s (Boeke, 1907). However, trends to 2023 (Graaf et al., 2015; Brunel et al., 2020; Amelot et al., 2021) show that catches since 2015 have declined considerably which is worrying.

Finally, Following exploratory trials with FADs (Fish Attracting Devices) to attract schooling pelagic fishes of commercial interest in Curaçao in the 1990s (Buurt, van 1995; 2000; 2002), recently the interest in the use of FADs has grown enormously with numerous of these devices being deployed in the waters around Bonaire, St. Eustatius and near the Saba Bank. If properly managed, a fishery more strongly based on FADs presents important opportunities to sustainably expand domestic fisheries yields for all islands (Debrot and van der Burg; 2019; Lotz et al., 2020). However, management and regulation of these fisheries are crucial to avoid excessive deployment of these expensive devices, as with too many, the added value of more will rapidly decline as a saturation point is gradually reached. In addition, for the species targeted with FADs, regulations on catches are also needed to avoid overfishing. For instance, recent observations by the Bonaire Agriculture, Livestock and Fisheries Service of Bonaire (F. van Slobbe, pers. comm.) suggest that large pelagic species have been on a long-term decline around Bonaire since 2010 but data on this remain totally lacking.

Current Distribution and Reference Values

There are no true reference values for exploited coral fish species (due to a lack of old reference data before these fish stocks became excessively exploited). The only thing certain is that for many large snappers and groupers, as well as large piscivorous predators (like jacks, and barracudas), densities and numbers could be much higher with proper management and protection than they currently are. Significant regional declines have been documented for many commercially exploited and threatened species.

Assessment of National Conservation State

Assessment of Natural Distribution: Favourable
None of the exploited species or other reef fish species are limited in distribution to one or more islands but are widespread throughout the Caribbean. There are also no areas around the islands that have become inaccessible to these species. However, many of the ecologically important large predatory fish species are scarcely present in large areas. This is largely due to overfishing and a decline in local habitat quality and availability.

Assessment of Population Size: Unfavourable – inadequate to Unfavourable - bad (depending on the island and the species)

Pelagic Fish: Most large pelagic predatory fish species are highly migratory and are fished throughout their range. Species such as Blue and White Marlin, and Yellowfin and Bigeye Tuna fall into this category. These species are subject to high fishing mortality (outside the Dutch Caribbean), and their biomass is below target levels in regional fishery management plans (FAO, 2011). However, there are some species that are important (or could become important) for local fisheries and are not, or possibly not, overfished. These include Blackfin Tuna (Buni pretu), Mahi-mahi (Dradu), Wahoo (Mulá) (Die, 2004), and the Blue Runner (Grastelchi' laman) (Smith et al., 2015).

Coral Reef: Large coral reef predatory fish such as groupers and snappers constituted the majority of catches until the 1960s and were still common until the 1980s. However, these large species are now virtually absent from current catches and fish counts. FAO (2011) characterizes these species as regionally overfished in the Caribbean. Many of these species are listed as threatened or vulnerable on the IUCN Red List. The least-affected fish stocks are found in the waters around Saba and the Saba Bank. Based on satellite images, the Saba Bank appear to show higher chlorophyl densities than the surrounding waters (Meesters, pers. comm.), indicative of nutrient enrichment due to upwelling which is commonly associated with the "island mass effect" (De Falco et al., 2022). Around Bonaire and St. Eustatius, many species of coral reef fishes are overfished (e.g., Sandin et al., 2008).

Deepwater: Very little is currently known about the ecology, fishing pressure, and population dynamics of relevant species (such as snappers). For the Saba Bank "redfish" fishery, the stable CPUE in recent years suggests a stable (yet low) population (Fig. 6). Deepwater snappers are notoriously susceptible to overfishing due to their predominantly slow growth and age of maturity. The CPUE is much lower than measured in 1970, which may indicate a structural, long-term, yet stable, decrease in population size. The catch of predominantly undersized Silk Snapper is inherent to fishing with fish traps with a small mesh size and selectivity for small fish. Since the introduction of fishing with fish traps, it is certain that most landed snappers are juveniles. Whether this effect has an impact on the recruitment of young fish through "recruitment" overfishing is not known. The overall conclusion is that the current population size seems favourable, but these species are also known elsewhere to be sensitive to overfishing.

Assessment of Habitat: Unfavourable – inadequate to Unfavourable - bad (depending on the island)

Pelagic: There is little to say about changes in the pelagic habitat other than that surface waters have likely become noticeably warmer due to climate change. The potential effects of this on fish populations are largely unknown. There are indications that migration patterns of certain fish, such as Mahi-mahi, which are strongly dependent on ocean temperature (Kleisner, 2008), could shift northward. With climate warming, it is expected that the productivity of tropical and subtropical marine ecosystems will decrease (Bari and Cochrane, 2011).

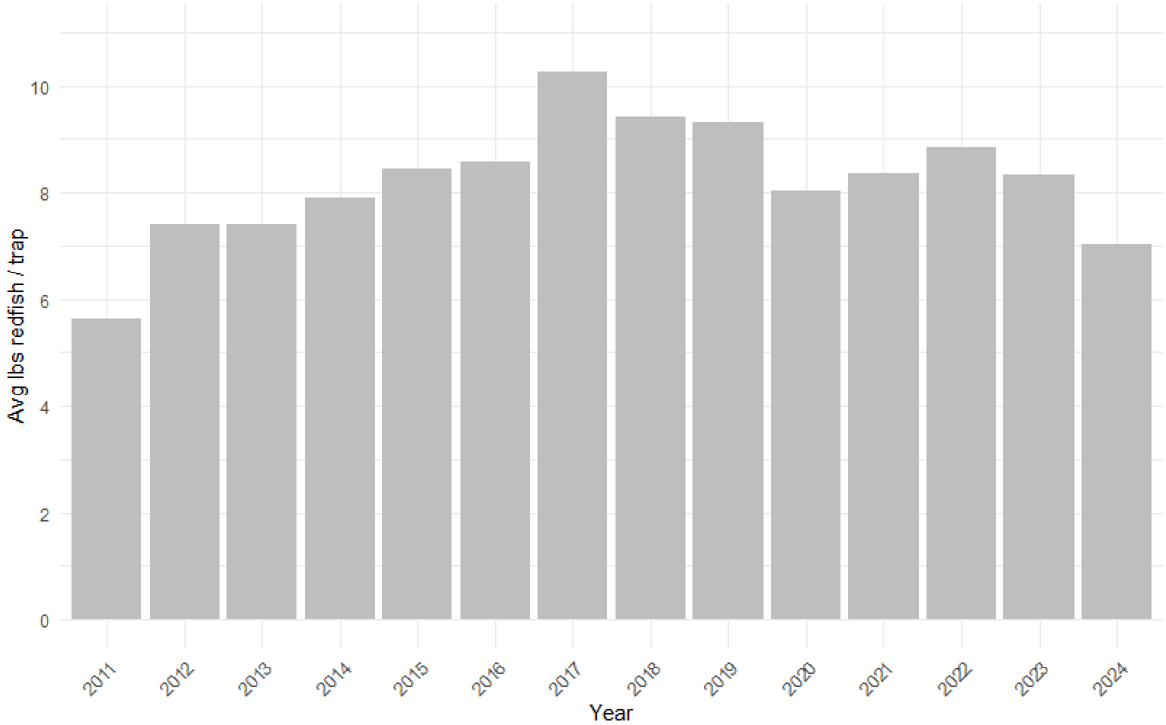

Figure 6. *Development in Catch Per Unit Effort (in terms of average catch per trap) for Redfish on the Saba Bank, 2011 to 2024.*

Coral Reef: Except for the Saba Bank, coral reefs mainly occur in a very narrow band of up to a few hundred meters around the islands. This makes them very vulnerable to influences from land. Throughout the Dutch Caribbean, including the Saba Bank, reefs have deteriorated significantly in recent decades (Bak et al., 2005; Debrot et al., 2014; Newman et al., 2015; Toller et al., 2010). This is due to a combination of factors, including both natural and human factors (Meesters et al., 2010). Studies by Hylkema et al. (2020, 2023) explore and compare the utility of different types of artificial reefs to enhance shelter availability for juvenile fish to increase fish production.

Mangroves and Seagrass Beds: Mangroves and seagrass beds are functionally part of the coral reef ecosystem and are crucial as nursery areas for numerous coral reef fish species, including species of great commercial importance (Debrot et al., 2012; Hylkema et al., 2014). Mangroves are only found on Bonaire in the Dutch Caribbean. On this island, the survival of both mangroves and seagrass beds is threatened by the siltation of Lac Bay (Debrot et al., 2019). Additionally, the native Turtle grass *(Thalassia testudinum)* seagrass beds of both Bonaire and St. Eustatius, and possibly even Saba, are threatened by the invasive seagrass *Halophila stipulacea* (Becking et al., 2014; Willet et al., 2014). The Turtle grass fields of Bonaire and St. Eustatius are in rapid decline due to the *Halophila* invasion. On the Saba Bank, the unique algal fields may play the ecological role of the seagrass beds near the islands but this should be further investigated.

Benthic and Coral Fish Stocks: The benthic and coral fish stocks are seriously threatened by the rapid increase in the numbers of the Lionfish (*Pterois volitans*), an invader in the Caribbean from the Pacific (Debrot et al., 2011). While the removal of Lionfish on Bonaire and Curaçao seems to have localized and temporary effects (De Leon et al., 2013), Barbour et al. (2011) predict, based on models, that structural removal of between 35 and 65% of Lionfish is necessary to curb their population growth. They have also shown that Lionfish can recover very quickly after removal from an area, probably due to recruitment from upstream areas where there is no active removal. Lionfish remain massively abundant on the reefs of neighbouring Curaçao as recently as 2023 (Debrot, pers. obs.) However, for the Saba Bank a large population increase, followed by a crash has recently been documented (Debrot et al., 2023). How trends will continue with the Lionfish are not known.

Deep Waters: Due to the predominantly steep island slopes of the islands, there is no continental shelf, and therefore little deep shelf area suitable as habitat for deepwater snappers. The relatively small populations mean additional vulnerability to overfishing. Only on the Saba Bank is the fishing area for deepwater snappers considerable, possibly up to 350 km^2 (Toller and Lundvall, 2008).

Assessment of Future Prospects: Unfavourable - bad (apart from a few small exceptions)

Apart from a few exceptions and bright spots, the future prospects of fish stocks is worrying.

- For the Saba Bank and St. Eustatius, there is a modest multi-year series of catch data on principal species, however, in general there is a pressing lack of basic knowledge needed for scientifically based management, and monitoring data are very limited and of questionable quality. This makes it extremely difficult to unequivocally demonstrate developments in fish stocks and their possible causes. Trends in catches and environmental correlates alone can never demonstrate cause and effect, which is the knowledge that is really needed for science-based management.
- There are very few resources and legal management measures available to limit fishing pressure on threatened and/or rare species. Most of these species are not formally protected.
- Significant declines have occurred in coral reef coverage, leading to decreased habitat quality for coral reef species. Natural habitat restoration will not offer a quick solution due to the slow growth of corals. Proactive interventions seem necessary and should be studied as a means to restore or improve degraded habitats and thereby maintain or restore fish stocks (coral farming, artificial reefs, mangrove restoration).
- Bright spots:
- Surveys indicate that the queen conch population of St. Eustatius is healthy enough to sustain limited exports (de Graaf et al., 2014). Also, on the Saba Bank, the density of queen conch appears to have increased again after foreign boats were banned from accessing the bank (since 1996) (de Graaf et al., 2017). This protection measure seems to have helped, and the stocks remain unfished.
- Overfishing is likely one of the main causes of the evident decline in fish stocks (or parts thereof) in Bonaire, St. Eustatius, and the Saba Bank. However, there are several species that have been minimally exploited or not yet overfished but have the potential to significantly contribute to the sustainable economic development of the islands. The opportunity to shift fishing pressure to alternative species likely exists but needs to be investigated more fully.

Table 3 provides a synoptic overview of the Conservation State of the fish stocks as a whole as well as separately per area.

Table 3. *Summary overview of the status of the fish stocks of the Caribbean Netherlands in terms of different conservations aspects.*

Caribbean Netherlands as a whole	
Aspect	**2024**
Distribution	Favourable
Population	Unfavourable - bad
Habitat	Unfavourable - bad
Future prospects	Unfavourable - inadequate
Overall Assessment of Conservation State	**Unfavourable - bad**

Bonaire	
Aspect	**2024**
Distribution	Favourable
Population	Unfavourable - bad
Habitat (reef)	Unfavourable - bad
Future prospects	Unfavourable - inadequate
Overall Assessment of Conservation State	**Unfavourable - bad**

Saba	
Aspect	**2024**
Distribution	Favourable
Population	Favourable
Habitat	Unfavourable - inadequate
Future prospects	Favourable
Overall Assessment of Conservation State	**Unfavourable-inadequate**

St. Eustatius	
Aspect	**2024**
Distribution	Favourable
Population	Unfavourable - bad
Habitat	Unfavourable - bad
Future prospects	Unfavourable - inadequate
Overall Assessment of Conservation State	**Unfavourable - bad**

Saba Bank	
Aspect	**2024**
Distribution	Favourable
Population	Favourable
Habitat	Unfavourable - inadequate
Future prospects	Unfavourable - inadequate
Overall Assessment of Conservation State	**Unfavourable - inadequate**

Comparison to the 2018 State of Nature report

Overall, in comparison to the 2018 assessment, no major changes can be meaningfully identified for the CS of the various fish stocks of the Caribbean Netherlands.

Key Threats and Management Implications

Globally, many fish stocks suffer from overfishing and ineffective management. This is also the case for many species within the Caribbean Netherlands, where the overall fish population has demonstrably declined over the past decades due to a combination of factors. Many species have even almost completely disappeared and are in urgent need of protection.

Table 4. *Overview of key threats to the fish stocks of the Caribbean Netherlands and implications for their management.*

Core Threats		**Management Implications**
Habitat degradation:	The coral reef ecosystem has been rapidly deteriorating over the past decades, resulting in the loss of habitat three-dimensional structure and nutrient cycling function. This is due to a combination of factors including eutrophication, sedimentation from land erosion, acidification, and ocean warming as a result of climate change.	• Reduction of livestock densities on land • Control of nutrient flows from land • Wastewater treatment • Integrated coastal management • Use of habitat restoration measures, such as coral farming and artificial reef construction.
Overfishing:	Overfishing is a direct threat to many endangered species, especially larger predatory fish.	• Integrated fisheries management. • Legal protection for endangered fish species • Development of alternative fisheries • Development of industries based on zero extraction of fish (dive tourism, catch and release sportfishing, scientific tourism
Invasive Species:	Invasive species such as exotic fish and seagrass species cause significant ecological disruptions.	• Implementation of an Invasive Alien Species Strategy (IASS).

Climate Change:	Predicted ocean acidification and warming will have major consequences on ocean currents and the quality of surface water, which is crucial for the development of fish larvae and the dispersal and migration of species.	• Active participation in international forums and advocacy against climate change

National Management Objective:
Determine and implement management measures to prevent further decline in fish stocks and potentially restore them. This is not only desirable from an ecological perspective but also essential for maintaining long-term fisheries for the local economy. The NEPP for the Caribbean Netherlands assigns a high priority to developing fisheries towards sustainability (Min. LNV et al., 2020).

Sub-goals for fish stock management:

- Overfishing remains one of the primary threats to fish stocks in the Caribbean Netherlands. There is an urgent need for monitoring and knowledge development to take targeted actions and to assess management measures aimed at sustainable conservation and management. Both fisheries-dependent and fisheries-independent studies will be needed for this.
- There is a need to reduce fishing pressure on heavily overfished large predatory reef fish through protective measures. These may include the effective designation of fish reserves, spawning season moratoria and size limits for capture.
- Most fish stocks are part of larger regional stocks, necessitating international management coordination. Therefore, full participation in international forums (Regional Fisheries Management Organizations) such as ICCAT and WECAFC is recommended.
- To help mitigate the negative impact of fishing, fishermen should be assisted in targeting new and alternative fish and shellfish species (e.g., deepwater squid, crabs, flying fish, and various pelagic predatory fish species that are not overfished). However, consideration must be given to the ecological role of targeted new fish stocks.
- Measures should also be taken to preserve and enhance habitat quality (both passively through regulations and proactively through habitat interventions; mangrove and coral reef restoration). Without suitable habitat, there will be little fish needing protection.
- Non-extractive use of fish populations should take precedence over extraction. This includes ecotourism, catch-and-release sport fishing, and scientific tourism.

Data quality and completeness

The available data on the status of fish stocks in the Caribbean Netherlands are highly fragmented and incomplete. Most studies only involve limited visual counts of larger, non-cryptic diurnal reef fish. Little to no information exists about other components of the fish fauna (night fishes, deepwater fish, pelagic fish stocks, small and cryptic species). There is also limited data on the impact of fishing. While available studies indicate overall overfishing and allow for some general recommendations, there is still a significant lack of basic scientific knowledge necessary for scientific management. Additionally, annual monitoring, necessary for thorough assessment and coordination of fisheries management, is partly lacking. Fortunately, monitoring of fishing and catches is now structurally funded by the Ministry of Agriculture, Fisheries, Food Security and Nature for Saba Bank and St. Eustatius, and as of 2023 also for Bonaire. This offers excellent prospects for the future. Diligent monitoring can gradually expand the data sets, enabling informed insights into fish stock developments. The recent reorganization of the SBMU also provides great promise for better data and greater understanding of trends and their causes so that better management comes closer within reach.

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Part 3: Threats to Biodiversity

As explained in the introduction, this report also discusses several of the major threats to nature separately. The state of certain far-reaching threats largely determines the Conservation State of nature and requires an integrated approach instead of an individual "species" or "habitat approach." The discussed threats are often problems that simultaneously have a significant negative impact on many species and/or habitats.

These include topics such as "invasive species," "free-roaming livestock," "overfishing," and "climate change," which require a separate, structured, and integrated approach, aside from policies focused on individual species and/or habitats. The issue of overfishing is already extensively discussed in the section on fish stocks and is not covered separately here, although doing so in the future is certainly recommended. Additionally, factors such as coastal development, erosion, and eutrophication due to wastewater should not be overlooked (e.g., Debrot and Sybesma, 2000). These issues are also not addressed here.

For many of the mentioned threats milestones have been established and are already significantly being addressed within the implementation agenda of the current NEPP (Min. LNV et al., 2020). For instance, these include milestones for a) the prevention and control of invasive species, b) the control of free roaming livestock, c) effective waste and wastewater management, d) investments in sustainable fisheries, e) coral reef restoration, and f) the conservation of keystone and flagship species.

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26 Roaming Livestock: a Threat to Caribbean Netherlands Climate Resilience and Biodiversity

Debrot, A. O., Bertuol, P., van Slobbe, F., Eckrich, C. and Francisca, R. 2025. From: State of Nature Report for the Caribbean Netherlands, 2024. WUR report C001/25.

Status

In the Caribbean Netherlands, overgrazing by introduced free-ranging livestock (especially goats but also donkeys, cattle, and increasingly pigs) is considered the most severe threat to terrestrial ecosystems (Min. LNV et al., 2020; Smith et al., 2014) and to climate adaptation. Some of the earliest authors to point this out and summarize the major negative impacts this has on vegetation, soil, and the nearshore ecosystem date from the middle of the last century (Duclos, 1954; Westermann and Zonneveld, 1956; Hoetink, 1969). For instance, Hoetink (1969, p397) comments on how uncontrolled grazing and extensive agriculture was accompanied by soil erosion, dust and heavy sedimentation of the inland bays and reefs (Figure 1). This process continues to this date and has been identified as the major reason for the loss of critical lagoonal seagrass habitat in Lac Bay (Debrot et al., 2019). Apart from being a direct threat to native vegetation and rare plants, overgrazing has many other adverse ecological, social and economic effects. In the past, several largely unsuccessful attempts were made on all three islands to address this problem. However, efforts remain ongoing with recently much better success on both Saba and in the WSNP, Bonaire. The NEPP for the Caribbean Netherlands assigns a high priority to culling and controlling roaming livestock(Min. LNV et al., 2020).

Figure 1. *Muddy brown freshwater with eroded sediment flowing onto the coral reefs of the National Marine Park of Bonaire from Saliña di Vlijt during heavy rains in 2004. Photo: F. van Slobbe*

With the exception of Klein Bonaire, and recently Saba, and a section of Slagbaai (amounting to a total of 318 ha), where all invasive grazers have been removed, goat densities in all other natural areas of the Caribbean Netherlands have remained too high for sustainability (St. Eustatius: 5.9/ha; Bonaire, entire island: 1.4/ha; Bonaire Washington-Slagbaai National Park (WSNP): 1.1/ha) and pose a very significant threat to ecosystem function on all islands of the former Netherlands Antilles (Coblentz, 1980). Due to centuries of overgrazing, the original groundcover of bromeliads and orchids has for the most part degraded to a vegetation now dominated by cacti and thorny plants (Debrot and de Freitas, 1993). This has significantly changed the structure, appearance, water management, and even the insect fauna of the forests (Debrot et al., 1999). However, by reducing livestock densities to 1 goat or sheep per 10 hectares, it has been empirically demonstrated that reforestation and rare plants can recover quickly (Figure 2).

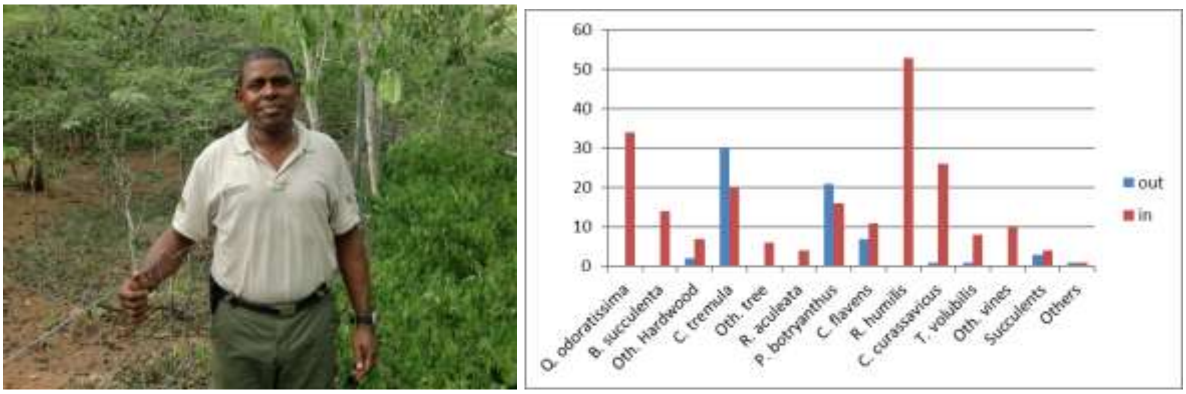

Figure 2. *Left: Park manager Juni Janga (2006) showing the sharp contrast between grazed and ungrazed habitat in the Washington-Slagbaai Park of Bonaire only three years after grazer exclusion; and right: the associated rapid increase in plant species composition. (Photo and data: A. O. Debrot). Full plant names from left to right: Quadarella odoratissima, Bourreria suculenta, other hardwood Casearia tremula, other trees, Randia aculeata, Phyllantus botryanthus, Croton flavens, Rivina humilis, Croton ovalifolius, Tournefortisa volubilis, other vines, Succulents, Others (plants).*

Description

Livestock farming in the Caribbean Netherlands has always been extensive. During the 18th and 19th centuries, various measures were enacted to protect forests and pastures from erosion and overgrazing (Van Grol, 1942; Westermann and Zonneveld, 1956), but these measures were never enforced and only applied to public domain areas (land owned by the island government), not to the large private plantations (De Freitas et al., 2005). The livestock roamed freely and reproduced in the wild. There was no pasture or herd management (Hoetink, 1969). Only a few larger plantations had some level of pasture management, with fenced livestock paddocks (Hoetink, 1969). According to Hoetink, small-scale farmers send their goats onto public land, “where they must find their own food.” This system of livestock farming continues to be largely maintained to this day. After arrival of oil refinery on Curaçao, growing economic prosperity heralded an end to the plantation era and small-scale agricultural fields became largely abandoned by the middle of the century (Hoetink, 1969, p525). With the declining interest in agriculture, and less need to maintain fencing, not only public but also the private domain became fully accessible to uncontrolled feral livestock grazing. Further, with the advent of motorized transportation, donkeys, that once had been an integral part of domestic life were abandoned to fend for themselves. This too became an important factor in further overgrazing especially on Bonaire and St. Eustatius. However, the feral grazer with the highest impact remained to be the goats.

Goats are among the most adaptable livestock species and can thrive in almost any environment. Goat populations can grow very quickly. Without specific controls, a goat population can increase by 60-75% per year (GSA, 2005). Parkes (1984) calculated an annual natural population growth rate of 0.424 for a healthy hunted population on Raoul Island in the Pacific, indicating that such a population could double every 20 months, complicating population management. Under even the poorest conditions, goat populations can also slowly grow (e.g., Southwell and Pickles, 1993). The observed rate of population growth depends on age-specific fertility and mortality, which are influenced by factors such as food availability, animal health, fertility, and sex ratios. Given the relatively good health and fertility and the suspected surplus of females in the Slagbaai area on Bonaire (Geurts, 2015), doubling times of 1-1.5 years are likely.

The consequence of this is that to achieve a real population reduction, perhaps 50% of all goats would need to be removed annually. For the average goat in Bonaire, we use the theoretical approach by Caughley and Krebs (1983) with a natural growth rate of 0.38, or 31% population growth per year. This would mean that to achieve a reduction in population growth, over 31% would need to be removed per year (Debrot, 2016; Figure 3).

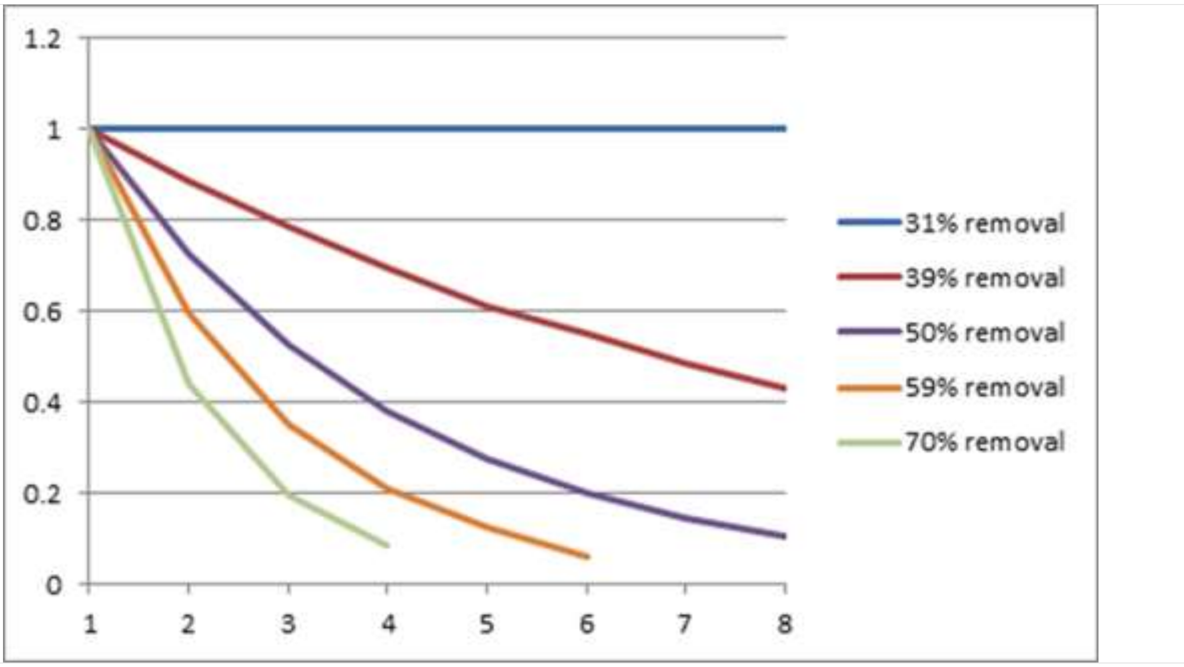

Figure 3. *Fractional population declines achieved year by year from the start of year 1 for different (discrete) culling rates. From: Debrot 2016.*

Effect of free-roaming livestock within the Caribbean region: massive
On the islands of the Caribbean Netherlands, the extensive form of livestock farming based on free-roaming livestock is a marginal source of income for a limited number of farmers (Neijenhuis et al., 2015). However, this form of livestock farming represents a major collective social cost (Neijenhuis et al., 2015), manifesting in:

Uncontrolled erosion with loss of topsoil (Westermann and Zonneveld, 1956);

Loss of biodiversity, lowering of groundwater levels, desertification, ambient temperature increase.

Damage to coral reefs and fisheries (Fabricius, 2005);

Damage to agriculture (the need for fencing);

Dust damage and nuisance (Nolet and van de Meer, 2009);

Traffic risks and damage (Bonaire Police Department, Division of Motor Safety); and

Damage and plundering by free-roaming livestock often poses a major obstacle to agricultural development (Neijenhuis et al., 2015). This problem is shared with much of the Caribbean region, where free-roaming livestock not only causes loss of biodiversity but also damages commercial crops and public green spaces (Grenada Govt, 2007; Rijo, 2014). The argument is occasionally made that roaming goats at least form some kind of island self-sufficiency in food production. Nothing seems farther from the case. The combined damage of excessive densities of roaming goats to soil and vegetation is clearly a major detriment to island food self-sufficiency (Neijenhuis et al 2015; Lotz et al. 2020). This is compounded by topsoil runoff, aridification, lowering of the groundwater levels resulting in poor soils that are leached out. Fencing and continuous maintenance is required to keep goats away from crops, which is relatively expensive.

To paraphrase Winston Churchill on the matter of goats:

"Seldom do so few benefit so little at the expense of so many"

Ecological Characteristics

Impacts
Since the early 1950s, the negative effects on ecosystems due to overgrazing by uncontrolled free-roaming livestock have been well-documented (Gilliland, 1952; Kolars, 1966; Pisanu et al., 2005; Bakker et al., 2010; Müller et al., 2011). Coblentz (1977 and 1978) was one of the first authors to discuss the sensitivity of island ecosystems to exotic livestock. Since then, many others have demonstrated the extremely harmful effects of exotic grazers on island ecosystems (Gould and Swingland, 1980; Debrot and De Freitas, 1993; Fernández-Lugo et al., 2009; Carrion et al., 2011). In a recent global evaluation of 251 campaigns for removing invasive mammals on islands, it was concluded that such removals almost always led to rapid and effective ecosystem restoration (Jones et al., 2016).

On Bonaire, the driest of the three Caribbean Netherlands islands, the situation with regards to the free-roaming livestock remains the most acute. Many tree species can no longer regenerate because the seedlings do not survive the grazing pressure (Debrot et al. 2018). For this, the Bonaire government has published a list of trees that are protected by law (OLB 2008). It is likely that native species on Bonaire have already gone extinct, and others will follow in the coming decades if no effective measures are taken (Lo Fo Wong and de Jongh, 1994; Proosdij, 2012; Freitas et al., 2005; Debrot et al., 2018). While the problem has long been recognized (Anonymous, 1985, 1989, 2006, 2009), few actual measures have been implemented so far. Of concern is how free-roaming goats and donkeys strip the bark from the columnar cacti, which leads to the death of these critically important trees. Columnar cacti bloom and bear fruit during the dry season, when most deciduous trees are bare, providing an essential food source for fauna ranging from birds, and bats to reptiles during the dry season (Petit, 1997).

Maximum permissible population densities
Various studies provide insights into the carrying capacity of semi-arid ecosystems for livestock. For semi-arid regions of Australia, densities of 0.1 goat per hectare are already considered a severe threat to the environment and agricultural productivity (Southwell et al., 1993; Southwell and Pickles, 1993). On the semi-arid island of St. Catalina, off the coast of California, the island's natural vegetation was already heavily depleted and overgrazed at densities of 0.25 goats/ha (Coblentz, 1977). On Pinta Island, in the Galapagos, a density of 1.69 goats/ha was considered excessive, and the vegetation and unique flora quickly recovered after the goats were removed (Hamann, 1993). In arid parts of Australia, Pople et al. (1996) indicated that average grazer densities of 0.25 goats/ha already posed a severe threat to agricultural production. Lastly, Brennan et al. (1993) described the need to cull goats at a density of 0.16 goats/ha. On Curaçao, the removal of goats from the Christoffel Park to densities of approximately 0.1 goats/ha was sufficient to lead to rapid ecological recovery (Debrot and de Freitas, pers. obs.). In the Labra/Brasiel area on Bonaire, where average densities of 0.45 goats/ha were measured, ecological recovery and regeneration of vulnerable species are not evident, indicating that goat density must be reduced below 0.45 goats/ha before ecological recovery is possible. Based on results from the nearby and comparable island of Curaçao, it appears that livestock densities of 1 goat per 10 hectares are sufficiently low to enable rapid ecological recovery, including the recovery of many rare species (Debrot, 2015). Of course, complete eradication (i.e., density is zero) is the best possible scenario. However, even where people no longer keep goats, total eradication is difficult at best and if goats are allowed to be legally kept on an island, eventual strays are practically impossible to prevent. Aside from islands like Klein Bonaire and Klein Curaçao where eradication has been

complete and the keeping of goats is not allowed, in practice this means that culling goats becomes a structural activity for nature management (e.g., Christoffel Park, Curaçao).

Present Distribution and Reference Values

To date, there have been very few quantitative studies on livestock density and distribution on the islands of the Caribbean Netherlands. Only recently have quantitative livestock counts been conducted on St. Eustatius (Debrot et al., 2015, Madden, 2020) and Bonaire (Lagerveld et al., 2015; Geurts, 2015). For Saba, there have as yet only been expert estimates, with no formal livestock counts.

Bonaire: Lagerveld et al. (2015) conducted livestock counts for Bonaire in 2014. Based on 75 line transects of 500 meters, the first quantitative estimates of the island's goat population were made. They used the so-called "Distance method," a modern, widely recommended, and accepted method to estimate animal density in natural areas. About 50% of the animals are found in agricultural areas, 12% in coastal areas, 37% in areas with thorny vegetation, and 1% in urban areas. In the forest, the density of goats was highest in the WSNP, where the lowest density would be expected based on its designation and management as a natural area.

For the entire island, a total of about 32,200 goats was estimated, with the number of animals in forested areas (about 12,000) possibly underestimated. The goat counts align with expectations based on previous professional qualitative estimates. However, despite considerable research effort (75 transects), there's a relatively large margin in the estimated minimum and maximum numbers. The estimates yield densities averaging 1.41/ha (minimum 0.86 and maximum 2.30). This is much higher than what is sustainable for extensive livestock farming. A new form of sustainable livestock farming is therefore recommended, not only to provide real opportunities for the sector but also to reduce the negative ecological and economic consequences of the current situation.

For the Slagbaai plantation, located within the WSNP, the density of goats was estimated at 2.69 goats/ha in 2014 (Geurts, 2015). This density is far above what is sustainable in unmanaged semi-arid natural areas (Geurts, 2015; Debrot, 2016). As a result of a prolonged lack of livestock management in this nature reserve, the vegetation of Slagbaai is among the sparsest and impoverished of all natural areas on Bonaire, with many tree species threatened with extinction (Freitas and Rojer, 2013). However, STINAPA data, as collected by Rivera-Milán et al. (2018, 2020, 2021, 2023) show that current efforts are making considerable headway in reducing livestock densities inside the park.

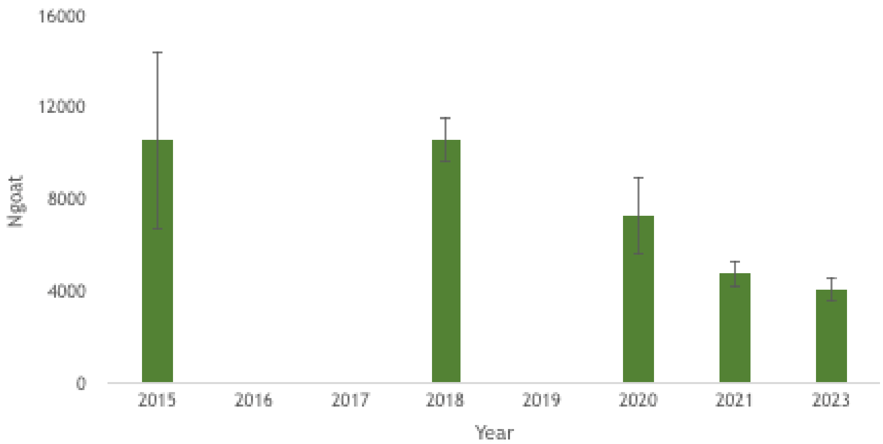

Figure 4. *Recent trends in estimated goat numbers inside the Washington-Slagbaai National Park. Based on counts by Rivera-Milán et al. 2018, 2020, 2021, 2023.*

The management authority (STINAPA) of WSNP started in 2014 with active measures to remove all goats from the park. After different approaches did not achieve the objectives, a combination of measures was adopted. This includes trapping and shooting whilst fencing and compartmentation of the area. The has led to fenced off sections of Slagbaai being currently completely goat free. In goat-free areas an increase in seedling densities has been documented (Fig. 5). STINAPA has bought out the remaining grazing rights for the Washington section of the WSNP.

Feral pigs

Feral pigs (*Sus scrofa*) are a major scourge to nature conservation worldwide (Risch et al., 2021). As habitat and food generalists they are extremely adaptable and because of their extreme fecundity they can overpopulate very quickly (Echo, 2019). The damage they inflict to fauna, flora and ecosystems can be summarized as follows: they damage soil and vegetation by consuming seedlings and digging up tree roots and have a major impact on small soil and ground-inhabiting fauna such as lizards, geckos (including their eggs), soil arthropods and ground-nesting birds like the endangered endemic White-tailed nightjar (*Caprimulgus cayenensis insularis*), endangered terns and gulls and iguana and sea turtle nests. In the WSNP feral pigs have uprooted and destroyed most of the formerly extensive historical aloë (*Aloë barbadensis*) fields (A. Debrot, pers. obs.). Due to their compact strength, they destroy even the best fencing, and it is impossible to fence them off crops or out of unguarded gardens. In addition, they also constitute a traffic hazard and if with piglets and cornered, can be hazardous to be nearby. The only proper way to keep pigs is enclosed behind special fencing. Between 2016 and 2019 a government sponsored culling project resulted in the capture and euthanisation of 175 swine.

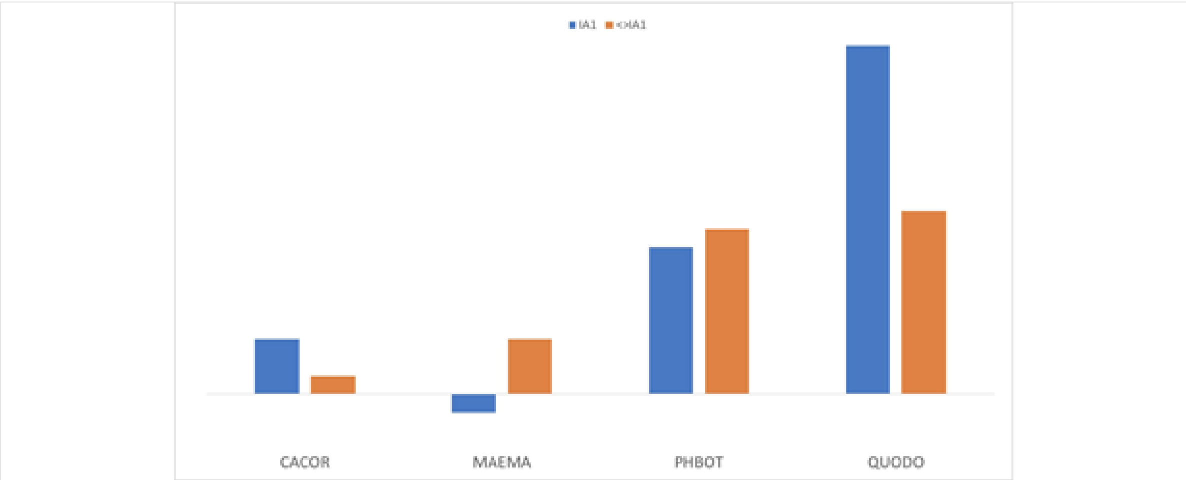

Figure 5. *A comparison of seedling densities before (brown) and after (blue) inside goat exclosure "Area 1" in the Slagbaai plantation (end 2024). CACOR = Caecalpinia coriaria, MAEMA = Malpighia emarginata, PHBOT = Phyllanthus botryanthus, QUODO = Quadarella odorata.*

Feral Donkeys

Feral donkeys (*Equus asinus*) are not native to the island of Bonaire but can have massive impacts on island ecosystems and biodiversity particularly in arid ecosystems (Malo et al., 2011; Symanski, 1996). During the colonial epoch and up to the mid-20th century they were a key form of transportation but were subsequently abandoned to the wild ("set free") to fend for their own (Hartog, 1954) and population densities have grown unchecked.

"The donkey (and mules) had an important place in the economy, in transportation, was part of the family, had its own technology and jargon, its own folklore (Fig. 6). Most donkeys belonged to people. The large plantation owners also always had some stray feral donkeys, but they were not [so abundant] like they are now" (Bòi Antoin, in interview with Dòlfi Debrot, 2014).

Figure 6. *"How it used to be". Plantation owner Shon Willem Schotborgh on his favourite mule at Sabaneta in the 1950s, Curaçao (A. Debrot, family photos). Equines (mostly donkeys, some mules) were an integral part of the family household but after WWII and the surge in motorized transportation, they were mostly abandoned ("set free") to fend for themselves.*

For biodiversity and resilience against climate change, population control actions for feral donkeys in Bonaire are highly needed. Recognizing this, the island government, in collaboration with various local parties, initiated a program in 2012 to remove free-roaming female donkeys from the wild and sterilize male donkeys. A total of 204 males were sterilized, earmarked and returned to the wild, 97 were euthanized as they were sick and suffering and 326 female donkeys and foals were adopted into the donkey sanctuary. This program was discontinued in 2014.

In 2020, Simal et al. (2020) estimated donkey density for the whole of Bonaire at 0.043 donkeys ha-1 and population size at 1084 donkeys. Donkeys graze differently than goats but highly damaging (Malo et al., 2011) so control of donkeys and preferably limiting donkeys to (as formerly) a controlled, captive and properly cared-for registered population is to be preferred. Aside from a donkey capture program for the island as executed by the local government (2012-2014), the only area where donkeys remain being culled is in the WSNP where the density of donkeys decreased from 0.119 to 0.018 individuals per hectare between 2018 and 2023. (Rivera-Milan et al., 2023).

St. Eustatius: In 2013, counts were conducted on St. Eustatius (Debrot et al., 2015). The densities of goats, cattle, sheep, and chickens were estimated over a total transect length of 33.5 km, along existing nature trails in six different habitat zones. Each of the 13 different trails was visited and counted five times. The results indicate that the density of especially chickens, cattle, and goats was high. Statistically significant differences in density between the different habitat zones could be demonstrated. Based on the counts and species-specific detection curves, the island's population estimate (± standard deviation) for goats was 2,470 ± 807. More recently, Madden repeated livestock density estimations using the same distance method and concluded a much higher goat population (7,602 ± 1,555) for the island as a whole and a more than fourfold increase in goat densities in the island's protected areas between 2015 and 2020 (Madden 2020). Recognizing that the situation seems to be spiralling out of hand, STENAPA has started to try to fence off some of the most critical vegetation areas around the Quill.

The estimated densities are far too high for the sustainable management of vulnerable semi-arid grasslands. At these densities, there is a loss of organic matter in the soil, reduced water retention capacity, and increased erosion. It is crucial that the livestock population is limited and better managed. Of all livestock species, goats are the most problematic because they have a strong preference for rough terrain. Such areas are much more vulnerable to erosion and house higher densities of rare species dependent on scarce microhabitats.

Saba: So far no livestock counts have ever been conducted on Saba. Many short-lasting culling efforts have been attempted in the past, but these generally met with strong local resistance. Fortunately, following a long history of start-up difficulties, radical culling has been taking place since 2020 with an estimated 90% reduction in the feral goat population (Public Entity Saba, pers. comm.). Due to the high fecundity of goats, and practical impossibility of total eradication (as sometimes possible on uninhabited islands), goats can bounce back in numbers very quickly and culling needs to be seen as a structural conservation measure that needs to continue steadily and in the long term. Until recently, average densities on Saba seemed to have been above 1 goat per hectare, but densities are said to have dropped by around 90% since 2020 (PES, pers. comm.). A quantitative assessment would be very good to conduct. On this island, to an untrained eye, the damage is less noticeable because it is generally much greener than St. Eustatius and Bonaire but based on the most recent assessment (Janssen and Proosdij, in this collection of reports) the vegetations of Saba are in unfavourable-bad CS and all show declining species richness since the last assessment. The gross impact of overgrazing is still strongly observable on the lowest slopes of the island, where these animals have especially numerous and significantly contribute to erosion (Debrot and Sybesma, 2000). The former lowland forests have been unable to recover since the *Tabebuia* die-off that took place in the 1980s (Freitas et al., 2016) and it seems certain that the high goat densities in the island's drier coastal areas are a major culprit in preventing forest regeneration.

Key Threats and Management Implications

Trends
Quantitative livestock counts have only recently become available for St. Eustatius (Debrot et al., 2015) and Bonaire (Lagerveld et al., 2015; Geurts, 2015). No livestock counts have been conducted for Saba, so it is impossible to say with certainty how livestock densities have changed on these islands in recent years. However, it can be assumed that the current livestock densities are indicative of structurally excessive densities on all three islands, something that older, often non-quantitative sources have already warned about (Duclos, 1954; Westermann and Zonneveld, 1956; Debrot and Sybesma, 2000).

Recent Developments
Several key recent developments can be listed as follows:

- On Bonaire, STINAPA, in collaboration with the island government and with funding from the Dutch Green Fund (currently the Nature and Environment Policy Plan – NEPP 2020), has been working on a project since 2014 to reduce the goat population within the WSNP (OLB/STINAPA 2014).
- On Bonaire, the island government, in collaboration with various local stakeholders and Wageningen Livestock Research, had been working to develop sustainable livestock farming options but this initiative has been discontinued.
- On Bonaire, the island government, in collaboration with various local stakeholders and Wageningen Livestock Research, is assessing to plans to fence off and remove goats from biological hotspots and sensitive areas such as Lac.
- On Bonaire, the island government, in collaboration with various local parties, initiated a program in 2012 to remove free-roaming female donkeys from the wild and sterilize male donkeys. This program was discontinued in 2014.
- On Bonaire, the Island government, in collaboration with local foundations a program was initiated in 2016 to eradicate feral pigs and prevent owners to allow them roam freely. Over 175 feral pigs were captured and euthanized. The program terminated in 2019.
- On St. Eustatius, the LVV department is implementing a structural program to enclose free-roaming donkeys and capture and slaughter free-roaming cattle.
- On Saba, radical culling of goats has been taking place since 2020 with an estimated 90% reduction in the feral goat population, as per 2024. (Public Entity Saba, pers. comm.).

Assessment of the Effect of Distribution on Biodiversity: Unfavourable-bad

Apart from the island of Klein Bonaire and a section of Slagbaai, goats are found everywhere in the nature areas of the Caribbean Netherlands. Notwithstanding some (short-lasting) initiatives as indicated previously for Bonaire, feral donkey densities contribute significantly to feral grazer densities (Simal et al., 2020) and highly fecund feral pigs are no longer being controlled (Echo 2019). On Bonaire, only in the WSNP are donkeys still being removed but they remain abundant in the Brasiel-Labra part of the park (F. van Slobbe, pers. obs.). Fortunately, neither on St. Eustatius nor Saba are feral pigs or donkeys a major problem (K. Wulf, pers. obs., Madden, 2020). Goats are found from sea level to the top of Mt. Scenery on Saba (880 m) and even in the crater of the Quill on St. Eustatius (600 m). They can be seen on the steepest mountain slopes on all islands, where climbing seems to be a sport. The only exceptions are a few small experimental plots on the slope of the Quill and a few long-term fenced vegetation plots in the WSNP on Bonaire.

Assessment of the Effect of Population Size on Biodiversity: Unfavourable-bad

As mentioned above, assessed livestock densities have been, on average, ten times or more than what is required to allow the recovery of rare species (Debrot et al., 2015; Lagerveld et al., 2015; Geurts, 2015; Debrot, 2016).

Assessment of Habitat: Unfavourable-bad

The negative effects are diverse and severe and have been extensively discussed in various studies (Debrot and De Freitas, 1993; Debrot and Sybesma, 2000; Debrot et al., 2014; de Freitas and Rojer, 2013; Freitas et al., 2005, 2014, 2016; Debrot et al., 2019; Lotz et al., 2020; Van Proosdij et al. and Janssen and van Proosdij in this collection of reports).

Assessment of Future Prospects: Unfavourable-bad

Control of free-roaming livestock in the Dutch Caribbean has only been successful in one park on Curaçao for more than 30 years, and since recently partially successful in WSPN on Bonaire. Eradication has however been fully successful on the small satellite islands of Klein Curaçao and Klein Bonaire. Although various efforts continue, it is unlikely that, in the short term, radical change for vaster natural areas in goat control and eradication will occur (in the case of Bonaire and St. Eustatius). This could result in several plant species going extinct within 10 years and is especially acute on Bonaire. One exception appears to be the WSNP where efforts to cull livestock have been ongoing for several years now with some success. Another exception is Saba on which radical culling has been taking place since 2020 with an estimated 90% reduction in the feral goat population as of 2024 (Public Entity Saba, pers. comm.). Only if culling efforts can be sustained in the long-term, can it lead to sustainable results and long-term improvements for biodiversity. This is especially so because both goats and pigs are extremely fecund and can quickly regain population size even if only temporarily left unchecked. On all islands the parks remain surrounded by goat keepers and the chance of goats escaping to the wild and re-entering park areas cleared of goats is high. Also, the temptation of goat keepers to make holes in the park fence so that their animals can enter unnoticed, will also remain as long as they feel they can get away with it. So even if areas have been fully eradicated of goats, the need for vigilance and culling of accidental entries will remain a priority so long as goats are kept on Bonaire.

Good news is that several recent reforestation experiments indicate that recovery can be rapid and extensive once free-roaming livestock is excluded from an area (Debrot, 2013; 2015). Additionally, based on successful eradication campaigns and long-term control on Curaçao (Oostpunt and Christoffelpark) and based on cost-benefit analyses for Slagbaai on Bonaire (Debrot, 2016), if effectively implemented, this critically important conservation measure can be achieved at minimum cost as in neighbouring Curaçao (simply because the market value of a goat caught is still less than the costs and effort required for catching it).

It is important to keep in mind that before slow-growing seedling trees grow to a size beyond which they are extremely vulnerable to grazers will take more than 20 years for most tree species (E. Houtepen, Carmabi, unpublished data). Hence, continuity needs to be guaranteed, and this is very difficult considering the less-than-solid public support and scanty and unstable attention to the feral grazer matter. The culling, removal or eradication of roaming livestock is about the most important nature protection role for any park management organization to fulfil. This activity needs to be elevated to top priority.

Table 1. *Summary overview of the threat of roaming livestock to biodiversity in the Caribbean Netherlands in terms of different criteria.*

Threat level	2024
Impact on biodiversity	Unfavourable-bad
Population size	Unfavourable-bad
Habitat impacts	Unfavourable-bad
Future prospects	Unfavourable-bad
Overall Assessment of Threat Status	**Unfavourable-bad**

Comparison to the 2018 State of Nature Report

On St. Eustatius the situation seems to have gotten much worse with roaming goats. On the positive side is that all the local park organizations remain more or less active in this area, however, on Saba the lead has been decidedly by the PES. There is especially hope for positive change for the WSNP in the near future so long as culling efforts can be sustained. On Saba too, major culling has been achieved since 2020 and if this can be sustained into the future, vegetation recovery will certainly get underway.

Recommendations for National Conservation Objectives

For protected natural areas, livestock densities should be reduced to the equivalent of 0.1 goat/ha or less.
First focus on goat control and eradication in areas which hold the largest biodiversity. Ascertain that goat keepers in, or adjacent to, nature areas keep their goats fenced in.

Conservation Sub-Goals

- Implement an information campaign to educate the public on the large societal costs exacted by excess feral roaming livestock.
- Introduce flexible but structured control of livestock densities as an integral component of nature conservation.
- Establish monitoring programs to evaluate and adjust control and recovery measures.

Apart from Klein Bonaire and a section of Slagbaai, where complete removal of goats has been achieved, total eradication of goats and other livestock in the other terrestrial nature areas of the Caribbean Netherlands is currently very difficult to achieve. Reducing populations to a maximum of 1 animal (goat) per 10 ha is currently the next best alternative to total eradication. For other livestock (donkeys and cattle), it is suggested to consider an equivalence of four goats for each donkey and six goats for each cow.

Key Threats and Management Implications

The core threats caused by free-roaming livestock and the main management implications are shown in the table below:

Table 2. *Overview of key threats to biodiversity caused by roaming livestock in the Caribbean Netherlands and implications for biodiversity management.*

Core threats		Management implications
Extinction of rare plants:	High grazing pressure prevents many plants from reproducing and regenerating.	- Reduce livestock densities - Exclude livestock from critical areas through fencing and control

Core threats		Management implications
		- Ban livestock from protected natural areas
Erosion:	- Overgrazing and trampling by livestock cause large-scale erosion - Lowers groundwater levels, and dries out the soil - This leads to loss of topsoil, freshwater, and soil nutrients - Limits agricultural potential and ecosystem resilience – Causes siltation of important aquatic habitats - Sediment kills coral reefs by smothering - Creates dust that causes nuisance and damage to mechanical and electrical devices	- Reduce livestock densities - Exclude livestock from critical areas through fencing and control - Ban livestock from protected natural areas
Traffic hazard:	Collisions caused by free-roaming livestock are a significant cause of traffic accidents and fatalities.	- Reduce livestock densities - Exclude livestock from critical areas through fencing and control - Raise awareness - Develop and introduce sustainable forms of livestock farming
Infrastructural costs:	Free-roaming livestock damages plants and properties and requires costly fencing to protect green infrastructure.	- Reduce livestock densities - Exclude livestock from critical areas through fencing and control - Raise awareness - Develop and introduce sustainable forms of livestock farming

Data Quality and Completeness

A few quantitative livestock counts have become available for St. Eustatius (Debrot et al., 2015) and Bonaire (Lagerveld et al., 2015; Geurts, 2015, Rivera-Milán et al., 2018, 2020, 2021, 2023). For St. Eustatius the last five years have seen an explosive increase in the total livestock herd and a more than fourfold increase in livestock densities in protected conservation areas (Madden, 2020). No livestock counts have been conducted for Saba, however according to the PES roughly 90% of all goats are believed to have been culled between 2020 and 2024. Generally speaking, the available data is insufficient to properly evaluate ongoing initiatives and to provide a basis for adjusting the approaches to culling even if needed. Now that culling has started in earnest, it is critical to start collecting data on vegetation recovery by means of permanent monitoring quadrats as has started in 2023 in the WSNP.

The exact number of livestock to capture and cull is difficult to determine in advance. Reference numbers for natural birth and mortality rates are uncertain and influenced by local conditions, so predetermined capture and culling targets are likely not to achieve the intended goal. The effects of capturing and culling free-roaming livestock can only be determined if the total numbers are monitored. These periodic counts should define the targets for the next period.

The "Distance" method used in all three studies for livestock counts yielded wide margins of uncertainty in estimated densities, despite extensive sampling (Debrot et al., 2015; Geurts, 2015; Lagerveld et al., 2015). Thus, the ”Distance method” is not recommended as a method for estimating densities. Instead, we suggest using a simplified and standardized method as an index for population density (Debrot, 2016). The possible use of drones for livestock counts should be investigated. With the culling of feral livestock, it is not so much about livestock counts but about key vegetation metrics like vegetation cover, height and species composition. As vegetation change resulting from goat removal seems much easier and less costly to reliably monitor therefore at present WMR and Stinapa are switching to vegetation plots as a proxy of grazer impact as a way to verify grazer reductions.

Many plant species on all the islands are critically threatened in their existence due to free-roaming livestock. For such species, it is not feasible to wait until the livestock is removed. Mature trees of rare species should be fenced off to protect the last surviving seed sources and seedlings. The effectiveness of current reforestation initiatives should be monitored and evaluated.

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27 Invasive Species: Major Threat to Caribbean Netherlands Biodiversity

Van den Burg, M. P., van Proosdij, A. S. J., Boeken, M., van Buurt, G., de Freitas, J. A., Houtepen, E., van Leeuwen, S. Mitchell, A., de Waart, S. and Debrot, A. O. 2025. From: State of Nature Report for the Caribbean Netherlands, 2024. WUR report C001/25.

Status

Not all introductions of exotic species will ultimately result in biological establishment or invasive tendencies but do carry that risk. Assessment of risks is complicated; assessment of invasions is somewhat easier after a non-native species has been present for a shorter or longer period and risks differ greatly depending on the species. However, once invasion takes place and becomes evident typically any action to reverse the problem is too late and the ecology of the area invaded will likely forever be impacted. Therefore, in this assessment of the invasive species problem we use exotic species as the barometer for the invasive alien species (IAS) problem.

A first assessment of invasive alien species (IAS) within the Dutch Caribbean was performed in 2011, which indicated the presence of 211 exotic, non-native species across different invasion stages. These included 27 marine, 65 terrestrial plant, and 72 terrestrial animal species as well as 47 introduced pests and diseases. Lists of these species, pests and diseases are found in respectively Debrot et al. (2011), van der Burg et al. (2012), and van Buurt and Debrot (2012; 2011). Even without an exhaustive review, we here now report an additional 710 new island occurrences of (potentially invasive) exotic taxa which have been documented from nature on one or more of the six Dutch Caribbean islands (Bonaire, Saba, St. Eustatius) since the 2011 inventory. These new island occurrences amount to for example, 40 records of exotic reptiles, 54 records of exotic snails, 10 records of non-native land flatworms, 448 records of exotic weedy plants, and 100 records of exotic insects (Table 1).

The NEPP for the Caribbean Netherlands assigns a high priority to the invasive species problem (Min. LNV et al., 2020), which worldwide is considered second only to habitat destruction as a long-term threat to biodiversity (Kaiser, 1999; Mooney and Hobbs, 2001).

Table 1. *Number of newly identified non-native species among the Dutch Caribbean islands. (see Appendix 3 for full listing).*

Species group	New records of exotic species
Mammals	6
Fish	8
Birds	12
Amphibians	4
Reptiles	40
Mollusca	53
Flatworms	10
Earthworms	1
Insects	100
Animal diseases, vectors and parasites	10
Plant diseases, vectors and parasites	15

Species group	New records of exotic species
Other invertebrates	7
Fungi	0
MLO's (Mycoplasma Like Organisms)	0
Plants	445
Total	**710**

Characteristics/Knowledge

The ever-increasing international traffic of persons and cargo has facilitated non-native species introduction throughout the Caribbean, including the islands of the Caribbean Netherlands. Insects are transported in suitcases, marine species are transported in ballast water, soil fauna hitchhikes with plant imports, and terrestrial plants and animals are escaping from cultivation, captivity, and particularly from cargo with construction materials, consumer goods, and plant imports. Although most translocated species cannot adapt to the new environment or do not survive long enough for reproduction, some can. These often remain unnoticed for years whilst adapting to the new environment; the so-called 'lag phase'. Once circumstances are right, they may proliferate exponentially because they occupy a 'niche' that was often empty or that belonged to a less-competitive native species. Establishing arrivals commonly proliferate due to the absence of natural enemies. During the time that native predators need to adapt, the new arrival can proliferate freely and outcompete local species, endangering them with extirpation of extinction. Examples of such species are the Lionfish (*Pterois volitans/miles*) that negatively affects all reefs of the Caribbean Netherlands and impacts fishery production by preying on fish larvae and outcompeting local fish (Albins and Hixon, 2008), the Common Green Iguana (*Iguana iguana*) that threatens the critically endangered Lesser Antillean Iguana (*I. delicatissima*) on St. Eustatius and the Saba Green Iguana on Saba, and the Madagascan Rubber Vine (*Cryptostegia grandiflora* and *C. madagascarensis*) that have no native natural enemies and are a problem to the vegetation on all three islands.

The arrival of non-native species within native communities is a large and ever-growing problem world-wide, including the Caribbean (Williams and Sinderman, 1992; Williams et al., 2001; Kairo et al., 2003; Lopez and Krauss, 2006). IAS cause major economic losses worldwide (Pimentel et al., 2005) and rank amongst the most important drivers of local and global reductions in biodiversity (World Conservation Monitoring Centre, 1992; Vitousek et al., 1996, 1997; Mooney and Hobbs, 2000; Butchart et al., 2010). Island ecosystems are especially vulnerable to biological invasions and often possess unique assemblies of endemic biodiversity, including the islands of the Dutch Caribbean which all lie within a global hotspot for biodiversity (Mittermeier et al., 1999; Myers et al., 2000). Islands are particularly at risk because of several factors: 1) their small size, resulting in vulnerable plant and animal populations; 2) relatively high numbers of endemic species which have evolved without the ecological pressures found on larger landmasses, often experiencing naturally lower levels of predation and competition, for example; 3) a relatively large border zone in relation to surface area which can be difficult to control; 4) small local economy, resulting in high amounts of imported cargo and goods; 5) a small human population lacking the "economies of scale" necessary to support the institutions, expertise and resources needed to effectively implement and take adequate measures.

The Netherlands is signatory to several international treaties and conventions which accord special emphasis to invasive species. These are the ***Convention on Biological Diversity (CBD)*** which in Article 8h calls on its members 'to prevent the introduction of, control or eradicate those alien species which threaten ecosystems, habitats or species', the 2004 ***IMO Ballast Water Convention*** and the ***Ballast Water Management Convention (BWM)*** which the Netherlands ratified in 2010, and finally the ***International Plant Protection Convention (IPPC),*** which principally aims to protect cultivated and wild plants by preventing the introduction and spread of pests. In 2014 strict new norms were implemented throughout Europe, including the Netherlands, to forbid importation of dangerous or risky exotic species. These international treaties call for an active IAS program, also within the Caribbean part of the Kingdom. However, until now any form of structural joint IAS management remains wanting. Several countries in the Caribbean have developed a strategy to address their invasive species problem, such as Jamaica (Townsend, 2009), the Bahamas (BEST Commission, 2003) and St. Lucia (Andrew and John, 2010; Chase et al., 2011). These may serve as examples for the Caribbean Netherlands on how to implement their own strategy to address this urgent issue.

Brown and Daigneault (2014) review economic impacts for case studies of invasive species in the Caribbean. Special cases are introductions that may affect human and animal health, such as disease-transmitting mosquito species. The costs of control grow exponentially with the growth of the invasive populations. For example, whilst over 5 million US dollars have been spent within a harvest management program to reduce invasive green iguanas on Grand Cayman, continuous removal and financial aid will be necessary to prevent population regrowth (Rivera-Milan and Haakonsson, 2020). Therefore, it is of utmost importance to try to prevent the introductions altogether or halt them at an early stage. This means strict biosecurity control at both import and export. Developing a system of monitoring, early detection, and control and management, requires knowledge about the species present in the region as well as legal authority and institutional capacity to take decisive action on land or at sea.

Ecological Aspects

Negative effects:
Invasive species cause major ecological effects (decimating native flora or fauna populations) as well as economic losses, across sectors such as agriculture (diseases, weeds, vectors, and animal pests), fisheries (fish diseases, the Lionfish, smothering coral and sponge species), industry (rodents and termites), tourism (roadside weedy species) and public health (mosquitos and introduced parasites). Ecological effects are numerous and often multiple per IAS; e.g., direct predation, (out)competition for food or complete niche space, hybridization, overgrowing, and spill-over of parasites, bacteria and diseases. Unfortunately, negative effects are often not immediately noticeable and often a species will persist at low and seemingly unharmful densities for years before becoming a major problem. Therefore, altogether prevention (or as next-best early detection and eradication) is preferable to letting a species of high risk come in persist and spread before doing an assessment of what to do.

Maximum allowed population density:
In principle, even the smallest presence of a non-native species is to be avoided as it alters the native community composition and somehow affects the ecological processes even though it may not yet be clear how or to what extent. For instance, at the lowest level of impact, the introduction of non-native plant species can take place gradually over time and ultimately change native forests into “novel” forests dominated by non-native species (Lugo et al., 2020). It is especially tropical island forests that are vulnerable to developing into novel forests (“fauna- en floravervalsing” in Dutch). While such novel forests (or novel animal communities e.g., Raymond-Léonard et al., 2018) may be able to fulfil many of the ecosystem processes and functions of the original communities and not always represent a “total environmental loss”, preventing establishment of exotic species at the preference of native species should always be the priority but is not always possible. This is particularly the case when incursions have not been halted at the national border or when eradication is unsuccessful at an early stage. In such cases, a form of tolerance may be the only option substantiated with or without using a local ongoing control program. Ongoing control is often a costly management measure, so strong prioritization is essential. A choice needs to be made between which species to control, where to do so, to what density, and to know whether the efforts actually have the desired effect.

Important IAS and Recent Developments

Recent developments within the Caribbean Netherlands:
Since the last IAS survey and the 2017 State of Nature (Debrot et al., 2018), even more potential damaging introductions have occurred and new data on historical introductions have become available. Our newest inventory adds an additional 714 new island occurrences of potentially invasive exotic taxa which have been documented on one or more of all six Dutch Caribbean islands, (not only

the three Caribbean Netherlands islands; Bonaire, Saba, St. Eustatius) since the 2011 inventory (Debrot et al., 2011; van Buurt and Debrot, 2011, 2012; van der Burg et al., 2012). It lies beyond the scope of this report to discuss in detail all newly reported exotic species, the risks they may represent and what, if anything, can be done about them.

However, especially alarming is the high number of new established reptile populations among the BES islands (including Green Iguana, geckos, and tegulets; Figure 1), guinea pigs and rabbits that have become established on Saba, as well as the presence of the New Guinean Flatworm on both Bonaire and Saba. Similarly alarming is the large number of non-native plant species that have escaped and naturalized from gardens and agricultural areas as well as were brought in unintendedly by residents and tourists (Figure 2). More than half of the flood of exotic species entering the wild in the Dutch Caribbean are plants, for which there is no phytosanitary legislation or control.

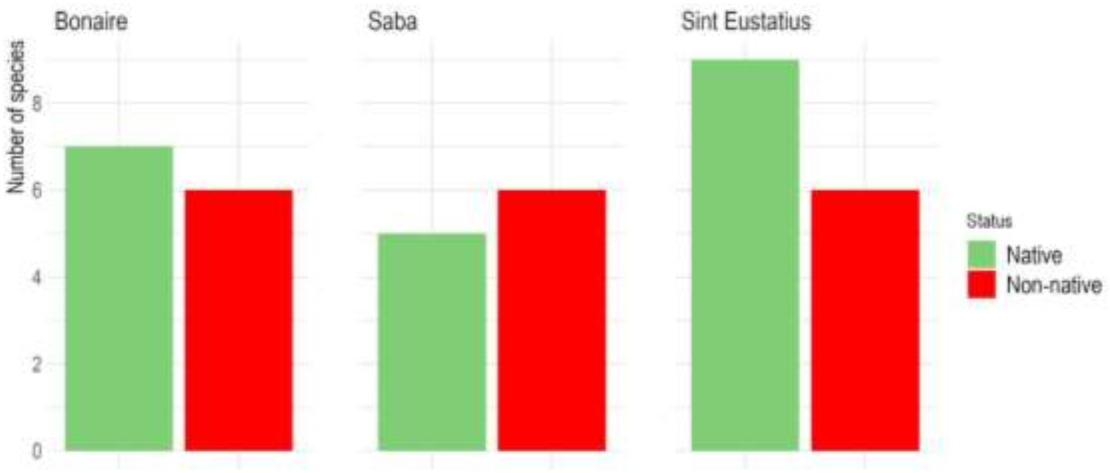

Figure 1. *Overview of the number of native and non-native species of herpetofauna on the BES islands recorded by 2024.*

Recently, further complexity to the ongoing incursions of Green Iguana from St. Maarten to both Saba and St. Eustatius have been identified. Genetic analyses have demonstrated that non-native ectoparasitic mites have been introduced to both populations, where these can spill-over to native iguanas (van den Burg et al., in prep). This was equally found for the tick *Ornithodoros puertoricensis*, which has been identified on Saba and St. Eustatius in 2023 and 2024 (van den Burg and Debrot, 2025). Alarmingly, *O. puertoricensis* is a known carrier of tick-borne diseases and has been found elsewhere on a variety of other hosts like rats, goats, cats, and humans (Endris et al., 1991; Paternina et al., 2009). Furthermore, a preliminary microbiome analysis of native and non-native iguanas on Saba suggests that non-native iguanas have introduced several invasive and potentially harmful bacteria, which are known to be able to transfer to other native reptiles (Hellebuyck et al., 2017). Additionally, there are numerous species present on other Caribbean islands, as well as the main regional hub of IAS, Florida (e.g. Witt, 2024), which have not (yet) reached the BES islands. Recent sightings of at least two reptile species (*Agama picticauda* and *Phelsuma laticauda*) from several Caribbean islands have led to major concern among regional stakeholders (van den Burg et al., 2024a; De Jesús VillaNueva et al., in prep.). Also worrisome are several species of *Anolis* which have already become established on St. Maarten (Dewynter et al., 2022).

While structural measures against the growing flood of introductions of known or potential invasive species are the greatest single priority (Smith et al., 2014), the need to be able to take effective measures against invasive species once present is also crucial. Therefore, in recent years Wageningen University and Research and its island partners have conducted several studies or field pilot interventions directed towards either:

a) species that have long been present and known to be highly deleterious (like goats, cats, and Lionfish) or

b) "newly"-arrived species of peak risk to native biodiversity and human health (e.g., non-native Green Iguana, the Giant African Land Snail and the New Guinea Flatworm).

c) exotic plants- discussed separately below.

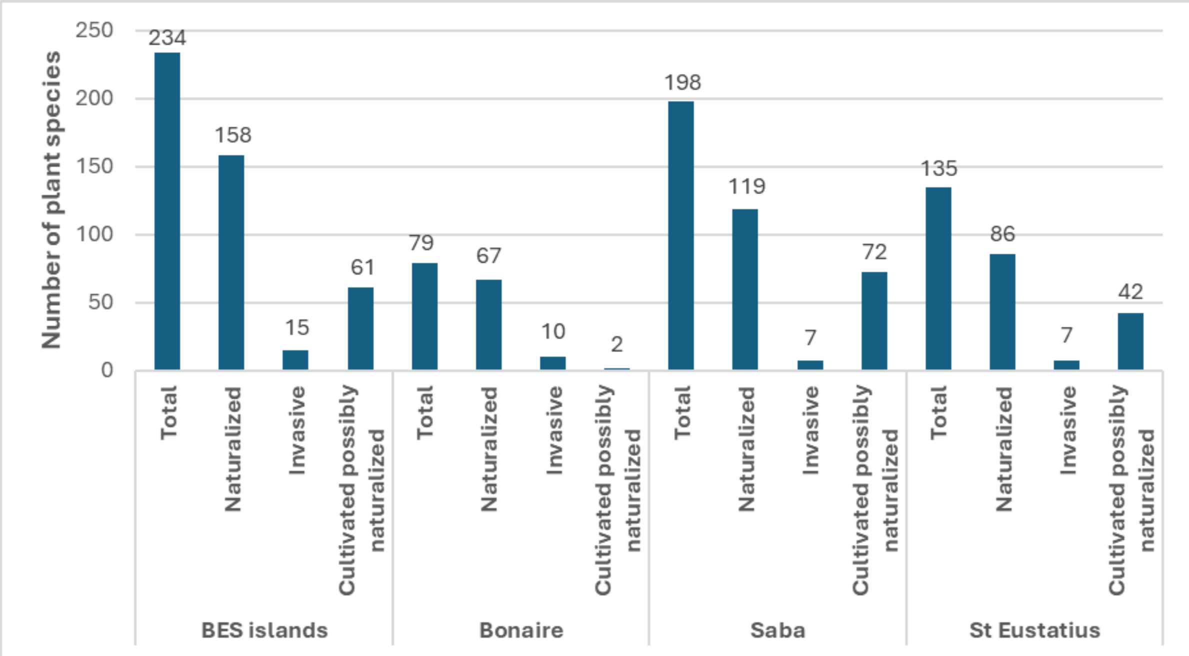

Figure 2. *Overview of the number of non-native plant species recorded in the wild on the BES islands recorded by 2024.*

A) Long present

Goats: Uncontrolled feral grazing by goats is likely the most serious and longstanding invasive species problem with wide-sweeping ecological consequences for both terrestrial and marine ecosystems and is therefore discussed elsewhere in a separate chapter in this collection.

Cats: Feral cats are believed to be directly responsible for some 26% of global species extinctions due to invasive mammalian predators since AD 1500, and today they are the primary existential threat to no less than 367 species worldwide (Medina et al., 2011; Doherty et al., 2016). They have long been present on the islands of the Dutch Caribbean and their impact is likely to be massive but also almost impossible to retrace. They can be very abundant on especially Bonaire where they might well be a factor contributing to the local absence of small endemic terrestrial mammals (the endangered Curaçao Vesper mouse and the Curaçao cottontail) and/or ground-nesting birds like the Crested bobwhite, *Colinus cristatus*, such as are (still) found on neighbouring Curaçao and Aruba.

Feral cats are a demonstrated serious threat to nesting Red-billed Tropicbirds (*Phaethon aethereus*) on Saba, where they became problematic after local animal advocates started dumping unwanted cats into the wild instead of humanely euthanizing them (Debrot et al., 2014). For each cat "saved" by abandoning it in the outdoors, excess predation pressure is put on multiple other species who frequently have no evolved defences to mammalian predators (Figure 2). Therefore, Debrot et al. (2022b) have urged animal advocates to take a more species-inclusive perspective on animal welfare that includes the consideration of collateral animal suffering. Others have argued that culling cats may be a bad idea as they would be the only possibility to contain rats that also prey on small nesting seabirds, their chicks and eggs. For the Red-billed Tropicbird however, which are a fairly large and aggressive bird largely able to fend off rats, nest success only became a problem when cats started being abandoned. In the case of Saba, data has further shown that cats (which are terrestrial predators) are most abundant in the lower drier and more barren parts of Saba, while rats (which are

mainly arboreal) prefer the lusher vegetation higher up the slopes of the island (Debrot et al., 2014). This means that due to such "habitat partitioning" in the Saba situation, cats can never serve to control rats.
In addition, studies elsewhere have shown that even in areas where feral cats and rats co-exist, cats tend to be very inefficient predators of rats, preferring other species which are easier and less dangerous to catch, such as birds and lizards. The perception that rat populations decreased after cat introductions was due to rats' avoidance behaviour towards cats also becoming apparent to human observers (Parsons et al., 2018).

Other recent work on the Dutch Caribbean island of Klein Curaçao, an island formerly populated with both cats and rats, shows the positive effect that cat removal can have even when rats are not removed. Within a decade (from 2009 to 2021), cat removal resulted in the number of seabirds breeding on the island increasing from a single breeding species with maximally 140 pairs to nine breeding species with upwards of 430 pairs annually (Debrot et al., 2023a). Removal or culling programs for mammals like cats (but also goats and donkeys) often evoke strong emotional sentiments with the public, whereby public opinion then may have major impacts on the continuity and effectiveness of such programs. Therefore, such programs require careful preparation to properly educate and inform the public on why it is so important to remove such invasive mammals. Work by P. Bertuol on Bonaire has further shown how a single stray cat can wipe out a colony of more than 30 breeding pairs of terns in a single night (video on file). So, while removing cats from seabird breeding areas and conservation areas on Saba and other islands is challenging, it is feasible and will give measurable positive results in terms of higher seabird nesting success quite directly (Terpstra et al., 2015; Debrot et al., 2022b). In this whole, rats should also not be forgotten as they can also greatly depress seabird breeding success (as on Klein Curaçao; Debrot et al., 2023a) but rats are a totally different challenge in terms of control or eradication due to the increased difficulty in locating them and their much higher reproductive potential.

Figure 2. *A feral house cat just moments before killing and removing an almost-fledged Saba Red-billed Tropicbird chick from its nest.*

<u>**Lionfish:**</u> *Pterois volitans/miles*, or Lionfish for short, have been present in the tropical Western Atlantic for more than 30 years and have widely spread throughout the region. While a massive amount of research has been devoted to the Lionfish question and major grassroots efforts have been made in terms of trying to control outbreaks locally, the ultimate conservation effectiveness of all this effort can now be highly questioned for two reasons: research has shown that the largest Lionfish populations are located at depths well below the maximum safe diving depth at which they can be

removed by spearing, and studies from various areas such as the Gulf of Mexico, the Bahamas and the Saba Bank (e.g., Debrot et al., 2023b) have shown large population crashes. This suggests that the species likely has peaked and is now stabilizing in the region at lower and more sustainable densities due to emergent biological controls. So while research initially included some exploratory fishing using different trap designs to potentially develop Lionfish as an alternative for the Saba trap fisheries (de Graaf et al., 2017), after the population crash on the Saba Bank (Debrot et al., 2023b) and plummeting trap catches, these efforts were abandoned as being of unlikely practical conservation value.

***Halophila* seagrass:** Another marine species that is similarly beyond "the point of no return" is the invasive seagrass *Halophila stipulacea*, well-established in the Caribbean region since around 2002. The species has widely spread, also invading original seagrass habitat massively throughout the Dutch Caribbean (Willette et al., 2014). Even though it greatly reduces the fish nursery habitat of seagrass beds (Becking et al., 2014) and is of lower nutritional value to sea turtles than the native Turtle Grass (*Thalassia testudinum*) which it is replacing (Christianen et al., 2018), there is likely little that can fruitfully be undertaken against this species at a scale at which it really makes a difference. Setting priorities in research and intervention efforts have never been more acute than now.

B. "Newly arrived"

Green Iguana: The Green Iguana, *Iguana iguana*, is a species that has become popular as a pet and has been and is being traded worldwide (CITES, 2024). Therefore, feral populations have been establishing themselves in tropical regions around the world where they create serious conservation problems (van den Burg et al., 2020; Knapp et al., 2021). This is particularly the case in the Caribbean where they not only compete with but also interbreed with the native iguanas (e.g. Vuillaume et al., 2015). Interbreeding is particularly problematic as the resulting gene swamping means that the native population genome is gradually overtaken by the invading *Iguana iguana* genome. As a result, all iguana populations of the Lesser Antilles are under threat and *Iguana delicatissima* has lost more than 91% of its former range due to hybridization with invasive Green Iguanas (van den Burg et al., 2023). In response to the relatively recent discoveries of invasive iguanas on St. Eustatius and Saba, the Netherlands Ministry of LNV financed research on rapid response removal campaigns which remain ongoing on both islands. The results demonstrate that early removal of invasive iguanas is feasible (Debrot et al., 2022a) and further methods are being developed to also be able to swiftly field-identify hybrid and introgressed iguanas for culling (van den Burg et al., 2023, 2024b), so that iguana invasions can be halted while still possible at an early stage.

Giant African Land Snail: The Giant African Land Snail, *Lissachatina fulica,* has been introduced to the Dutch Caribbean islands in recent years. The most recent island on which its presence has been found is Bonaire (van Leeuwen et al., 2023, in prep.), while on St. Eustatius it was first detected in 2013 (Debrot et al., 2016). Its impact on native vegetation and agriculture can be serious, which makes it one of the most significant agricultural pest species in tropical areas (Rauth and Barker, 2002). The snails are also a potential risk to human health because they can be the host of the nematodes *Angiostrongylus cantonensis* and *A. costaricensis*, which can both cause serious diseases in humans (meningoencephalitis and/or eosinophilic meningitis respectively abdominal angiostrongylosis). And third, the snail can carry the bacterium *Aeromonas hydrophila* (Chester) Stanier, 1943, that has caused a variety of bacterial infections (bacterioses) in humans (Smith, 2005; CABI, 2018). Its initial introduction to the region was probably based on its potential value as a food or pet species but its further spread is likely largely due to it hitching rides in shipments of ornamental plants. The species is abundant on Sint Maarten, which is the main port for goods transported to Saba and Sint Eustatius. On Sint Eustatius, trials to contain and eradicate the species were conducted fairly early in the process when it was almost only found on a few streets in a single neighbourhood (Debrot et al., 2016). While the results of the trials were very promising, the local agricultural authorities have since found it impossible to sustain the eradication effort. Consequently, more than 10 years after its introduction, it has spread across the island and eradication may no longer be possible. On Bonaire in 2023, its distribution was still limited to 2-3 small areas (Van Leeuwen, 2024; Van Leeuwen et al., in prep), suggesting that eradication might still be possible before it spreads more widely. An assessment of the current situation on Bonaire, followed by a more thorough and systematic approach to control or eradicate the snail from Bonaire is highly recommended.

New Guinean Flatworm: The New Guinean Flatworm (*Platydemus manokwari*) was only discovered on Bonaire in 2023 (De Waart and Van Leeuwen, 2024; De Waart et al., in prep). The species is notorious as a predator of land snails and is responsible for wiping out whole endemic land snail communities on islands in the tropical Pacific and in Florida (Suguiura, 2010; Lopez et al., 2022). Its presence on Bonaire threatens the rich native land snail fauna of the island, which includes nine regional endemic species, of which eight are restricted to Bonaire. Thanks to emergency funding provided by the Netherlands Ministry of LNV, a rapid response assessment was conducted in 2024 to determine the current distribution and what if any measures can be taken to halt further spread or accomplish eradication. The species already appears to be present on Saba at moist higher elevations where its potential impact can even be expected to be worse than on Bonaire which is a much drier island. However, since the first record on Saba, no study has yet been performed to assess its status and distribution there. Equal to the Giant African Land Snail, the New Guinean Flatworm can be a host of *A. cantonensis*, which can cause diseases that can lead to blindness and death in rare cases (Smith, 2005; Thunnissen et al., 2020). Whilst the species is not known to occur on St. Eustatius, no land-flatworm study has been performed there.

C. Exotic terrestrial plants

Exotic plants are entering and establishing themselves in nature at an ever increasing rate and amount to more than half of the 714 new island records for invasive species recorded since the 2011 and 2012 inventories. For centuries, many plant species have been cultivated on the Dutch Caribbean islands. Most of these do not survive outside the garden environment where competition from native plant and animal species is largely absent. Several however have spread into surrounding areas and once established managed to invade natural areas. Some have become a true pest, outcompeting native species by covering entire areas, smothering all other plants species present. The most illustrative example is Coralita or Mexican bellcreeper (*Antigonon leptopus*) that is currently covering some 15% of St. Eustatius' land surface. The list of exotic plant species that have been recorded (far) outside gardens is long (see Figure 2 for numbers) and can increase even further considering the even larger number of species that is currently cultivated in gardens. Additionally, the number of pantropical weeds is huge and globalisation facilitates transport of plant material on a massive scale. Import of plant propagules occurs via several pathways. Most important is the largely uncontrolled import of ornamental plants as seeds, cuttings, bare-rooted plants or in containers for horticultural purposes. In addition, via bulk import of fodder for cattle and soil for building activities pantropical weeds are imported. Finally, the unintended transport of seeds, spores and vegetative parts by humans, both residents and tourists enables exotic weeds to arrive on the islands. In appendix 3 a all non-native plant species are listed that have been documented to occur outside gardens. For Bonaire, the 2012 edition of the Flora (van Proosdij, 2012), as well as additional plant records present in the CACTUS database (Janssen et al., 2023) have been used. For Saba and St Eustatius, the recently published checklists (Axelrod 2017 & 2021), as well as additional plant records present in the CACTUS database (Janssen et al., 2023) have been used. In total, 234 exotic vascular plant species are recorded for the BES islands. Occurrence on Aruba, Curaçao and St Maarten is provided too for reasons of comparison, although data on St Maarten are particularly sparse. Often, a species is cultivated on one island but has become naturalized on another island. For several species listed in Axelrod (2017, 2021) as occurring outside gardens the actual distribution remains unclear as often the number of observations is very small. Viewing the long list of non-native species and the ongoing establishment of additional exotic species, strict phytosanitary regulations are urgently needed in addition to a much more in-depth inventory of the IAS currently present.

Other species: While an expansive review of all exotic species is beyond the scope of this report, on all three islands the number of potentially invasive species continues to grow. On Bonaire, feral cats continue to exact a high toll in the tern nesting colonies but remain unaddressed as does the continued spread of the invasive Neem Tree, *Azadirachta indica*. While the Neem Tree invasion is far advanced and likely little meaningfully can be done about its further spread, combatting cats at tern nesting colonies should yield large and easily measurable results in terms of enhanced nesting success for several tern and other shorebirds (see results of Terpstra et al., 2015 and Debrot et al., 2023a). Bonaire has long been of high international importance as a nesting island for several tern species and protection of this international importance deserves a high priority (Debrot et al., 2009). For St. Eustatius, two of the most ecologically impactful recent introductions are the Giant African Snail and continuous Green Iguana incursions (van den Burg et al., 2018; Debrot et al., 2022a). The most economically damaging recent introduction so far is the Lethal-yellowing virus that has killed a large fraction (maybe 30%) of the coconut trees. Especially troubling is the recent prediction of future coverage by Coralita across the island (Huisman et al., 2021) as well as the continued rapid spread of the invasive Neem Tree into the forested protected areas of the Northern Hills (A. Debrot, pers. Obs.). The recent discovery of the Agave Weevil (*Scyphophorus* spp.) adds additional pressure to the island's dry-adapted flora, already decimated by the Cactus Moth *Cactoblastis cactorum*. For Saba, the growing number of invasive reptiles is especially noticeable (van den Burg et al., 2021; van den Burg and Debrot, 2024). The most disturbing recent development is the establishment and spread of introduced Guinea pigs and rabbits at The Level, and Green Iguanas both at the harbour area and in Windward Side (van den Burg et al., 2023). These species can likely still be eradicated if rapid action programs are implemented. Feral cats also pose a serious threat to nesting seabirds, especially in the lower areas of the island. At higher elevations, rats are very abundant in the rainforest, where they presumably have a significant impact on the reptile and avifauna species, and likely on multiple plant species, several of which are already locally threatened. There is no recent report on the spread of Coralita on Saba, which is well established and likely continues to expand into gullies where disturbance by torrents during the rainy season is high. However, on Saba the spread of Coralita has been and remains much less than on St. Eustatius mainly due to the much lower extent of anthropogenic vegetation disturbance.

Assessment of National Status

Recent developments:

Assessment of distribution: Unfavourable-bad
Overall, distributions of establishing IAS continue to increase in the absence of structural financial aid for eradication/management programs. However, very little knowledge about the distribution of the identified exotic species is present. The distribution per IAS per island is often highly dependent on its establishment state, mode of introduction and time since first incursion. For example, some species appear initially only present around the port of entry, like *Hemidactylus frenatus* on Saba (van den Burg and Debrot, 2024), whilst non-native Green Iguanas on Saba have additionally been reported from Windward Side (van den Burg et al., 2023), and another non-native reptile for Saba, *Gymnophthalmus underwoodi,* was able to spread across the entire island in only 5 years' time (van den Burg et al., 2021). Whilst the Giant African Land Sail is limited to few locations on Bonaire, on St. Eustatius it has spread much wider from its initial limited range. IAS that have long been established generally occurs across the islands, such as cats and rats. The Caribbean Netherlands are surrounded by island nations that likewise have very poor to non-existent IAS intervention plans. Given the extent of inter-island travel and trade within the Caribbean Netherlands, between the Caribbean Netherlands and the rest of the Caribbean, which include the three other islands of the Dutch Caribbean, the risks of further introductions, be it intentional or unintentional, are very high.

Assessment of population: Unfavourable-inadequate (variable depending on the species)
Apart from some conducted inventories and surveys on the occurrence of feral cats on Saba (Debrot et al., 2014), the Lionfish on Bonaire (White, 2011; De Leon et al., 2013), and the African Giant Land Snail on St. Eustatius (Debrot et al., 2016), there is no data available on IAS densities from the BES islands.

Assessment of impact Unfavourable-bad
The magnitude of impact varies per IAS, with some species seemingly having no immediate and clearly observable effect; although adequate impact studies are rare. However, numerous IAS present on the BES islands have an immediate and sometimes disastrous impact on native species and ecosystems. These include feral cats, Coralita, and recently arrived non-native green iguanas and the New Guinea Flatworm.

Assessment of management intervention: Unfavourable-bad

There is no structural management strategy and associated financial backing for addressing the IAS issue in the Caribbean Netherlands islands. Some temporary projects have seen reactive financial aid from the ministry, often when new problematic arrivals have recently appeared, but frequently these 'new arrivals' will have been present unnoticed for some time, making their control more difficult. Harmful species list of high-risk species to watch out for, similar as are known for the mainland EU, are urgently needed. Biosecurity intervention should be prioritised by all relevant ministries and local governments.

Assessment of future prospects: Unfavourable-bad

If no measures are implemented, the invasion process will continue to accelerate, with all its consequences. This is especially problematic given several planned major infrastructural projects that will require a large quantity of imported materials, e.g., the planned harbour on Saba at Black Rocks. If no rigorous measures are implemented, the many unique and endangered species of the islands will increasingly be at risk of extinction, ecological functioning will be profoundly compromised, and native flora and fauna will gradually be replaced by entirely artificial nature, "novel tropical forests" (sensu Lugo, 2009; Lugo et al., 2020).

Table 2. *Summary overview of the threat status of invasive alien species to biodiversity of the Caribbean Netherlands in terms of different criteria.*

Aspect invasive species	**2024**
Distribution	Unfavourable-bad
Population	Unfavourable-inadequate
Impacts	Unfavourable-bad
Future prospects	Unfavourable-bad
Overall Assessment of Threat Status	**Unfavourable-bad**

Comparison to the 2018 State of Nature Report

Overall, the situation with respect to invasive species and the risks they present has significantly worsened since the 2018 State of Nature report. No less than 714 new island occurrences of non-native species can be reported since the 2011 and 2012 inventories (Debrot et al. 2011; van Buurt and Debrot 2011; 2012; van der Burg et al., 2012; Smith et al., 2014). This is an average of no less than 54 species per year. However, the current rate of increase is certainly higher because the process is exponential which means that the "average" always underestimates the present status.

Recommendations for National Conservation Objectives

National protection targets:

Implementation of a proactive strategy towards IAS (Townsend, 2009) should be based on:

a) Prevention – to limit the number of incursions and IAS that enter each island's borders: develop infrastructure and measures to minimize incursion risk of non-native species

b) Early detection and eradication – to detect, track down and eliminate potential threats before their establishment and subsequent spread: develop monitoring programs and awareness with harbour personnel.

c) Inventory of IAS present in cultivated areas and in natural areas. Particularly for plant species, the list of known IAS present on the BES islands is far from complete, hampering effective detection and eradication as well as raising public awareness.

d) Control and management of species already established - to minimize impact: create structural financial program.

e) Rehabilitation - of areas rendered useless by invasive species: promote ecological restoration and reforestation.

f) Public awareness - proper public awareness towards travel with live biological materials, the risks with importation of materials from abroad and towards early detection and eradication are essential and have already proven of utmost value in St. Eustatius efforts for control (e.g., Debrot et al., 2016; 2022a).

Protection-sub targets:
- Prevent new introductions by
    - Creating Alert and Watch lists for invasive species, as well as White lists
    - Make IAS an integral part of infrastructural planning (especially for new harbours)
- Eradicate or manage the most damaging established species

Most important threats and management implications:
The introduction of new IAS that form a threat to nature, healthcare and the economy is proceeding at an accelerated pace.
- There are several urgent problem species (e.g., exotic predators and iguanas) for which pilot projects demonstrate that eradication or control are practical and financially feasible. In addition to prevention of new introductions, these species should be preferred for targeted intervention.
- The economic costs of IAS evidently become enormous but have not yet been made transparent. Making these costs visible will provide a significant argument for a more proactive stance by governments and nature managers and is therefore also recommended by us as a top priority.
- Because the IAS problem is so extensive and involves so many partners and stakeholders, the development of broader policy frameworks is necessary within which individual legislation needs to be elaborated for the different jurisdictions. This could involve the establishment of so-called Invasive Species Management Teams (ISMTs). A common vision, a so-called Invasive Species Strategy (and Action Plan) (ISSAP), has already been largely developed for the islands.
- From the meetings held on the Dutch islands the consensus is that the IAS problem should be addressed via a three-tiered approach (a) prevention, b) rapid response and c) control and mitigation). Parties agree that prevention of entry should be the focus with which to limit and contain the IAS problem.

The two biggest bottlenecks to implementation are the almost total lack of useful legislation, and lack of capacity. The exception is where it concerns a few species of public health concern, such as the yellow-fever mosquito and rats which do some capacity and some funding, often as part of regional WHO (World Health Organization) programs.

Data Quality and Completeness

Apart from a few studies (see above), there is no data available on IAS densities from the Caribbean Netherlands. This is the same for many other aspects such as distribution, but often also the impact on native species. Monitoring is an expensive endeavour, and priorities must be sharply set. For many species that have already established themselves, and for which the meaningfulness of conducting action is questionable, monitoring is discouraged. Monitoring of IAS should certainly focus largely on the borders of the islands (importation harbours) to prevent introduction of new agents. As well as during the period of first discovery after any new introduction. However, for successful eradication of invasive plants or animals, monitoring may be necessary for several years during and after eradication efforts to be sure that no invasive individuals have been overlooked. For future major infrastructural projects (new Saba harbour), especially those that require large quantities of imported construction materials, we recommend the budgetary inclusion of strict biosecurity measures and species monitoring during the project.

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28 The Climate Change Threat to Biodiversity in the Caribbean Netherlands

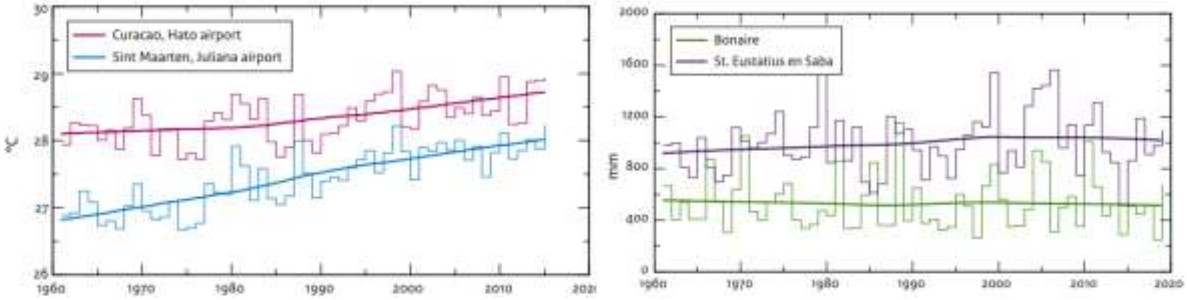
Verweij, P. J. F. M., van Klaveren, S. and Molenaar, R. E. 2025. From: State of Nature Report for the Caribbean Netherlands, 2024. WUR report C001/25.

Introduction

Climate refers to the average weather conditions (temperature, humidity, air pressure, wind, cloud cover, and precipitation) over a specific period. Climates are not stable and change under both natural and anthropogenic influences (KNMI, 2016; KNMI, 2023). The warming of the climate system is unequivocal, and the changes observed since 1950 have been unprecedented for decades to millennia. The last ten years make up the top ten hottest years on record of the Earth’s surface. By 2024, the global average temperature has increased by more than the Paris policy target of one-and-a-half degree Celsius for the first time above pre-industrial level (Copernicus, 2025). Most of the heat is absorbed by the oceans, resulting in thermal expansion which is one of the factors leading to rising sea levels (Widlansky et al., 2020). The current sea-level rise in the Caribbean is 3.40 ± 0.3 mm/year (1993–2019), which is similar to the 3.25 ± 0.4 mm/year global mean sea-level (1993–2018) (Maitland et al., 2024). The world’s oceans will continue to warm, with the heat reaching the deep sea and affecting ocean circulation (Van Westen et al., 2024). The atmospheric concentration of carbon dioxide, methane, and nitrous oxides is higher than at any point in the past 800,000 years. Due to the absorption of 30% of human-emitted carbon dioxide (from fossil fuel emissions and land use changes), the ocean has become more acidic and will continue to acidify. On a global scale, the contrast in precipitation between wet and dry regions will further increase (IPCC, 2022).

Current Climate

The Caribbean climate can be characterized as a tropical climate with dry and wet seasons. Storms and hurricanes are the primary sources of rainfall, with significant local differences due to elevation and topography. The windward islands (St. Eustatius and Saba) have a tropical monsoon climate with a wet late summer and autumn. The average daily temperature is around 30 degrees Celsius. The leeward island (Bonaire) has a tropical arid climate. Annual and local climate variations can be significant.



Annual temperature average

Annual precipitation sum

Figure 1. *Observed trends of temperature and precipitation (KNMI, 2021).*

Long-term trends are observed via weather stations in the vicinity of the Dutch islands, via Curaçao as representation of Bonaire (leeward islands), and via St. Maarten as representation for Saba and St. Eustatius (Windward islands) (*Figure 1*). Every 10 years, the average temperature has increased by 0.15 degrees Celsius for the Leeward islands and 0.23 degrees Celsius for the Windward islands. Since the 1960-ies annual precipitation patterns have varied, but without any derivable statistical trend (KNMI, 2021). Annual variations in weather in the Caribbean vary and are strongly influenced by recurring events like El Niño and La Niña. KNMI (2023) illustrates this annual deviation of the long-term average in *Figure 2*. Based on the long-term trend, projections have been made.

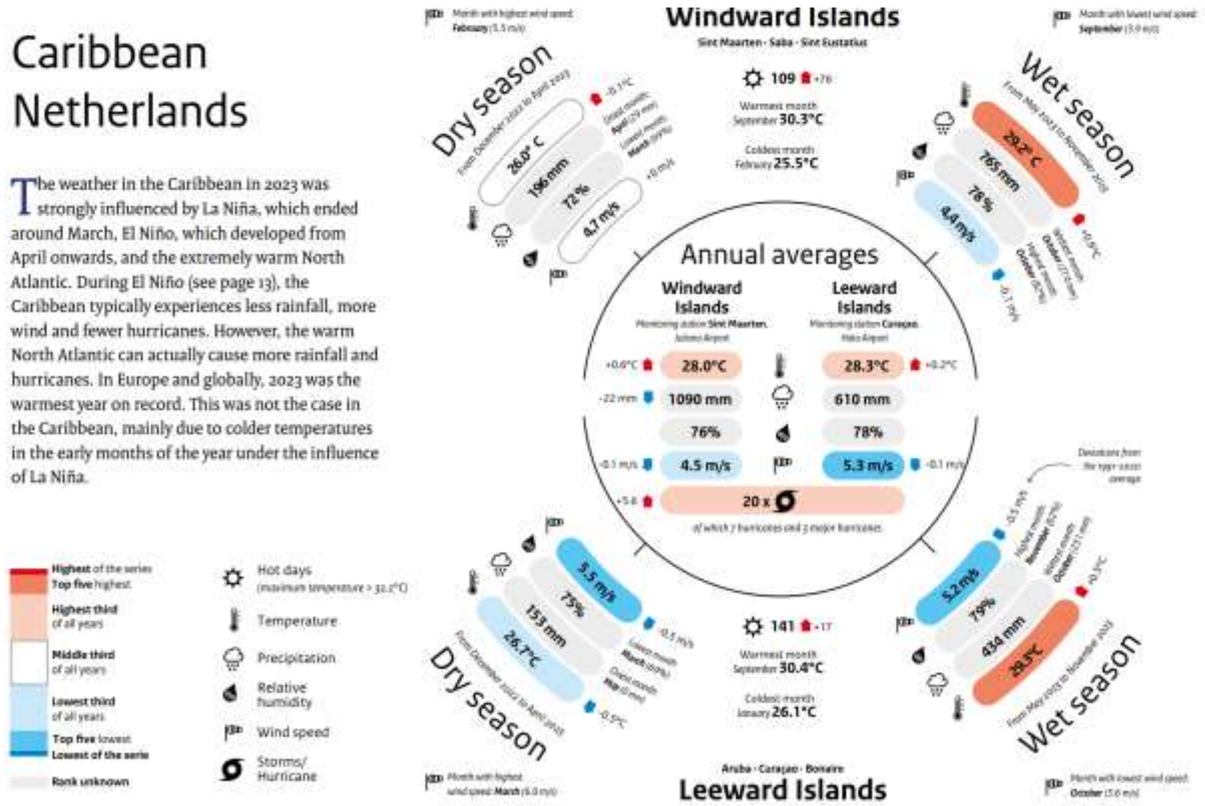


Figure 2. *Deviation of weather parameters of the year 2023 from the long-term average (1991-2020). In 2023, the dry season (December-April) was wetter and colder than the long-term average, while the wet season was dryer and warmer for all Dutch islands in the Caribbean (KNMI, 2023).*

Anthropogenic Influences on Climate Change

Greenhouse gas emissions from small islands are negligible compared to total global emissions, but the threats from climate change, sea-level rise, and global warming are significant for those same small islands. In recent decades, human use of land and sea has intensified considerably. Many small islands in the Caribbean have experienced coastal erosion, negatively affecting buildings, amenities, infrastructure, agriculture and the (natural) vegetation.

Projected Climate Change

KNMI (2023) projects a rising temperature (1 to 3.5 degrees Celsius) for 2100, increasing average windspeed and decreasing precipitation (0 to 48%), especially in the dry season (December – April) (Figure 3). The median of the projections (2081-2100) for temperature increase is 1.4 degrees Celsius, a 5% decrease in precipitation, and a sea-level rise of 0.5 to 0.6 meters for the RCP 4.5 scenario (a comparatively low emission scenario) (KNMI, 2024). Figure 3 shows the historical trends of temperature and precipitation projected into the future. For all islands, the climate is projected to become warmer and dryer (KNMI, 2023).

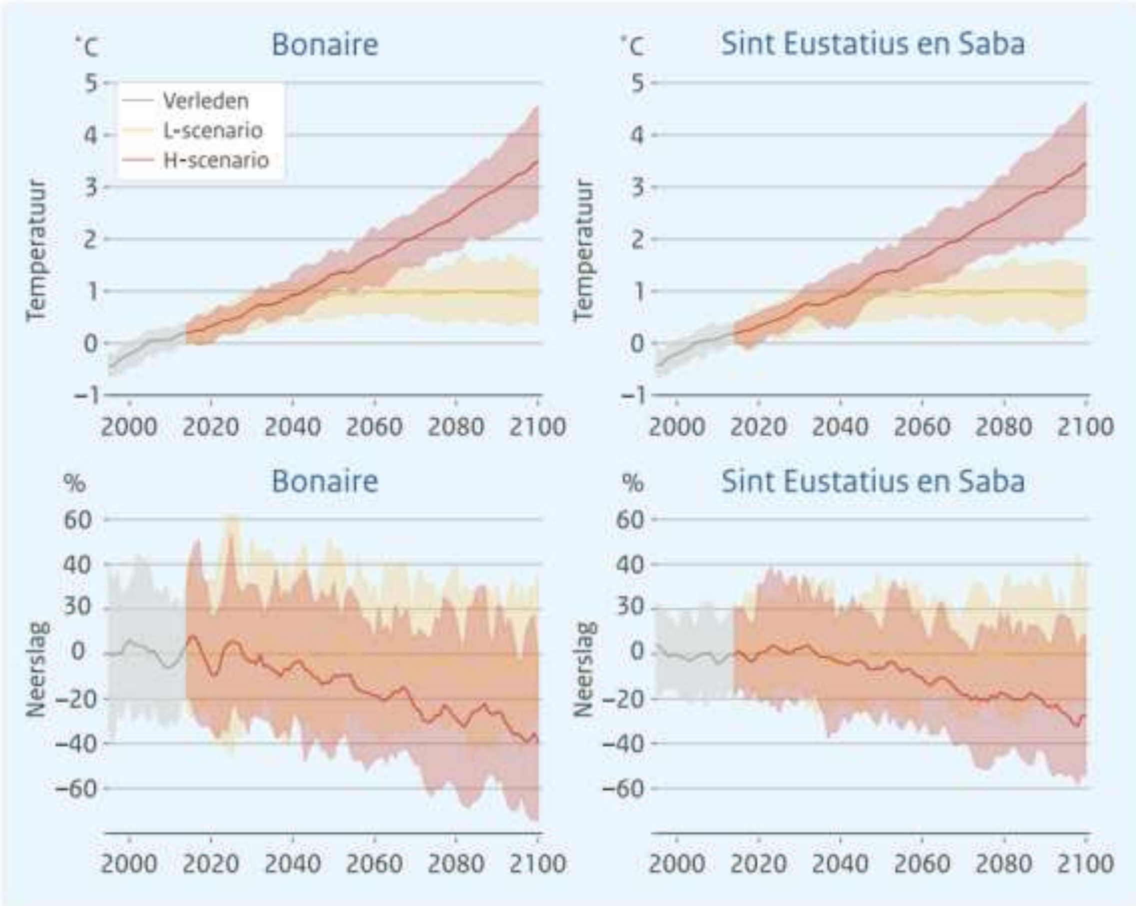


Figure 3. *Projected precipitation and temperature for Bonaire, St. Eustatius and Saba (KNMI, 2023).*

The Dutch Leeward islands are located outside the hurricane belt, resulting in significantly fewer hurricanes than on the windward islands (*Figure 4*).

Climate Risks

Because small islands have close connections between the settlements and coastal environments, they are particularly exposed to climate hazards associated with the ocean and water cycle, including sea-level rise (and surges), tropical cyclones, marine heatwaves, and ocean acidification (Thomas et al., 2020; Nurse, 2014). Human influence closely impacts climate vulnerability: for example, poor land management has greatly influenced erosion, increasing the vulnerability of natural areas, agro-ecological systems and waterways to climate hazards, such as heavy rainfall. Due to the mix of changes, it can sometimes be difficult to attribute specific effects to a specific cause (IPCC, 2022). The effects of climate change will be most significant where the natural environment is already under pressure from human activities (IPCC, 2022; Bijlsma et al., 1996). Climate change poses a serious threat to the sustainable development of the countries in the Caribbean community (CARICOM) and may even jeopardize the long-term existence of those countries (CCCCC, 2009). *Figure 5* illustrates the observed effects of climate change on small tropical islands and biodiversity hotspots. Climate risks for the Dutch Caribbean include loss of livelihoods, damage to coastal settlements and infrastructure, loss of ecosystem services and ultimately risk of loss of economic stability. Specifically for nature climate change poses a risk to coral reefs.

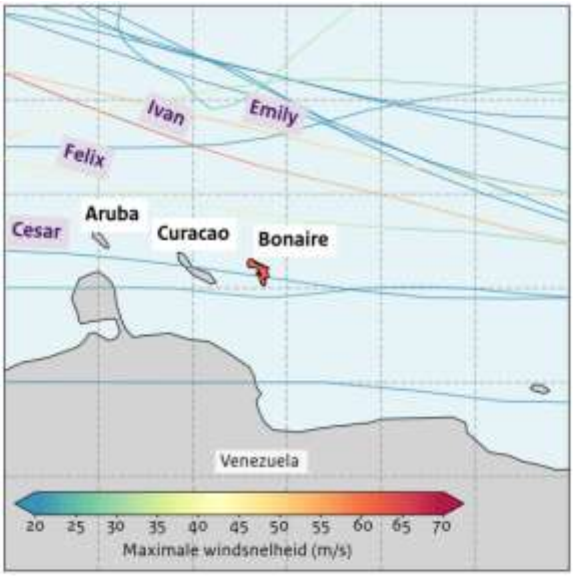
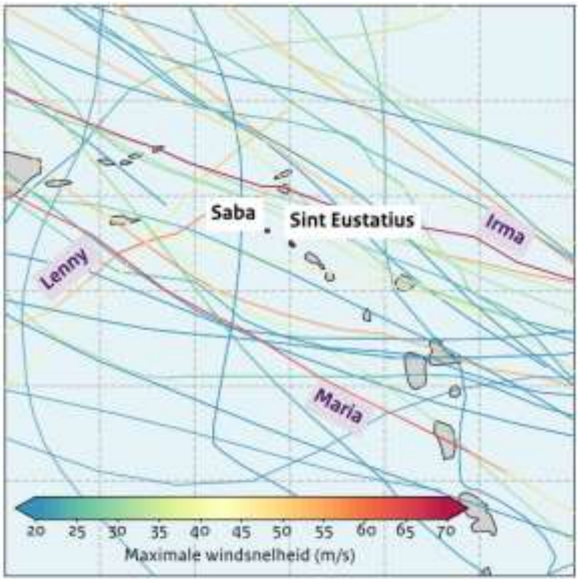
	
Bonaire (leeward islands)	Saba and St. Eustatius and Saba (windward islands)

Figure 4. *Hurricanes with windspeeds over 18 m/s within a 250 km radius between 1981-2020 (KNMI, 2021).*

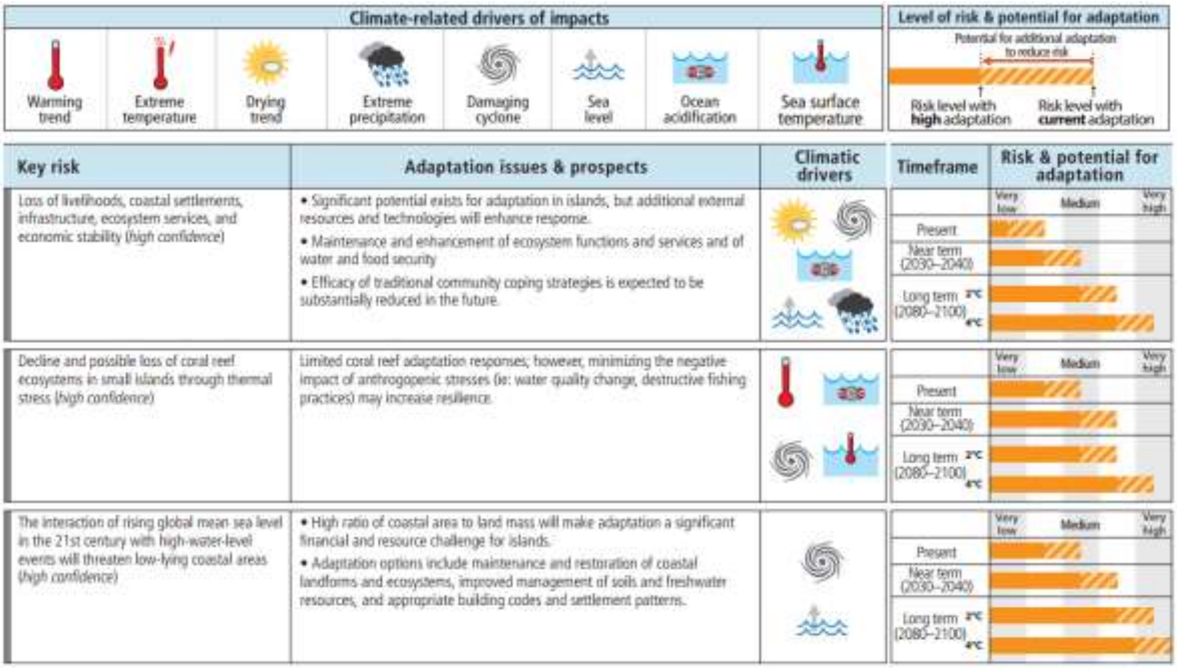


Figure 5. *Climate risks for small islands (Nurse et al., 2014).*

Impacts of Climate Change Within the Caribbean

The climate affects all natural systems and the functions they provide that are important for the Caribbean Netherlands (see *Figure 6*), including:

Coastal effects – Barrier coral reefs are dying due to warmer and more acidic seawater (coral bleaching) (Frieler et al., 2013). Additionally, the intensity of extreme storms that damage coral reefs, mangrove forests, and seagrass beds is increasing. A potential shift or widening of the hurricane belt to the south would also increase hurricane/storm risks for the leeward islands of the Dutch Caribbean.

Fisheries – The deterioration of coral reefs as fish habitat negatively impacts the entire food chain, including important commercial fish species such as snappers (Bari and Cochrane, 2011). Shifts in migration patterns of key deep-sea fishery target species due to warming ocean water could also have negative effects.

Exotics and pests – A warmer and more humid climate provides favourable conditions for mosquito populations (and the associated risk of related diseases such as West Nile virus, Dengue Fever (Mokhtar, 2024) of which the Antilles experienced an outbreak around 2010 and in 2023, Chikungunya, and Zika) and increases the likelihood of foodborne infections (e.g., Salmonella) and animal infections (e.g., Lyme) (EPA, 2014; de Hamer, 2015).

Mass stabilization and erosion control – Coral bleaching leads to coral death, and corals then no longer produce sediment. An increase in the intensity of storms, and possibly their frequency (more uncertainty exists regarding the latter), will erode coasts and beaches (Esteban et al., 2009). Healthy vegetation holds soil in place; at the coast, but also inland. An increase in extreme drought and rainfall will affect vegetation health and increase erosion.

Biodiversity – The islands are part of the international Caribbean biodiversity hotspot based on species richness and the presence of endemic species (Myers et al., 2000; Roberts et al., 2005) but face significant and increasing human pressure. This includes coral reefs, seagrass beds, mangrove forests, salt flats, cactus landscapes, and cloud forests. All habitats are strongly influenced by the climate; for instance, rainforests and cloud forests are sensitive to extreme drought and damage from severe storms (van 't Hof, 2010). Sandy beaches are warming, causing sea turtle eggs to become too hot to hatch, and causing changes in sex ratios to occur. Also, sandy beaches disappear underwater due to rising sea levels, resulting in the loss of nesting habitat for sea turtles (Laloë et al., 2016; Patino-Martinez et al., 2014; Fish et al., 2005). Hurricanes can decimate the island populations of endangered or vulnerable species (Van den Burg et al. 2022; Rivera-Milán et al., 2021). Furthermore, climate change and deteriorating habitat conditions create more opportunities for invasive species to establish themselves (Winkel, 2003). Finally, because in the tropics temperatures are already closer to the lethal maximum for most higher life forms than in temperate and polar regions, such areas are also believed to be more sensitive to the effects of global warming (Calosi et al., 2008; Gutiérrez-Pesquera et al., 2016; Diamond, 2017). Major species shifts can be expected.

Tourism – Rising temperatures, an increased likelihood of severe storms, and dead or deteriorating coral reefs, cactus landscapes, rainforests, and cloud forests, along with diminishing (coral) sandy beaches, make the area less attractive as a tourist destination. The current impacts of climate change on the nature and biodiversity of the Dutch Caribbean islands are summarized below based on the categories by Nurse (2014):

- Loss of coastal habitat (quality) – very unfavourable
- Coral bleaching – very unfavourable
- Elevation shifts in cloud forest – moderately unfavourable
- Acidification of surface waters – moderately unfavourable
- Deterioration of groundwater – moderately unfavourable
- Coastal erosion – moderately unfavourable
- Declining coastal fish catch – moderately unfavourable
- Loss of terrestrial habitat quality – moderately unfavourable.

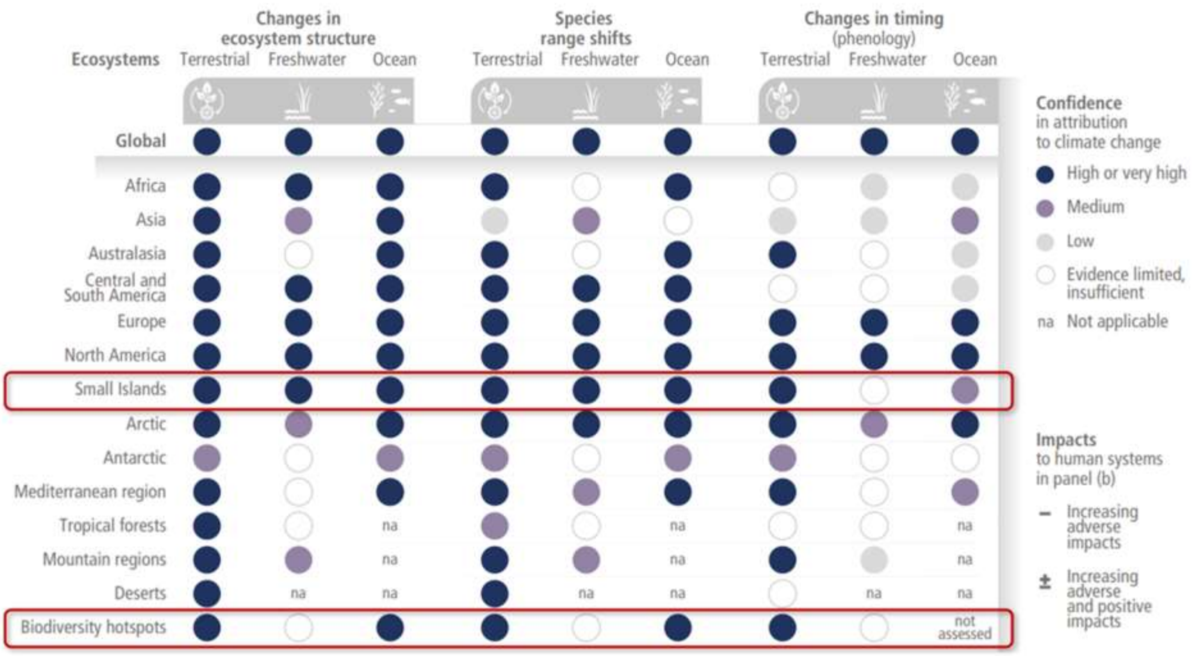


Figure 6. *Observed Impacts of Climate Change on ecosystems (IPCC, 2022). Small island and biodiversity hotspots, both characterising the Dutch Caribbean islands, are highlighted with a red circle.*

Assessment of Future Prospects

For the future, the islands face a high risk regarding the categories mentioned above (see also *Table 1*). Even optimistic scenarios about climate change (RCPs) and biological evolutionary adaptation predict dramatic prospects for coral reefs (Frieler et al., 2013; Lindeboom and Jackson, 2016), cloud forests, and rainforests.

Many of the effects of global climate change are beyond the control of small island nations. In the Caribbean, the largest costs are associated with storm damage, loss of tourist revenue, and damage to infrastructure. The impact of hurricanes Irma and Maria in September 2017 is illustrative of the size of potential damage. The Netherlands prepared 550 million euros for the Island of Sint Maarten in 2017 (Algemene Rekenkamer, 2018). Smaller occurrences, such as the recent floods in Kralendijk are illustrative (see box) of a high exposure to flooding in the built environment and coastal areas.

Annual costs for the Caribbean region are estimated at USD 22 billion around 2050 and USD 46 billion around 2100, which represent 10% and 22% of the total Caribbean economy, respectively. It is important to note that these figures pertain only to the three largest damage categories resulting from climate change, assuming no action is taken (Bueno et al., 2008). The results regarding threat status are summarised in Table 2.

Table 1. *Expected climate risk to natural and human systems.*

	Climate change impact (11)	Risk at 1°C (17)	Risk at 1.5°C (17)	Risk at 2°C (17)
Ecosystem				
Coral	Negative	H	VH	VH
Coastal wetlands	Negative	M[a]	H[a]	H[a]
Mangrove	Negative	U	M	M
Human system				
Fisheries	Negative	M	H	VH
Tourism	Negative	U-M	M	M

[a]The specified data use coastal flooding risk provided in Reference 17.

Abbreviations: H, high; M, moderate; U, undetectable; VH, very high. (11, 17).

Table 2. *Summary overview of the threat status of climate change to biodiversity of the Caribbean Netherlands via habitat impacts and Future prospects.*

Aspect Climate Change	2024
Habitat	Unfavourable-bad
Future prospects	Unfavourable-bad
Overall Assessment of Threat Status	**Unfavourable-bad**

Heavy rainfall and climate change on Bonaire – flood event November 2022

Bonaire – one of the Dutch Caribbean islands – experienced heavy rainfall in November 2022. This event led to flooding in the urban area of Kralendijk and damaged the coral reefs in the Marine Park surrounding the islands. Extreme weather events and their subsequent impacts add to the many challenges the island is already facing, managing tourism and influx of new inhabitants, high erosion rates, rapid urban expansion, wastewater management, and reversing the degradation of terrestrial and marine ecosystems. The outlook of climate change - changing weather patterns and sea level rise – underpins the urgency to start working on climate resilience in Bonaire.

Bonaire is situated in the so-called Southern Caribbean Dry Zone and is characterized by a semi-arid to arid climate, with a distinguishable dry and rainy season, and sustained moderate easterlies (Caribbean Meteorological Department Curaçao, n.d.; Verweij et al., 2020). The dry season runs from February till June, whereas the rainy season starts in September and ends in January. The months of July and August can be considered as transitional months. During the rainy season, rain showers occur usually during the early mornings or early to late evening hours (Meteorological Department Curaçao, n.d.; Schmutz et al., 2017).

From June to November, but especially from August to October, Atlantic tropical cyclones pose a significant threat to communities in the Caribbean. True hurricanes are relatively rare at the latitudes of Bonaire compared to the rest of the Caribbean, as Bonaire is situated on the southern fringes of the

Atlantic hurricane belt. However, hurricanes passing by at relatively short distance, and less-intense tropical storms and depressions and the associated hazards of heavy rainfall and large swells can still cause significant damage on Bonaire (Bries et al., 2004; Scheffers and Scheffers, 2006).

Neglecting the natural environment in the future development of Bonaire will exacerbate many of these issues but also misses an opportunity to let nature aid Bonaire in its societal challenges. Instead, restoring the natural environment can improve the climate resilience of Bonaire while simultaneously addressing several key issues like biodiversity loss and flood security. The concept of using nature to enhance resilience is known as 'Nature-Based Solution' (NbS). Tackling urban flooding and the impact of heavy rain on the Marine Park requires an integral approach with hybrid solutions. NbS implementations can support in ameliorating flood resilience on Bonaire. A study on NBS concluded that restoring and revitalizing the natural system on Bonaire has potential (De Boer et al., 2023).

Runoff over the coral reefs near Kralendijk (Bonaire) after heavy showers (Photo: Caspar Douma, 8 November 2023)

Recommendations for National Conservation Objectives

The vulnerability of natural and human systems to climate change must be reduced within the Caribbean Community (CARICOM countries) (CCCCC, 2012). By removing anthropogenic stressors, ecosystems become more resilient and better able to withstand climate change (IPCC, 2022). The NEPP for the Caribbean Netherlands (Min. LNV et al., 2020) states: "It is not possible to influence climate change from the islands; however, it is possible to improve the resilience of ecosystems so that they can better withstand changes and minimize the consequences." The key sectors for conservation policy are spatial planning and terrestrial and marine nature conservation policy (Debrot and Bugter, 2010), by interweaving nature in all sectors (i.e. 'Nature Inclusive', Verweij et al., 2020), as well as building regulations, maintenance and restoration of coastal areas, habitats, and improved management of soil and freshwater resources (IPCC, 2022).

The Kralendijk Declaration (2016) confirms that the communities of the Caribbean region are threatened by the combined effects of climate change along with ecological degradation resulting from local human activities:

- The Caribbean coasts will face the consequences of more frequent and intense storms and rising sea levels.
- Caribbean landscapes and cultural heritage will be impacted or even destroyed by a combination of poor management and coastal erosion.
- Coastal ecosystems are one of the most important (economic) resources for the livelihoods of Caribbean communities. Population development and the associated pressure on ecosystems, combined with climate change, require a re-evaluation of how people live and utilize the coast.

- Environmental disruption from increasing coastal development, climate change, and rising sea levels will negatively affect tourism, which is the primary source of income for many Caribbean islands.

Climate change can often be viewed as an additional pressure factor on top of other pressures, many of which are caused by human activities. Solutions must therefore be developed in conjunction with these other factors.

Key Threats and Management Implications

Coastal Protection Through Spatial Planning
Rising sea levels and increased intensity of tropical storms pose a direct threat to all coastal human constructions (Min. HEN, 2014). Additionally, these infrastructures disrupt the proper functioning of natural coastal protections such as reefs and mangroves, and they destroy the coastline as a green-blue connection zone that many animals depend on for survival, such as land crabs, hermit crabs, and freshwater shrimp. A spatial policy aimed at implementing a coastal development set-back from the shoreline (setback policy) has numerous economic and ecological benefits (IUCN, 2007; Debrot and Bugter, 2010).

Increased Resilience of Ecosystems by Maintaining or Strengthening Connections Between Ecosystems
Healthy ecosystems have a higher resilience to the pressures of climate change. An ecosystem encompasses all habitats necessary for communities of organisms in all their life stages. Furthermore, these habitats need to be large enough and connected to each other to function effectively (Soule and Simberloff, 1986). A coherent system of nature reserves with connection zones contributes to greater resilience and robustness of systems (van der Sluis et al., 2004) and allows species to adapt their range to changing climatic conditions and vegetation zoning (Cormont, 2011; Vonk et al., 2010).

Reduced Erosion Through Reforestation and Protection of the Food Web by Combating Overfishing

Warmer and more acidic seas make coral sensitive to bleaching and mortality. Coral is vital for coastal defence by serving as wave breakers, generating sediment supply, providing habitat for fisheries, and supporting dive tourism. Additional stressors, such as suffocating erosional materials from land, nutrient enrichment from wastewater (Gast et al., 1999; Duyl et al., 2002; Slijkerman et al., 2011), or disruption of the food web balance due to overfishing (Roberts, 1995; Coblentz, 1997), make coral reefs more vulnerable to the effects of climate change.

Sensitivity to erosion is determined by many factors such as geology, terrain slope, rainfall levels but in wilderness areas is primarily exacerbated by free-roaming livestock (goats, donkeys, cattle, and chickens) (de Freitas et al., 2005; Debrot et al., 2013; Coolen, 2015), which destroy soil-holding vegetation while simultaneously causing mechanical erosion. The hardening of the substrate (buildings, (semi-) paved roads) accelerates water runoff, leading to increased erosion, especially in combination with poor drainage, such as a lack of water buffering areas and flow-reducing systems. Free-roaming livestock also creates opportunities for invasive species (such as Coralita and Rubber vine) to establish themselves and weaken the health of terrestrial ecosystems. Smith et al. (2014) provide comprehensive advice on how to curb the spread of invasive species, including border controls, a mandate to remove invasive species (on private land), making resources available for action, and monitoring for early intervention.

Reforestation of damaged areas counters erosion and offers a chance for ecosystem recovery. Recent reforestation efforts in Curaçao and (Klein) Bonaire have been successful (Debrot, 2009). A reforestation plan has been proposed for Saba (Debrot, 2006).

Overfishing has resulted in the disappearance of (endangered) large grouper (Toller et al., 2010), creating a niche for invasive species such as the Lionfish, which is now spreading across all islands

(Debrot and Bugter, 2010). In addition to the existing ban on spear fishing, a control mechanism could be implemented for the amount of fish caught, fishing methods, and the species and size limits.

Data Quality and Completeness

The climate is a global system with regional differences. Local data has been used for the current situation. The KNMI (Royal Netherlands Meteorological Institute) monitors meteorological data in the Caribbean Netherlands. The scenarios for the future come from global models that have been specified for island regions such as the Caribbean (Nurse et al., 2014). Bugler et al. (2017) warn that higher resolution can be misinterpreted (more precision does not necessarily mean more accuracy). However, downscaling increases overall uncertainty. As part of the PRECIS project (Providing Regional Climates for Impact Studies), Tailor et al. (2013) are working on a Caribbean regional model that may provide more reliable detailed predictions for the Caribbean in the future. This method is being applied in regional training workshops. Besides the official meteorological stations on the islands, also numerous personal weather stations that are openly available can be used as a data source for event analysis (De Boer et al., 2023).

The climate has usual episodes of extreme weather phenomena. Annual measurements of climate parameters (temperature, precipitation, number of cyclones, etc.) are not a direct representation of the climate. For management, it is effective to monitor effect indicators: species and habitats. However, the number of monitored indicators is low, making it difficult to make quantifiable statements. Verweij et al. (2015) recommend that, in addition to maintaining ongoing monitoring activities, several monitoring activities should be added to track the health of habitats. It is important to analyse how observed changes are related to climate change and other environmental and socio-economic developments. However, existing uncertainties do not justify ignoring the aforementioned serious threats.

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Conclusions and Recommendations

Assessment of Conservation State and Comparisons to the EU

Based on the newly available assessments in this report, we conclude that, without exception, the combined SoN in the Caribbean Netherlands must be assessed as unfavourable to unfavourable-bad. However, it is important to note that due to the general lack of data after 2020, our assessment cannot fully measure the more recent effects of the NEPP as implemented in 2020. Figure 1 provides an overview of the pooled assessments for habitats and species/species groups within the Caribbean Netherlands, and allows a comparison with the latest update in the EU (EEA, 2020). Almost, without exception, the current Conservation State (CS) of biodiversity in the Caribbean Netherlands is assessed as unfavourable to unfavourable-bad. This applies to both the habitats and the species(-groups) that depend on them. This appears a bit worse than the situation in the European Union where a significant portion of assessed habitats (15%) and species(-groups) (27%) are considered as being in a favorable CS (Fig. 1).

In the previous assessment period (see Debrot et al., 2018), 45% of assessed habitats and 50% of assessed species and species groups were considered to be in an unfavourable-bad CS. The percentages of habitats and species or species groups assessed to be in an unfavorable-bad state has increased to respectively, 60% and 71%. This may not be due to a measurable decline in CS, but due to the inclusion of several more sensitive species or species groups which were not included in the 2017 assessment. The focus for conservation purposes has been on species/species groups that are at risk (like bats and butterflies) and not species that are widespread and abundant (like the Bananaquit, *Coereba flaveola*, and the Tropical Mockingbird, *Mimus gilvus*). Because most monitoring worldwide (and in Europe) is done on common and widespread species (Forister et al., 2023), composite metrics can easily hide declines in sensitive species, whereas in the Caribbean Netherlands most of our work has focused on sensitive species. Also, composite trends over time should really be compared using the identical habitats and species. With sufficient numbers of species and hopefully the research needed for a quantitative update, a valid comparison of temporal changes in CS should be possible with the next SoN reporting.

The CS for both habitats and species/species groups assessed in this reporting period is lower than in the prior reporting period but (in both periods) the assessed habitats remain in a better CS than species/species groups (Fig. 1). In the EU, the opposite is the case whereby the percentage of species/ species groups) in a favorable CS is higher than the percentage of habitats.

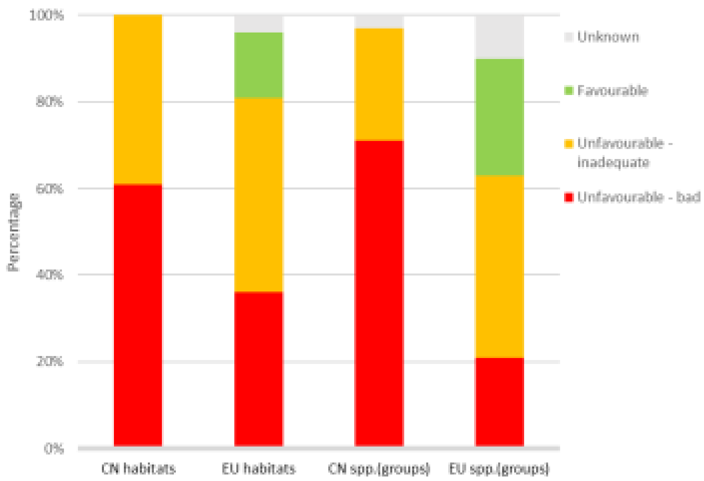

Figure 1. *Assessment of the current Conservation State (CS) (2024) of 18 habitats and a selection of 31 species(-groups) in the Caribbean Netherlands with a comparison to the most recent assessment for the EU as a whole (EU data from: EEA, 2020).*

For the European Netherlands, the opposite is also the case whereby in 2022, 61% of species were not seriously threatened but only 38% of habitat coverage was considered to be fairly high to high in terms of CS (IPO and LNV, 2023).

The opposing comparisons between the CS of habitats between the Caribbean Netherlands and the EU (and the European Netherlands) we especially ascribe to the still fairly low influence of urbanization, agriculture and industry on natural habitats of the Caribbean Netherlands. Only recently is urbanization pressure coming into play on Bonaire and, to a lesser extent on St. Eustatius. As for the quite contrasting comparisons between the CS of species/species groups between the Caribbean Netherlands and the EU, we especially ascribe that to the large number of rare species that survive on these islands in critically low population sizes. Low population sizes are inherent to the small sizes of islands that provide low total habitat availability. However, this generalization is in no way meant to negate the major effects of habitat degradation which have clearly also occurred in the Caribbean Netherlands (e.g., chapters 3, 4 and 10, this report).

When examining the four indicators (distribution, area, quality, future prospects) used to assess habitats (on which the species depend), 43% of the scores are "favourable" (Fig. 2a). Habitat distribution and habitat area score most favourable while habitat quality scores least favourable and future perspective is a mix of "unknown" and low scores. Habitats already in poor condition lack the resilience needed to withstand the current and future impacts of climate change and will likely be unable to sustain sufficient populations of vulnerable species. As climate change is further difficult to influence directly, there is an urgent need for holistic management measures to reduce the cumulative stressors on ecological systems.

Using the same four indicators to assess species status, 31% of the scores are 'favourable' (Fig. 2b). For species, distribution scores are most favourable while population size and habitat suitability have the highest proportion of unfavourable-bad scores, while future prospects are scored principally as unfavourable-inadequate. Hence, in the Caribbean Netherlands, habitat distribution and area, as well as species distribution are least of an issue, but population size and habitat quality are all-around poor. Future prospects for habitats are unfavourable-inadequate to uncertain and for species unfavourable-bad to unfavorable-inadequate.

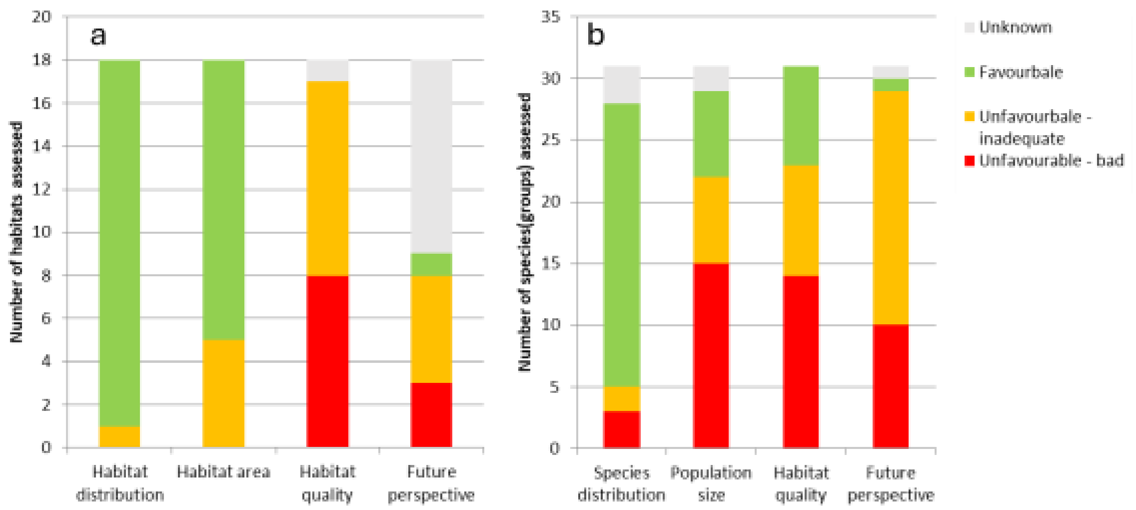

Figure 2. *Scores on the various indicators used to assess the Conservation State of a) 18 habitats and b) 31 species(-groups).*

A Handful of Drivers of Decline

Two key findings from the most up-to-date EU State of Nature report (EEA, 2020) were that climate change is a rising threat, and that agricultural activities, land abandonment and urbanization are the major pressures on habitats and species, followed by pollution. As for the Caribbean Netherlands, climate change is also clearly a rising threat, whereas the factors agriculture, land abandonment and urbanization, only the latter has started to develop into a more serious threat.

Natural/semi-natural habitat areas are relatively much more abundant (in terms of cover percentage) in the Caribbean Netherlands than in the EU where terrestrial Natura 2000 areas on land only are 18% of the land surface and marine Natura 2000 areas are only 10% of marine waters (EEA, 2020). In the EU human land-use has longer been a major threat. Today, on Bonaire increasing urbanization is taking place around the capital of Kralendijk and infringing on rural and wilderness areas, and seemingly unbridled urbanization is starting to become a factor in the evergreen forests on the south-western slopes of the Quill volcano of St. Eustatius.

Aside from climate change, in the Caribbean Netherlands key threats to habitats and species at present inside terrestrial natural habitat areas, are roaming livestock and invasive species. Of course, for specific habitats, additional factors come into play. For colonial nesting birds (like terns and flamingos) and bats, which depend on quiet caves for pupping and roosting, human recreational disturbance can be added as a growing risk. Because in the tropics, temperatures are already closer to the lethal maximum for most higher life forms than in temperate and polar regions, such areas are also believed to be more sensitive to the effects of global warming (Calosi et al., 2008; Gutiérrez-Pesquera et al., 2016; Diamond, 2017).

For marine habitats, climate change is also a serious risk while for coral reefs in particular, the eutrophication of coastal waters and diseases must be added as a key threat (e.g., Pepe et al., 2025), including excess fishing pressure in near-shore areas.

All these factors may be seen as partially linked to increasing urbanization and tourism development which go hand in hand with a rapidly increasing human population size (for Bonaire). For Bonaire, with a current population of around 24,000, and projected to grow towards 50,000 by 2050 (CBS, 2023) and with 80% of sewage produced on the island estimated to enter the coastal zone through dysfunctional septic tanks and cesspits and not through the sewage treatment plant (Haskoning, 2023), the prospects for coral reef recovery are not bright. Even at the governmental level, the environmental risks of further population growth are not acknowledged, as the governments of Bonaire and the Netherlands have agreed to facilitate even further growth of the population (Rijksoverheid, 2024). Current population size increases and clustered urbanization for Bonaire wouldn’t even be so problematic if it were not for the lack of sufficient restrictions (to recreational densities and behavior) or environmental safeguards (like sewage treatment, development planning and guidelines for land clearance). More strict enforcement of existing regulations is also urgently needed. It must be kept in mind that nature can not only be managed merely through nature policies but requires incorporation by and integration through other policy areas. Nature policy does not function independently from other essential policy areas such as land use, spatial planning, agriculture, waste(water)management, tourism, immigration, and economic development. However, the focus in this report is on those policy issues directly affecting CS and which normally fall inside the scope of nature management.

For the mangroves of Lac Bay in Bonaire, accumulated sediments (from runoff due to overgrazing and other suboptimal land use practices) which reduce the aquatic surface of the bay and thereby destroy mangrove and seagrass habitat, can be identified as the principal threat (Debrot et al., 2019). Added to this are unrestrained and excessive recreational use of the bay which is a threat to water quality and the sea grass beds due to trampling (Eckrich and Holmquist, 2000; Skilleter et al., 2006; Debrot et al., 2012), as well as to the larger iconic fauna that depend on these habitats. These include the (IUCN) *Vulnerable* Rainbow Parrotfish, the Queen Conch and the *Endangered* Green Turtle.

Finally, new research on contaminants leaching from the landfill at Lagun suggests that, in addition to eutrophication and bacterial water-quality stressors, serious chemical contaminants are an emerging environmental threat to marine habitat quality, certainly around Bonaire (de Leijer et al., 2023; Dogruer et al., 2024).

Our assessments show that none of the habitats studied are considered to be in a favorable CS (Fig. 1). The habitats in the poorest condition are the terrestrial vegetation habitats affected by goats, the beach habitat affected by sea-level rise, intensive development and recreational pressure by man, and nearshore reefs and seagrass habitats affected by local overfishing, sedimentation, aquatic pollution impacts and climate change-related meteorological effects. Conversely, habitats where disturbance and exploitation by goats and man are less, are in better shape. Examples are the mangrove habitat, saltpans and salt lakes, and cave systems, all of which are areas that are less visited or used by man or goats. Included in this category of less-impacted habitats are also the deep sea habitat which is difficult for man to influence directly and algal fields which are principally found on the Saba Bank and along the exposed eastern coasts of the islands where human disturbance is restricted due to the heavy wave and surf conditions.

Of the species/species groups studied this time, the fraction found in unfavourable-bad Conservation State has worsened from 50% to 71% since the last assessment (Debrot et al., 2018). For the marine species and species groups studied, overfishing and habitat degradation (coral reef decline) are principal factors impacting their CS. For terrestrial species/species groups, the three main deleterious factors causing a reduced CS are overgrazing, principally by uncontrolled roaming livestock (which cause aridification, erosion, plant species loss and greater vulnerability to climate change; Debrot et al., this issue), predation by invasive predators (foremost of which are the feral cat; van den Burg et al., this issue) and genetic swamping due to introduced invasive iguanas (van den Burg et al., this issue). Hence, all three of these impacts are directly ascribable to the overarching problem of invasive alien species.

Since the last inventories (2011 and 2012), no less than 710 new records of non-native (exotic) species have been recorded in the wild on one or more Dutch Caribbean islands. This is an average increase of no less than 54 species per year. However, the current rate of increase is certainly higher because the process is exponential and which means that the "average" always underestimates the most recent status. Invasive species are a veritable flood and while they are an enormous risk to biodiversity, only few have so far only been addressed in pilot studies and short-term opportunistic projects.

Best chances for combatting or preventing invasive species are on land. A great deal of research has been spent on the Lionfish but, quite predictably, there is little that can be meaningfully done about this species that achieves its highest densities in deep waters well-beyond safe diving limits. Most urgent is to address invasive species structurally throughout the Caribbean Netherlands. This is because, following habitat loss (due to a variety of factors), invasive species are considered the second-most important threat to biodiversity world-wide; Kaiser, 1999; Mooney, 2001). For instance, according to the US national Wildlife federation, approximately 42 percent of endangered species on land in the USA are at risk of extinction due to invasive species (https://www.nwf.org/Educational-Resources/Wildlife-Guide/Threats-to-Wildlife/Invasive-Species). In the Caribbean Netherlands the situation is at least as urgent as in the USA. In the Caribbean region there are several Red List predatory invasive species for which accidental introduction to the Caribbean Netherlands would be no less than disastrous (e.g., Mongoose, Boa Constrictor). However, at present there is no legislation or effective control to prevent such an introduction, neither accidental nor purposeful.

Intervention Approach

The passive biodiversity management approach, in which management came down to simply putting a "fence around nature" and managing park visitor access and behaviour is no longer sufficient. This old approach to natural areas management was based on the premise that nature was still resilient enough to bounce back on its own. That approach may work in very large undisturbed tracks of nature but today we know that under current circumstances this will not be sufficient. The impact of man's activities and invasive species has become so large and pervasive, that the old passive approach to nature conservation no longer is sufficient. For the 2020-2030 first phase of the NEPP for the Caribbean Netherlands, the Kingdom has made available a total budget of 35 million Euro's (IPO and LNV, 2023). Therefore, for the coming years there should be considerable scope for achieving some principal objectives.

Active intervention to give nature a helping hand has become more important than ever and essential to reversing negative feedback loops (for instance between overgrazing, plant species loss and climate vulnerability). Also, in the Caribbean Netherlands this realization has come with many recent initiatives to intervene, and these have often been proving successful. Examples are the success in vegetation recovery on Klein Bonaire following goat removal (Debrot, 1997; Debrot, 2016; Proosdij et al., this issue) as well as the success in various reforestation initiatives aimed at bringing back rare and endangered plants, such as (among others) on Klein Curaçao (Debrot, 2015). Other examples of the successes of active intervention are those that show that removing cats from the wild can save many smaller animals such as endangered seabirds from being killed (Terpstra et al., 2015; Debrot et al., 2022a; 2023), the successful removal of invasive iguanas which is key to saving the genetic integrity of endangered island populations of iguanas (Debrot et al., 2022b; van den Burg et al., 2023), and the construction of artificial islands to protect nesting terns against predation exposure (Bertuol et al., 2015).

Other initiatives that show potential and are bearing results are the culturing of corals for outplanting (Cook et al., 2022; Dehnert et al., 2023), the use of artificial reefs to help restore fish populations (Hylkema et al., 2020) and efforts with potential to contain or even eradicate invasive Giant African Land Snails (Debrot et al., 2016). Also, the importance of joint management to maintain and improve productive fisheries, such as those of the Saba Bank and St Eustatius (Amelot et al., 2021; Brunel et al., 2021), appear high. Fisheries management for both these areas are based on active and productive cooperation between science, management and the fishing sector (Fig. 3).

So today, much more than ever, nature protection and management are much more than maintaining territorial integrity of conservation areas as it has been practiced for decades. A much more active approach to nature management in the Caribbean Netherlands continues to grow especially now that different pilot projects have shown potential. Now it is time to institutionalize this intervention approach into nature and park management for lasting success. However, in selecting from the wide range of possible interventions it is important to prioritize those which have broad and proven impact.

Figure 3. *LVV Bonaire and WMR in conversation with the fishing sector, November 2023. Photo: LVV, Bonaire).*

The current NEPP (Min. LNV, 2020) for the Caribbean Netherlands has a total of 96 points requiring serious attention. Many of the mentioned threats are already significantly being addressed within the implementation agenda of the current NEPP (Min. LNV et al., 2020). For instance, these include action points for a) the prevention and control of invasive species, b) the control of free roaming livestock, c) effective waste and wastewater management, d) investments in sustainable fisheries, e) coral reef restoration, and f) the conservation of keystone and flagship species. While ambition is good to have, too much ambition (96 items on the agenda) can also complicate decision making, especially in light of limited funding. Where do we begin and what comes first? Therefore, in suggesting and setting priorities for action it may be helpful to focus on actions that have multiple cascading benefits instead of actions directed to single species solutions. An example would be the issue of coastal eutrophication. Adequately addressing that issue will help not only all endangered coral species but also all coral reef fish species which depend on shelter created by corals and on clean water to remain free from disease. Addressing the problem of roaming livestock is another good example. Removal of goats allows vegetation to recover, which will reduce vulnerability to erosion, increase available habitat for threatened terrestrial species, and help prevent sedimentation runoff that stresses reefs and contributes to climate resilience. As climate change is difficult to influence directly, there is an urgent need for holistic management measures that together reduce the cumulative stressors on ecological systems. Many such interventions can be considered as being " Nature-based Solutions" (NbS) whereby interventions focus on key issues whereafter most recovery will be based on the natural resilience of the system. For instance, instead of planting mangroves and culturing larval fish, the less-complicated removal of sediments threatening mangrove wetlands can restore water depth for both mangroves and larval fish. Once suitable conditions are created, then nature largely takes over. Another example would be how the efficient removal of a small island population of an invasive predator (cats) was sufficient to open the way for major seabird recovery on Klein Curaçao (Debrot et al., 2023).

Exceptions to this rule might be highly specific actions needed to safeguard iconic endemic species or populations. An example of that might be to remove small populations of highly impactful invasive species before they spread and become unmanageable (like invasive iguanas that interbreed with native iguanas; see van den Burg et al., this issue). Nevertheless, in suggesting priorities, ideally, the focus needs to be on "holistic" actions, which are preferably "nature-based" and that trigger cascading positive benefits.

Priority Conservation Actions

Terrestrial Nature

1) Address the roaming livestock and poultry issues through intensified culling and modernization of husbandry practices (e.g., Neijenhuis et al., 2015). (All three islands with variations per island: Feral goats and chickens for Saba; Feral goats, chickens and cattle on St. Eustatius and; Feral goats, swine and donkeys on Bonaire)
2) Focus reforestation efforts on propagating and reestablishing rare and endemic plant species. (All three islands; priority species differ per island)
3) Take special measures at seabird breeding sites to combat invasive predators (cats and rats) and eradicate them if feasible (such as on uninhabited islands or outcrops). (All three islands, with as exception that rats only seem to be a major problem on Saba, but not Bonaire or St. Eustatius)
4) Introduce and enforce biosecurity legislation and measures to stem the flood of invasive species, and establish rapid action teams to reduce risks of accidental introduction of deleterious species and carry out needed eradications. (All three islands are in crisis on this matter; see also Smith et al., 2014)
5) Continue culling of invasive iguanas (introduced to Saba and St. Eustatius) with as aim eradication of non-native iguana presence. Also, identify other bio-risk species which still are in an early stage of introduction and eradicate those which are still feasible for successful eradication.

Marine Nature

1) Address the problem of eutrophication and pollution of nearshore waters, particularly around Bonaire, by implementing large-scale sewage effluent treatment and reuse. (Urgent especially on Bonaire due to high and growing population pressure)

2) Restore freshwater catchments on land to limit unnecessary runoff while recharging depleted subterranean water tables. (All three islands, but especially urgent on Bonaire)
3) Assess and address the apparent emerging pollution threat emanating from landfills of Bonaire. (All three islands have urban landfills close to the sea)
4) Introduce measures directed at reducing the vulnerability to fishing for the formerly abundant large reef piscivores (using any of a combination of measures like area closures, fish reserves, size limits, gear restrictions) (All three islands, but most urgent on Bonaire due to the high and growing population pressure)
5) In addition to keeping mangrove channels clear, also dredge and reuse eroded topsoil from the backwaters of Lac Bay in order to restore water depth needed for healthy mangrove and fish habitat. Circular reuse of dredged sediment helps close the sediment cycle by restoring soil fertility on land, without the need for costly imports. (Only relevant to Bonaire as only Bonaire has mangrove has conditions suitable for mangrove habitat)

The ranking of priority conservation actions differs between islands. For ease of overview, the results are ranked by urgency as follows: none = green; low = yellow; intermediate = orange; high = red; highest = dark red in Table 1.

Table 1. *Priorities in conservation action and intervention for the three islands of the Caribbean Netherlands ranked according to urgency from none (green) to most high (dark red).*

Interventions for Terrestrial Nature

	Bonaire	Saba	St. Eustatius
1) Controlling/culling feral livestock and poultry			
goats	red	red	red
cattle	green	green	orange
swine	yellow	green	green
donkeys	orange	green	yellow
chickens	green	red	red
2) Reforestation of rare plant species			
various	red	red	red
3) Controlling introduced predators at seabird breeding sites			
cats	red	red	yellow
rats	green	red	green
4) Introduce biosecurity legislation and establish rapid action teams			
	dark red	dark red	dark red
5) Continue successful and start new removal campaigns for non-native species where still feasible			
green iguana	green	dark red	dark red
New Guinea flatworm	dark red	green	green

Interventions for Marine Nature

	Bonaire	Saba	St. Eustatius
1) Marine eutrophication and pollution			
	red	yellow	yellow
2) Restore freshwater catchments			
	red	yellow	orange
3) Address emergent pollution from landfills			

4) Reduce fishing pressure on large reef piscivores		
5) Magrove habitat restoration		
restore water depth		
restore connectivity		

Monitoring Needs

In the European Netherlands, trends for nearly all major species groups, such as birds, butterflies, and plants, are monitored through the *Netwerk Ecologische Monitoring* (NEM). Most of the NEM monitoring networks are carried out by Private Data-Managing Organisations (*PGO's*), and Statistics Netherlands (CBS) processes these data into nature statistics, enabling close tracking of nature and policy results. NEM serves as the backbone of terrestrial nature monitoring in the European Netherlands, ensuring high data quality and availability. In addition, it is worth mentioning the program for Statutory Research Tasks (WOT; "Wettelijke Onderzoekstaken"), and the aquatic monitoring program of the Netherlands Ministry of Infrastructure and Water management ("Monitoring Waterstaatkundige Toestand des Lands"; MWTL) that both fund a large portion of the ecosystem and environmental monitoring needs in the European Netherlands. The Caribbean Netherlands, which boasts the highest biodiversity within the Kingdom of the Netherlands, and are highly vulnerable to climate change and other environmental pressures, does not have any comparable monitoring systems. To assess the results of biodiversity policies in the future and meet international obligations (e.g., Cartagena, CBD, CMS, SPAW and Ramsar Conventions), trend analyses are essential. This requires selecting indicator species and parameters and developing monitoring plans. However, so far very little biological monitoring has actually taken place in the Caribbean Netherlands, even for the many legally protected species (see Appendix 1).

A few notable exceptions regard coral reefs (Meesters et al., this issue), sea grass beds (van der Geest and Engel, this issue), sea turtle nesting (Dogruer et al., this issue) and fisheries (Debrot et al., this issue). Past and current level of investment in monitoring is insufficient to accurately track population trends as can be witnessed by wide error margins for even those species/species groups for which some monitoring is available. In addition, the KNMI provides some selected long-term monitoring of meteorological parameters. Other than that, practically no long-term monitoring efforts are available for the Caribbean Netherlands. Firstly, monitoring programs are costly and such activities have been considered to primarily be a (management) responsibility residing first and foremost with the island public entities and the designated management entities. Hopeful in this respect is that the need for monitoring is given ample attention in the implementation agenda of the 2020-2030 NEPP, where it is mentioned no less than six times (Min. LNV et al., 2020).

A consequence of this all is that very few of the conservation state assessments in this report are based on long timeseries of consistently and uniformly collected measurements. Most are based on accumulated results of opportunistic and chance research that was lucky enough to be completed. For reliable and consistent assessments, consistent monitoring of important habitats and species (and environmental pressures) should ideally become statutory research tasks, as should be the storage and accessibility of such data (such as currently being done within the Caribbean Biodiversity Data Base (CBDB) project of Wageningen).

Monitoring priorities for the Caribbean Netherlands should be to:

1) Decide on selected monitoring required to assess and evaluate the success of chosen management interventions which are intended to reverse certain declining trends (see Verweij et al., 2015).

2) Aside from monitoring single dependent variables through time (like flamingo counts) always also monitor relevant independent variables (like salinity, temperature or food density) to be able to assess the causes of changes and trends.
3) Focus monitoring on indicator variables and indicator species that preferably are also endemic, rare and/or endangered. For instance, monitoring the endemic but also hardy and ubiquitous subspecies of the Caribbean Mockingbird, *Mimus gilvus rostratus*, has little added value even though it is endemic.

A list of top monitoring priorities is presented in Table 2.

Research Needs

Analysing trends can indicate correlations between monitored variables and suggest causality but typically additional research is needed to demonstrate real cause and effect. Also, the ecology of many Caribbean Netherlands species and especially rare endemic species remains practically unstudied. Most research from the past has had an "academic" observational focus on ecosystem functioning but very little work has been further done on the "applied" question of how to best enhance or restore certain systems or species. Therefore, significant additional applied research will be needed to understand different systems and species and on how to best protect and restore them. As in the case of monitoring, priorities for hypothesis-driven research also need to be decided. Storage of, and accessibility of, the resulting data, reports and publications (such as currently being done within the CBDB project) are essential.

Research priorities should include:

1) Quantitative baseline understanding of species functioning and use of different systems. Such data also happen to be an important starting point for biodiversity monitoring. Examples to mention are the total lack of quantitative community baselines for the various habitats found on the Saba Bank (e.g., Meesters et al., 2024) and the vegetations of the steep inner slopes of the Quill on St. Eustatius (van Proosdij et al., this issue).
2) Quantitative understanding of the distribution, abundance and ecological conservation needs of the many endemic and rare species of the islands.
3) Investigate which conservation measures and interventions can be most effective for ecosystem and species recovery.

A breakdown of key conservation research priorities is presented in Table 3.

Table 2. *Priorities in monitoring needs for the three islands of the Caribbean Netherlands.*

Conservation Monitoring Priorities

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1) Environmental indicators

meteorology of terrestrial habitats
marine water quality
environmental contamination

2) Biological indicators

forest cover and species
seabirds and land birds
butterflies
several protected terrestrial species (e.g., iguanas, bats)
coral, seagrass and mangroves
fish communities
fish catches

sea turtles
marine mammals

3) Biodiversity management and intervention effectiveness

Public awareness and support
Invasive species at ports of entry
Eradication and control programs
Fishing reserves
Propagation and reforestation success

Table 3. *Priorities in conservation research needs for the three islands of the Caribbean Netherlands.*

Conservation Research Priorities
1) Baseline habitat descriptions and faunal use
St. Eustatius Quill crater vegetation
Bat shelter habitat of Saba and St. Eustatius
Saba Bank benthic habitats
St. Eustatius benthic habitats
Bonaire east coast algal fields and reefs
Fish species distribution Saba Bank
2) Ecological research on endangered species
Quantifying rare plant abundance and distribution
Ecological needs of rare and endemic animals
Nest habitat use and needs for endemic iguanas
3) Research into intervention options and effectiveness
Livestock and poultry removal
Predator control for seabirds
Mangrove restoration
Artificial reefs
Reef restoration
Fishing reserves

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Quality Assurance

Wageningen Marine Research utilises an ISO 9001:2015 certified quality management system. The organisation has been certified since 27 February 2001. The certification was issued by DNV.

Justification

Report: C001/25
Project Number: 4318100345 and 4318100346

The scientific quality of this report has been peer reviewed by a colleague scientist and a member of the Management Team of Wageningen Marine Research

Approved: dr. A.O. Debrot (reviewed chapters 1, 2, 3, 4, 5, 8, 10, 14, 18, 19, 22, 23, 28)
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Approved: dr. E.H.W.G. Meesters (reviewed chapters 7, 9, 11, 20, 24, 25)
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Business Manager Projects

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Date: 30 June 2025

Appendix 1. Species with direct policy relevance in the Caribbean Netherlands

Species of policy relevance in the Caribbean Netherlands based on their status on the IUCN Red List, SPAW, CMS and/or CITES treaty lists. Bird species for which the population in the Caribbean Netherlands exceeds 1% of the total population (Ramsar-criteria) are in bold. The (potential/expected) presence of these species in the various habitats found in the Caribbean Netherlands are also indicated, as are the availability of suitable monitoring data.

Latin name	English common name	Dutch common name	Available data	TTropical mist, rainforest	Dry tropical forest	Caves	Salinas	Coastal cliffs	Beaches	Mangroves	ZSeagrass, algal fields	Coral reefs	Open sea and deep sea	IUCN Category	SPAW Annex	CMS Annex	CITES Appendix
Plants																	
Cedrela odorata	Spanish cedar	Spaanse Ceder	Little		x									VU			
Swietenia mahagoni	West Indian Mahogany	Mahokboom	Little		X									EN			
Guaiacum officinale	Common Lignum Vitae	Pokhout	Little		X									EN	3		II
Guaiacum sanctum	Holywood Lignum Vitae	Pokhout	Little		X									EN	3		II
Nectandra krugii	Black Sweet Wood		Little		X									EN			
Zanthoxylum flavum	West Indian Satinwood		Little		X									VU			
Rhizophora mangle	Red Mangrove	Rode Mangrove	Little							X				LC	3		
Avicennia germinans	Black Mangrove	Zwarte Mangrove	Little							X				LC	3		
Laguncularia racemosa	White Mangrove	Witte Mangrove	Little							X				LC	3		
Conocarpus erecta	Buttonwood	Mangelboom	Little							X				LC	3		
Syringodium filiforme	Manateegrass	Zeegras	Little								X			LC	3		
Thalassia testudinum	Turtlegrass	Zeegras	Fair								X			LC	3		
Halophila baillonis	Clovergrass	Zeegras	Little								X			VU	3		

Latin name	English common name	Dutch common name	Available data	TTropical mist, rainforest	Dry tropical forest	Caves	Salinas	Coastal cliffs	Beaches	Mangroves	ZSeagrass, algal fields	Coral reefs	Open sea and deep sea	IUCN Category	SPAW Annex	CMS Annex	CITES Appendix
Halophila decipiens	Paddlegrass	Zeegras	Little								X			LC	3		
Halodule wrightii	Shoalgrass	Zeegras	Little								X			LC	?		
Ruppia maritima	Wigeongrass	Snavelruppia	Little								X			LC	3		
Mammals			Little														
Tursiops truncatus	Bottlenose Dolphin	Tuimelaar	Little										X	LC	2		II
Lagenodelphis hosei	Fraser's Dolphin	Sarawakdolfijn	Little										X	LC	2		II
Delphinus capensis	Long-beaked Common Dolphin	Kaapse Dolfijn	Little										X	LC	2		II
Stenella attenuata	Pantropical Spotted Dolphin	Slanke Dolfijn	Little										X	LC	2		II
Stenella coeruleoalba	Striped Dolphin	Gestreepte Dolfijn	Little										X	LC	2		II
Grampus griseus	Risso's/Grey Dolphin	Gramper	Little										X	LC	2		II
Ziphius cavirostris	Cuvier's Whale	Dolfijn van Cuvier	Little										X	LC	2		II
Mesoplodon europaeus	Gervais's Beaked Whale	Spitssnuitdolfijn van Gervais	Little										X	DD	2		II
Pseudorca crassidens	False Killer Whale	Zwarte Zwaardwalvis	Little										X	DD	2		II
Orcinus orca	Orca - Killer Whale	Orka	Little										X	DD	2	2	II
Kogia breviceps	Pygmy Sperm Whale	Dwergpotvis	Little										X	DD	2		II
Kogia simus	Dwarf Sperm Whale	Kleinste Potvis	Little										X	DD	2		II
Peponocephala electra	Melon-headed Whale	Witlipdolfijn	Little										X	LC	2		II
Globicephala macrorhynchus	Shortfin Pilot Whale	Indische Griend	Little										X	DD	2		II
Balaenoptera borealis	Coalfish Whale	Noordse Vinvis	Little										X	EN	2	1	I
Balaenoptera edeni	Bryde’s Whale	Edens Vinvis	Little										X	DD	2	2	I
Balaenoptera musculus	Blue Whale	Blauwe Vinvis	Little										X	EN	2	1	I

Latin name	English common name	Dutch common name	Available data	TTropical mist, rainforest	Dry tropical forest	Caves	Salinas	Coastal cliffs	Beaches	Mangroves	ZSeagrass, algal fields	Coral reefs	Open sea and deep sea	IUCN Category	SPAW Annex	CMS Annex	CITES Appendix
Balaenoptera physalus	Fin Whale	Gewone Vinvis	Little										X	EN	2	1	I
Megaptera novaeangliae	Humpback Whale	Bultrug	Little										X	VU	2	1	I
Physeter macrocephalus	Sperm Whale	Potvis	Little										X	VU	2	1	I
Trichechus manatus	West-indian Manatee	Caribische/West-Indische Zeekoe	Little								X			VU	2		I
Leptonycteris curasoae	Lesser Longnosed Bat	Curaçaose Bladneusvleermuis	Fair		X	X								VU			
Birds																	
Amazona barbadensis	Yellow-shouldered Amazon	Geelvleugelamazone	Good		X									VU	2		I
Aratinga pertinax	Brown-throated Conure	West-Indische Parkiet	Little		X									LC			II
Buteo albicaudatus	White-tailed Hawk	Witstaartbuizerd	Little		X									LC			II
Buteo jamaicensis	Red-tailed Hawk	Roodstaartbuizerd	Little		X									LC			II
Caracara cheriway	Northern Caracara	Caracara	Little		X									LC	2		II
Falco sparverius	American Kestrel	Amerikaanse Torenvalk	Little		X									LC			II
Falco columbarius	Merlin	Smelleken	Little		X									LC			II
Tyto alba	Barn Owl	Kerkuil	Little		X									LC			II
Chrysolampis mosquitus	Ruby-topaz Hummingbird	Rode Kolibri	Little		X									LC			II
Chlorostilbon mellisugus	Blue-tailed Emerald	Groene Kolibri	Little		X									LC			II
Eulampis jugularis	Purple-throated Carib	Granaatkolibri	Little	X										LC			II
Eulampis holosericeus	Green-throated Carib	Greenkeelkolibri	Little		X									LC			II
Orthorhyncus cristatus	Antillean Crested Hummingbird	Antilliaanse Kuifkolibri	Little		X									LC			II
Cinclocerthia ruficauda	Brown Trembler	Siddersportlijster	Little	X										LC	2		
Contopus cooperi	Olive-sided Flycatcher	Sparrenpiewie	Little		X									NT			

Latin name	English common name	Dutch common name	Available data	TTropical mist, rainforest	Dry tropical forest	Caves	Salinas	Coastal cliffs	Beaches	Mangroves	ZSeagrass, algal fields	Coral reefs	Open sea and deep sea	IUCN Category	SPAW Annex	CMS Annex	CITES Appendix
Dendroica caerulea	Cerulean Warbler	Azuurblauwe Zanger	Little		X									VU			
Pterodroma hasitata	Black-capped petrel	Zwartkapstormvogel	Little										X	EN	2		
Puffinus lherminieri	Audubon's Shearwater	Audubon's Pijlstormvogel	Little	X									X	LC	2		
Dendrocygna arborea	West Indian Whistling Duck	West-Indische Fluiteend	Little							X				VU	3	2	
Dendrocygna bicolor	Fulvous Whistling Duck	Rosse Fluiteend	Little							X				LC	3	2	
Sarkidiornis melanotos	Comb Duck	Knobbeleend	Little							X				LC		2	II
Pelecanus occidentalis	Pelican	Bruine Pelikaan	Little							X				LC	2		
Pandion haliaetus	Osprey	Visarend	Little							X				LC		2	II
Egretta rufescens	Reddish Egret	Roodhalsreiger	Good							X				NT			
Patagioenas leucocephala	White-crowned pigeon		Little												3		
Phoenicopterus ruber	Flamingo	Caribische Flamingo	Good				X							LC	3	2	II
Calidris canutus rufa	Red Knot	Kanoet	Little				X		X					-		1	
Calidris pusilla	Semi-palmated Sandpiper	Grijze Strandloper	Little				X		X					NT		1	
Charadrius melodus	Piping Plover	Dwergplevier	Little				X		X					NT	2		
Fulica caribaea	Caribbean Coot	Caribische Koet	Good				X							NT			
Sterna antillarum antillarum	Least Tern	Dwergstern	Good				X		X				X	LC			
Sterna dougallii dougallii	Roseate Tern	Dougal's Stern	Little				X		X				X	LC	2	2	
Sterna hirundo	Common Tern	Visdief	Good				X		X				X	LC			
Thalasseus sandvicensis	Cayenne Tern	Grote Stern	Good				X		X				X	LC			
Phaethon aethereus	Red-billed tropic bird	Roodsnavelkeerkringvogel	Good					X					X	LC			
Falco peregrinus	Peregrine Falcon	Slechtvalk	Little					X						LC	2	2	I
Reptiles			Little														
Alsophis rufiventris	Red-bellied Racer	Roodbuik Grasslang	Little	X	X									VU			

Latin name	English common name	Dutch common name	Available data	TTropical mist, rainforest	Dry tropical forest	Caves	Salinas	Coastal cliffs	Beaches	Mangroves	ZSeagrass, algal fields	Coral reefs	Open sea and deep sea	IUCN Category	SPAW Annex	CMS Annex	CITES Appendix
Crocodylus acutus	American crocodile	Amerikaanse krokodil	Little									X		VU	2		I
Iguana delicatissima	Lesser Antillean Iguana	Antillenleguaan	Fair	X	X									EN	3		II
Iguana iguana	Green Iguana	Groene Leguaan	Little		X									-	3		II
Chelonia mydas	Green Turtle	Groene Zeeschildpad	Good						X		X			EN	2	1	I
Eretmochelys imbricata	Hawksbill Turtle	Karetschildpad	Good						X			X		CR	2	1	I
Caretta caretta	Loggerhead Turtle	Onechte Karetschildpad	Good						X			X		LC	2	1	I
Lepidochelys olivacea	Olive Ridley	Warana	Good						X			X		VU	2	2	I
Dermochelys coriacea	Leatherback Turtle	Lederschildpad	Good						X				X	VU	2	1	I
Fishes																	
Alopias superciliosus	Bigeye Thresher Shark	Grootoogvoshaai	Little										x	VU			II
Alopias vulpinus	Thresher Shark	Voshaai	Little									X		VU			II
Carcharhinus falciformis	Silky Shark	Zijdehaai	Little										x	NT			II
Carcharhinus leucas	Bullshark	Stierhaai	Little										x	NT			
Carcharhinus limbatus	Blacktip shark	Zwartpunthaai	Little										x	NT			
Carcharhinus longimanus	Oceanic Whitetip Shark	Witpunthaai	Little										X	CR	3		II
Carcharhinus perezi	Caribbean Reef Shark	Caribische Rifhaai	Fair									X		NT			
Carcharodon carcharias	Great White Shark	Witte Haai	Little										X	VU		1, 2	II
Cetorhinus maximus	Basking Shark	Reuzenhaai	Little										X	VU		1, 2	II
Galeocerdo cuvier	Tiger Shark	Tijgerhaai	Little										x	NT			
Hexanchus griseus	Bluntnose Sixgill Shark	Stompsnuitzeskieuwhaai	Little										X	NT			
Isurus oxyrinchus	Shortfin Mako	Kortvin Makreelhaai	Little									X		VU		2	
Isurus paucus	Longfin Mako	Langvin Makreelhaai	Little									X		VU		2	
Negaprion brevirostris	Lemon Shark	Citroenhaai	Little										x	NT			

Latin name	English common name	Dutch common name	Available data	TTropical mist, rainforest	Dry tropical forest	Caves	Salinas	Coastal cliffs	Beaches	Mangroves	ZSeagrass, algal fields	Coral reefs	Open sea and deep sea	IUCN Category	SPAW Annex	CMS Annex	CITES Appendix
Pristis pectinata	Smalltooth Sawfish	Kleintand Zaagvis	Little							X				CR	2		I
Rhincodon typus	Whale Shark	Walvishaai	Little										X	EN	3	2	II
Sphyrna mokarran	Great Hammerhead Shark	Grote Hamerhaai	Little									X	X	EN	3		II
Sphyrna lewini	Scalloped Hammerhead	Geschulpte Hamerhaai	Little									X	X	EN	3		II
Sphyrna zigaena	Smooth Hammerhead	Gladde Hamerhaai	Little									X	X	VU	3		II
Manta birostris	Giant Manta Ray	Reuzenmanta	Little										X	VU	3	1, 2	II
Manta alfredi	Reef Manta Ray	Manta Alfredi	Little									X		VU	3		
Aetobatus narinari	Spotted Eagle Ray	Gevlekte Adelaarsrog	Little									X		NT			
Albula vulpes	Bone Fish	Gratenvis	Little										X	NT			
Anguilla rostrata	American Eel	Amerikaanse Paling	Little										X	EN			
Hippocampus reidi	Slender Seahorse	Zeepaardje	Little								X	X		DD			II
Hippocampus erectus	Lined Seahorse	Zeepaardje	Little								X	X		VU			II
Thunnus albacares	Yellowfin Tuna	Geelvintonijn	Little										X	NT			
Dermatolepis inermis	Marble Grouper	Gemarmerde Zeebaars	Fair								X	X		NT			
Epinephelus flavolimbatus	Yellowedge Grouper	-	Fair								X	X		VU			
Epinephelus itajara	Goliath Grouper	Reuzenzeebaars	Fair							X	X	X		CR			
Epinephelus morio	Red Grouper	Rode Zeebaars	Fair								X	X		NT			
Epinephelus niveatus	Snowy/Spotted Grouper	Gevlekte Zeebaars	Fair								X	X		VU			
Epinephelus striatus	Nassau Grouper	Nassaubaars	Fair							X	X	X		EN	3		
Balistes capriscus	Grey Triggerfish	-										X		VU			
Balistes vetula	Queen Triggerfish	Koningin Trekkervis	Fair									X		VU			
Lachnolaimus maximus	Hogfish	Zwijnsvis	Fair									X		VU			
Lutjanus analis	Mutton Snapper	Schaapssnapper	Fair							X	X	X		VU			

Latin name	English common name	Dutch common name	Available data	TTropical mist, rainforest	Dry tropical forest	Caves	Salinas	Coastal cliffs	Beaches	Mangroves	ZSeagrass, algal fields	Coral reefs	Open sea and deep sea	IUCN Category	SPAW Annex	CMS Annex	CITES Appendix
Lutjanus cyanopterus	Cubera Snapper	Cubera Snapper	Fair							X	X	X		VU			
Lutjanus synagris	Lane Snapper	-	Fair									X		NT			
Mola mola	Ocean Sunfish	Maanvis	Fair										x	VU			
Mycteroperca bonaci	Black Grouper	Zwarte Zeebaars	Fair								X	X		NT			
Mycteroperca interstitialis	Yellowmouth Grouper	Geelbekbaars	Fair								X	X		VU			
Mycteroperca venenosa	Yellowfin Grouper	Geelvinbaars	Fair								X	X		NT			
Rhomboplites aurorubens	Vermilion Snapper	-	Little									X		VU			
Scarus guacamaia	Rainbow Parrotfish	Regenboog Papegaaivis	Little							X	X	X		VU			
Thunnus obesus	Bigeye Tuna	Grootoogtonijn	Little										X	VU			
Thunnus thynnus	Atlantic Bluefin Tuna	Blauwvintonijn	Little										X	EN			
Thunnus alalunga	Albacore Tuna	Witte Tonijn	Little										X	NT			
Corals																	
Acropora palmata	Elkhorn Coral	Elandgeweikoraal	Fair									X		CR	2		II
Acropora cervicornis	Staghorn Coral	Hertshoornkokraal	Fair									X		CR	2		II
Agaricia lamarcki	Lamarck's Sheet Coral	Lamarck's Plaatkoraal	Fair									X		VU	3		II
Agaricia tenuifolia	Thin Leaf Lettuce Coral	Dun Bladkoraal	Fair									X		NT	3		II
Dendrogyra cylindrus	Pillar Coral	Pilaarkoraal	Fair									X		VU	3		II
Dichocoenia stokesi	Elliptical Star Coral	Elliptisch Sterkoraal	Fair									X		VU	3		II
Mycetophyllia ferox	Rough Cactus Coral	Ruw Cactuskoraal	Fair									X		VU	3		II
Millepora striata	Bladed Box Firecoral	Brandkoraal	Fair									X		VU	3		II
Oculina varicosa	Large Ivory Coral	Ivoorkoraal	Fair									X		VU	3		II
Orbicella annularis	Head Star Coral	Kinderhoofdjeskoraal	Fair									X		EN	2		II
Orbicella faveolata	Boulder Starcoral	Pagodekoraal	Fair									X		EN	2		II

Latin name	English common name	Dutch common name	Available data	TTropical mist, rainforest	Dry tropical forest	Caves	Salinas	Coastal cliffs	Beaches	Mangroves	ZSeagrass, algal fields	Coral reefs	Open sea and deep sea	IUCN Category	SPAW Annex	CMS Annex	CITES Appendix
Orbicella franksi	Bumpy Star Coral	Bobbelig Sterkoraal	Fair									X		VU	3		II
Porites branneri	Blue Crust Coral	-	Fair									X		NT	3		II
Invertebrates																	
Conus aurantius	Golden Cone	-	Little									X		NT			
Strombus gigas	Queen Conch	Grote Kroonslak	Good								X	X		-	3		II
Insects																	
Danaus plexippus	Monarch Butterfly	Monarch Vlinder	Little		X									-		2	

Appendix 2. List of Protected Species for the Island Ordinance Nature Management Bonaire (A.B. 2008, no. 23)

INFORMATIEBLAD BESCHERMDE DIER- EN PLANTENSOORTEN BONAIRE

Eilandsverordening natuurbeheer Bonaire

Inleiding

Het toerisme is een belangrijke pijler van onze economie. En ons toerisme steunt voor een belangrijk deel op onze natuur. We danken onze welvaart en ons welzijn voor een groot deel aan onze natuur. Daarom moeten we er zorgvuldig mee omgaan. Nos naturalesa ta nos teroso! De nieuwe Eilandsverordening natuurbeheer Bonaire maakt bescherming van onze natuur mogelijk. Dit informatieblad legt uit welke dieren en planten op ons eiland worden beschermd en waarom.

Beschermd door verdragen

In onze regio, dus ook op Bonaire, zijn sommige dieren beschermd door verdragen. Dat geldt bijvoorbeeld voor de lora, maar ook alle zeeschildpadden, dolfijnen en walvissen. Het gaat hierbij om soorten die wereldwijd of in het Caribische gebied met uitsterven worden bedreigd.

De bescherming van deze dier- en plantensoorten wordt geregeld in de Eilandsverordening natuurbeheer Bonaire. Volgens artikel 11, lid 1 worden alle dier- en plantensoorten beschermd die zijn genoemd in:

- bijlage 1 van het CITES-Verdrag (Convention on International Trade in Endangered Species of Wild Fauna and Flora);
- bijlage 1 van de Bonn-Conventie (Verdrag inzake de bescherming van migrerende wilde diersoorten);
- bijlagen 1 en 2 van het SPAW-Protocol (Protocol concerning Specially Protected Areas and Wildlife in the Wider Caribbean Region);
- bijlage 1 van het Zeeschildpaddenverdrag.

Beschermd door het eilandgebied

Met de nieuwe Eilandsverordening natuurbeheer kan het bestuurscollege zelf ook dier- en plantensoorten aanwijzen die bescherming verdienen.
(artikel 11, lid 2). Dat is belangrijk, want op ons eiland kan de situatie voor bepaalde soorten anders zijn dan in de regio of wereldwijd.

Voor het samenstellen van de lijst van te beschermen soorten die per 1 september 2010 van kracht is geworden zijn de volgende voorwaarden gehanteerd. De soorten op de lijst
dienen aan één of meer voorwaarden te voldoen.

- Rode lijst soorten. Vermelding op de rode lijst van bedreigde soorten van de World Conservation Union, IUCN, categorie CR (*critically endangered*), categorie EN (*endangered*) of categorie VU (*vulnerable*).
- Endemisch (niet elders voorkomend) en daarnaast zeldzaam, bedreigd of andere overwegingen.
- Lokaal bedreigd of zeldzaam.
- Ecologisch belang (sleutelsoorten).
- Onderhevig aan grote exploitatie druk.
- Toeristische waarde (vlaggeschip soorten).
- (Potentieel) verzamelobject.
- Handhavingsoverwegingen. De verschillende soorten zijn door leken niet uit elkaar te houden, daarom wordt de hele groep beschermd.

Overigens is het los van de bescherming van soorten ook verboden om zonder vergunning van het bestuurscollege levende of dode dieren of planten en delen of producten hiervan uit een natuurpark mee te nemen. Dit geldt niet voor de traditionele visserij in het onderwaterpark voor zover dit is toegelaten.

Beschermde dier- en plantensoorten op Bonaire
op grond van de Eilandsverordening natuurbeheer Bonaire

Latijnse naam	Papiamentse naam	Nederlandse naam	Engelse naam	
Zeezoogdieren				
Balaenoptera acutorostrata	bayena	dwergvinvis	minke whale	▲
Balaenoptera edeni	bayena tompoes, topo	Brydevinvis Brydewalvis	Bryde's whale	▲
Balaenoptera physalis	bayena	vinvis	fin whale	▲
Delphinus delphis	dòlfein	gewone dolfijn	common dolphin	▲
Globicephala macrorhynchus	kabes di keshi	Indische griend kortflippergriend	shortfin pilot whale	▲
Grampus griseus		grijze dolfijn gramper	grey dolphin	▲
Kogia breviceps		dwergpotvis	pygmy sperm whale	▲
Kogia simus		kleinste potvis	dwarf sperm whale	▲
Lagenodelphis hosei		sarawakdolfijn dolfijn van Fraser	Fraser's dolphin	▲
Megaptera novaeangliae	bayena	bultrug	humpback whale	▲
Mesoplodon europaeus		spitssnuitdolfijn van Gervais	Gervais's beaked whale	▲
Orcinus orca		orka zwaardwalvis	orca, killer whale	▲
Peponocephala electra		witlipdolfijn, witlipgriend, veeltandgriend, elektradolfijn	melon-headed whale	▲
Physeter catodon	kachalote	potvis	great sperm whale	▲
Pseudorca crassidens		zwarte zwaardwalvis	false killer whale	▲
Stenella attenuata	dòlfein	slanke dolfijn, pantropische gevlekte dolfijn	pantropical spotted dolphin	▲
Stenella clymene		clymenedolfijn	Atlantic spinner dolphin	▲
Stenella coeruleoalba		gestreepte dolfijn	striped dolphin	▲
Stenella frontalis		Atlantische vlekdolfijn	Atlantic spotted dolphin	▲
Stenella longirostris	toniwa	langsnuitdolfijn, spinnerdolfijn	spinner dolphin	▲
Tursiops truncatus	dòlfein	grote tuimelaar	bottlenose dolphin	▲
Ziphius cavirostris		dolfijn van Cuvier	Cuvier's beaked whale	▲
Haaiachtigen				
Aetobatus narinari	chuchu águila	gevlekte adelaarsrog	spotted eagle ray	•
Dasyatis Americana	chuchu ròk	Amerikaanse pijlstaartrog	southern stingray	•
Manta birostris	manta	mantarog	manta ray	•
Selachimorpha (Euselachii)	tribon	haaien	sharks	•
Zeereptielen				
Caretta caretta	kawama	dikkopzeeschildpad	loggerhead	▲
Chelonia mydas	tortuga blanku	groene zeeschildpad	green seaturtle	▲
Dermochelys coriacea	drikil	lederrugzeeschildpad	leatherback	▲
Eretmochelys imbricata	karet	karetschildpad	hawksbill	▲
Lepidochelys olivacea (kempi)	bastardo	warana	olive ridley	▲
Zeevissen				
Balistes vetula	pishiporko rabu di gai	koningstrekkervis	queen triggerfish	•
Dermatolepis inermis	olitu		marbled grouper	•
Epinephelus itajara	djukvis	itajara	Goliath grouper, jewfish	•
Epinephelus striatus	jakupepu	Nassau tandbaars	Nassau grouper	•
Lachnolaimus maximus	hokfis	everlipvis	hogfish	•
Lutjanus analis	kapitán	snapper	mutton snapper	•

Latijnse naam	Papiamentse naam	Nederlandse naam	Engelse naam	
Lutjanus cyanopterus	karaña pretu	cubera snapper	cubera snapper	●
Pagrus pagrus	djent'i maishi	rode zeebrasem	red porgy	●
Scaridae	gutu	papegaaivissen	parrotfishes	●
Thunnus obesus	buni wowo grandi	grootoogtonijn	bigeye tuna	●
Ongewervelde zeedieren				
Panulirus argus	kref	kreeft	Caribbean spiny lobster	■
Panulirus guttatus	kref	gevlekte kreeft	spotted spiny lobster	■
Panulirus laevicauda	kref	kreeft	smoothtail spiny lobster	■
Koraalachtigen				
Antipatharia	koral pretu	zwarte koralen	black corals	●○
Gorgoniacea		waaierkoralen	gorgonians	●
Milleporidae		brandkoralen	fire corals	●○
Scleractinia		steenkoralen	stony corals	●○
Stylasteridae		kantkoralen	lace corals	●○
Zeeschelpdieren				
Strombus gigas	karkó	roze vleugelhoorn grote kroonslak	queen conch	●■○
Zeegrassen				
Syringodium filiforme (Cymodocea manitorum)		zeegras	manatee grass	●
Thalassia testudinum	yerba di kaña	zeegras	turtle grass	●
Zoogdieren				
Chiroptera	raton di anochi	vleermuizen	bats	●
Vogels				
Amazona barbadensis	lora	geelvleugelamazone	yellow-shouldered amazon	▲
Aratinga pertinax xanthogenius	prikichi	West Indische parkiet	brown-throated parakeet	●○
Buteo albicaudatus	gabilan di seru, falki	witstaartbuizerd	white tailed hawk	●○
Caracara cheriway	warawara	kuifcaracara	crested caracara	▲
Falco peregrinus	falki peregrino	slechtvalk	peregrine falcon	▲
Margarops fuscates bonairensis	chuchubi Spaño palabrua boka duru	witoogspotlijster	pearly eyed trasher	●
Pandion haliaetus	gabilan piskadó	visarend	osprey	●○
Pelecanus occidentalis	ganshi	bruine pelikaan	brown pelican	▲
Phoenicopterus ruber	chogogo	Caribische flamingo	Caribbean flamingo	●○
Tryngites subruficollis		blonde ruiter	buff-breasted sandpiper	▲
Tyto alba	palabrua	kerkuil	barn owl	●○
Reptielen				
Iguana iguana	yuana	groene leguaan	green iguana	○
Zoetwaterdieren				
Typhlatya monae		blinde garnaal	Mona cave shrimp	●
Mangrovesoorten				
Avicennia germinans	mangel blanku	witte mangrove	black mangrove	■
Conocarpus erectus	mangel, mangel blan-ku	grijze mangrove	buttonwood	■
Laguncularia racemosa	mangel blanku	mangrove	white mangrove	■
Rhizophora mangle	mangel tan	rode mangrove	mangrove	■
Bomen				
Amyris ignea (A. simplicifolia)				●
Capparis tenuisiliqua				●
Celtis iguanaea				●
Clusia sp.	tam machu			●
Crateva tapia	ishiri			●
Euphorbia cotinifolia	manzaliña bobo			●
Ficus brittonii	palu di mahawa, ma-hòk di mondi			●
Geoffroea spinosa (G. superba)	palu di taki			●

Latijnse naam	Papiamentse naam	Nederlandse naam	Engelse naam	
Guaiacum officinale	wayaká	pokhout	lignum-vitae	●■○
Guaiacum sanctum	wayaká shimaron	pokhout	roughbark lignum-vitae	●■○
Guapira fragrans (Pisonia fragans)				●
Guapira pacurero (Pisonia bo-nairensis)	mafobari, mushi bari			●
Krugiodendron ferreum	kaobati			●
Manihot carthaginensis	marihuri			●
Maytenus tetragona (M. sieberi-ana)	palu di kolebra (A)			●
Maytenus versluysii	bèshi di yuana			●
Phoradendron trinervium				●
Sabal cf. causiarum (Sabal sp.)	kabana	sabalpalm	sabal palm	●
Salicornia perennis				●
Schoepfia schreberi	mata combles (A)			●
Spondias mombin	hoba			●
Strumpfia maritima				●
Ximenia americana	kashu di mondi			●
Zanthoxylum flavum (Fagara flava)	kalabari		West Indian satinwood	●
Zanthoxylum monophyllum (Faga-ra monophylla)	bosùa, koubati			●
Planten				
Bromelia humilis (B. lasiantha)	teku	bromelia	bromeliad	●
Cereus repandus (Subpilocereus repandus)	kadushi	boomcactus	candle cactus	○
Melocactus macracanthus	bushi, kabes di indjan, melon di seru	bolcactus	Turk's cap cactus	●■○
Opuntia caracassana (Opuntia wentiana)	infrou, tuna	Spaanse juffer	prickly pear	○
Orchidaceae	orkidia	orchideeën	orchids	● ○
Pilosocereus lanuginosus (Cepha-locereus lanuginosus)	kadushi di pushi, kadushi spaño	zuilcactus	candle cactus	○
Stenocereus griseus (Lemaireocereus griseus, Ritterocereus griseus)	yatu, datu	zuilcactus	candle cactus	○
Tillandsia flexuosa	teku di palu	bromelia	bromeliad	●
Varnes		varens	ferns	●

Legenda

▲ = Beschermde dier- of plantsoort op grond van verdragen.
○ = Beschermde dier- of plantsoort op bijlage 2 van het CITES-Verdrag. Deze soorten mogen niet zomaar worden uitgevoerd naar andere landen.
● = Beschermde dier- of plantsoort aangewezen op grond van de Eilandsverordening natuurbeheer Bonaire.
■ = Beschermde dier- of plantsoort aangewezen op grond van de Eilandsverordening natuurbeheer Bonaire waarvoor ook beheersmaatregelen gelden.

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Appendix 3. Update of exotic (and possibly) invasive species documented in the wild in the Dutch Caribbean islands, as recorded since last Wageningen UR inventory (2011) (the list of plants is most complete for Bonaire, Saba and St. Eustatius but outdated for Curaçao, Aruba and St. Maarten)

Definition: Invasive species are non-indigenous species (or exotic species) introduced by historic human actions, whose introduction causes, or is likely to cause, economic or environmental harm or harm to human health (US government definition)

Colour legend for the 'island' columns:

Pr	Present non-native population
Oc	Occasionally reported
Uc	Unclear
In	Indigenous
Po	Potential non-native species
Px	Previous non-native population present

Column: **habitat** has options: Marine, Land; Freshwater.
Column: **date** means date/year first recorded for a species (if known)

The following species groups are present in the table below (listed here in order of occurrence):
Mammals; Fish; Birds; Amphibians; Reptilia; Mollusca; Flatworms; Earthworms (Annelida); Insects; Animal disease vectors parasites; Plant disease vectors parasites; Other mites and ticks; Fungi; MLO (Mycoplasma Like Organisms); Plants.

Species/species group	common name	habitat	date	Aruba	Bonaire	Curacao	Saba	St Eustatius	St Maarten
Mammals									
Rattus rattus	black rat	Land		Pr	Pr	Pr	Pr	Pr	Pr
Rattus norvegicus	brown rat	Land		Pr	Pr	Pr			
Canis familiaris	dog	Land		Pr	Pr	Pr	Pr	Pr	Pr
Mus musculus	house mouse	Land		Pr	Pr	Pr	Pr	Pr	Pr
Sus scrofa	pig	Land			Pr	Pr	Pr	Pr	
Ovis aries	sheep	Land		Uc	Pr			Pr	
Felix domesticus	cat	Land		Pr	Pr	Pr	Pr	Pr	Pr
Capra hircus	goat	Land		Pr	Pr	Uc	Pr	Pr	Pr
Herpestes auropunctatus	mongoose	Land				Po		Po	Pr
Equus assinus	wild ass	Land		Pr	Pr	Pr	Pr	Pr	
Chlorocebus pygerythrus	vervet monkey	Land					Po	Po	Pr
Procyon minor	raccoon	Land					Po	Po	Pr
Bos sp.	cow	Land						Pr	
Cavia porcellus	guinea pig	Land					Pr		
Oryctolagus cuniculus domesticus	rabbit	Land					Pr		
Fish									
Poecilia reticulata	guppy	Freshwater		Pr	Pr	Pr	Po	Po	Po
Oreochromis mossambica	Mozambique tilapia	Freshwater		Pr	Pr	Pr			Pr
Pterois volitans	red lionfish	Marine		Pr	Pr	Pr	Pr	Pr	Pr
Pterois miles	common lionfish	Marine		Uc	Px	Uc		Pr	

Species/species group	common name	habitat	date	Aruba	Bonaire	Curacao	Saba	St Eustatius	St Maarten
Birds									
Icterus icterus	Venezuelan Troupial			In	Pr	In			
Streptopelia decaocto	Eurasian dove						Pr	Pr	Pr
Passer domesticus	house sparrow			Pr	Pr	Pr	Pr	Pr	Pr
Gallus gallus	Jungle fowl						Pr	Pr	Pr
Sicalis flaveola	saffron finch			Pr	Pr	Pr			
Corvus splendens	house crow					Px			
Quiscalus lugubris	Caribbean grackle			Pr	Pr	Pr			Pr
Quiscalus mexicanus	Boatbilled grackle			In		Uc		Oc	
Bubulcus ibis	cattle egret			Pr	Pr	Pr	Pr	Pr	Pr
Molothrus bonariensis	shiny cowbird			Pr		Pr		Pr	
Ploceus cucullatus	village weaver-bird					Pr			
Eupsittula pertinax	West-Indian parakeet			In	In	In	Pr		
Columba livia	rock dove			Pr	Pr	Pr	Pr	Pr	Pr
Amazona ochrocephala	Yellow-crowned amazon					Pr			
Psittacula krameri	rose-ringed parakeet					Pr			
Estrilda troglodytes	Black-rumped waxbill					Px			
Patagioenas corensis	Bare-eyed Pigeon			In	In	In			Pr
Amphibians									
Eleutherodactylus johnstonei	Johnstone's frog			Pr	Pr	Pr			

Species/species group	common name	habitat	date	Aruba	Bonaire	Curacao	Saba	St Eustatius	St Maarten
Eleutherodactylus martinicensis	Martinique Robber Frog								Pr
Eleutherodactylus planirostris	Greenhouse Frog								Pr
Osteopilus septentrionalis	Cuban tree frog				Pr	Pr	Po	Po	Pr
Bufo marinus	Cane Toad, Marine toad			Pr		Po			
Pleurodema brachyops	dori			In	Pr	Pr			Pr
Reptilia									
Anolis carolinensis	North American Green Anole	Land							Pr
Anolis cristatellus	Puerto Rican Crested Anole	Land		Pr					Pr
Anolis gingivinus	Anguilla Bank Anole	Land		Pr					In
Anolis porcatus	Cuban Green Anole	Land		Pr					
Anolis sagrei	Cuban brown anole	Land		Uc					Pr
Boa constrictor constrictor	Boa	Land		Pr		Po			Oc
Boiga irregularis	Brown tree snake	Land				Po			
Cnemidophorus arenivagus	Rainbow Whiptail	Land		Pr					
Diadophis punctatus	Ringneck Snake	Land				Uc			
Elaphe guttata	Corn snake	Land				Po			
Epictia albifrons	Wagler's Blind Snake	Land			Uc				
Gekko gecko	Tokay gecko	Land				Pr			
Gonatodes albogularis	White-throated Clawed Gecko	Land		Px		Oc			

Species/species group	common name	habitat	date	Aruba	Bonaire	Curacao	Saba	St Eustatius	St Maarten
Gonatodes antillensis	Venezuelan Coastal Clawed Gecko	Land		Px	In	In			
Gonatodes vittatus	Striped Clawed Gecko	Land		Pr		Oc			
Gymnophthalmus underwoodi	Underwcod's Spectaded Tegu	Land					Pr	Pr	Pr
Hemidactylus frenatus	Common House Gecko	Land		Pr	Pr	Pr	Pr	Pr	Pr
Hemidactylus mabouia	African House Gecko	Land		Pr	Pr	Pr	Pr	Pr	Pr
Iguana iguana	Green Iguana	Land		In	In	In	Pr	Oc	Pr
Iguana melanoderma	Melanistic Lesser Antilles Iguana	Land					In		Pr
Ramphotyphlops braminus	Brahminy Blindsnake	Land		Pr		Pr	Pr	Pr	Pr
Lepidodactylus lugubris	Mourning Gecko	Land			Pr	Pr			Pr
Liotyphlops albirostris	Whitenose Blind Snake	Land		Uc		In			
Micrurus fulvius	Eastern Coral Snake	Land				Uc			
Pantherophis guttatus	Eastern Corn Snake	Land				Oc			Uc
Pseudemys floridana	Coastal plain scooter	Land		Uc					
Storeria dekayi	Florida Brown Snake	Land				Oc			
Thamnophis cyrtopsis		Land				Uc			
Thamnophis cyrtopsis subsp. ocellatus		Land				Uc			
Trachemys scripta	Common slider	Freshwater		Pr		Uc		Oc	Oc
Anolis sabanus	Saban Anole	Land					In	Oc	
Epicrates cenchria	Rainbow Boa	Land							Oc

Species/species group	common name	habitat	date	Aruba	Bonaire	Curacao	Saba	St Eustatius	St Maarten
Python curtus-group	Blood Python	Land							Oc
Python regius	Ball Python	Land							Oc
Mollusca									
Allopeas gracile	Graceful Awlsnail, Traveling Tramp	Land		Uc	Oc	Uc	Oc	Pr	Pr
Archachatina marginata	Giant West African Snail, Banana Rasp Snail	Land	Not yet	Po	Po	Po	Po	Po	Po
Bulimulus guadalupensis	West Indian Bulimulus	Land		Po	Po	Pr	Pr	Pr	Pr
Cryptelasmus canteroiana cienfuegosensis		Land				Oc	Oc		
Gulella bicolor	Two-tone Gulella, Toothed Gulella, Carrot snail	Land			Oc				Oc
Hawaiia minuscula		Land			Oc				
Leptinaria unilamellata		Land				Oc			Oc
Limicolaria aurora	Nigerian land snail	Land	Not yet	Po	Po	Po	Po	Po	Po
Lissachatina fulica	Giant African snail, African Giant Snail	Land		Po	Pr	Pr	Po	Pr	Pr
Melanoides tuberculata	Red Rimmed Melania	Land		Po	Pr	Uc	Po	Po	Pr
Naria turdus		Marine		Pr	Pr	Pr	Po	Po	Pr
Neosubulina gloynii		Land				In			Oc
Opeas hannense	Dwarf Awlsnail	Land				Oc	Pr	Pr	Oc
Pallifera sp.	Mantleslugs	Land					Po	Po	Pr
Paropeas achatinaceum	Indonesian Awlsnail	Land		Po	Pr	Po			
Physella acuta	Acute Bladder Snail	Freshwater		Po	Pr	Po	Po	Po	Pr

Species/species group	**common name**	**habitat**	**date**	**Aruba**	**Bonaire**	**Curacao**	**Saba**	**St Eustatius**	**St Maarten**
Planorbella duryi	Seminole Ram's-horn, American ram's horn snail, Florida's Ram's-horn, Miniature ramshorn snail	Freshwater		Po	Pr	Pr		Oc	
Polygyra cereolus	Southern Flatcoil	Land		Po	Pr	Pr			Pr
Praticolella griseola	Vagrant Scrubsnail, Central American Scrubsnail	Land			Oc				Oc
Sagdidae		Land							Oc
Streptartemon glaber		Land			Oc		In		In
Subulina octona	Miniature Awlsnail, Glossy Subulina, Wandering Awlsnail, Subulina Snail	Land		Po	Oc	Pr	Pr	Pr	Pr
Succinea concordialis	Spotted Ambersnail	Land			Oc				
Tomostele musaecola		Land		Uc			Pr	Po	Pr
Veronicellidae	Leatherleaf Slugs	Land		Po	Oc	Po	Oc	Po	Pr
Zachrysia provisoria	Cuban Brown Snail, Cuban garden snail	Land	2000 <	Po	Oc	Pr	Pr	Po	Pr
Flatworms									
Bipalium vagum		Land	2014				Oc	Oc	Oc
Geoplanidae indet		Land	2023	Po	Oc	Oc	Oc	Po	Po
Platydemus manokwari	New Guinean flatworm	Land	2020	Po	Oc	Oc	Oc	Po	Oc
Earthworms (Annelida)									
Pontoscolex corethrurus						Px			Px

Species/species group	common name	habitat	date	Aruba	Bonaire	Curacao	Saba	St Eustatius	St Maarten
Dichogaster affinis				Px		Px			
Dichogaster bolaui				Px	Px	Px			
Dichogaster modighlianii						Px	Px		
Dichogaster saliens								Px	
Polypheretima elongata					Px				
Pontodrilus litoralis				Px	Px	Px	Px		
Eudrilus eugeniae							Px		Px
Ficopomatus miamensis						Pr			
Insects									
Aedes aegyptii	yellow fever mosquito			Pr	Pr	Pr	Pr	Pr	Pr
Aedes albopictus	Asian Tiger mosquito			Po	Po	Po	Po	Po	Po
Aeneolamia reducta			1986			Pr			
Anoplolepis gracilipes	Crazy ant			Po	Po	Po			
Apis mellifera	European Honey Bee			Pr	Pr	Pr	Pr	Pr	Pr
Apis mellifera scutellata	Africanized honey bee			Po	Po	Po	Po	Po	Po
Bactrocera invadens	African fruit fly								
Blatella germanica	german cockroach								
Cardiocondyla emeryi	Emery's Sneaking Ant			Pr	Uc	Pr			Uc
Cardiocondyla mauritanica	Moorish Sneaking Ant				Pr	Pr	Uc		
Carpophilus sp.						Uc			
Caryedon gonagra				Pr		Oc			
Cercyon nigriceps							Pr		

Species/species group	**common name**	**habitat**	**date**	**Aruba**	**Bonaire**	**Curacao**	**Saba**	**St Eustatius**	**St Maarten**
Chalepides barbatus									Pr
Coelophora inaequalis							Uc	Uc	
Coptotermes formosanus	Formosan subterranean termite			Po	Po	Po			
Crocothemis servilia	Crimson darter dragonfly	E/SE Asia							
Cryptolaemus montrouzieri									Uc
Cryptotermes brevis						Pr			
Curelius japonicus							Uc	Uc	
Digitonthophagus gazella	Gazelle Scarab			Uc				Oc	Oc
Hemianax ephippiger	Vagrant Emperor Dragonfly			Uc		Px			
Hybosorus illigeri								Uc	
Hypothenemus hampei				Oc		Uc			
Hypothenemus obscurus						Uc	Pr		
Kallima paralekta	Indian Leafwing								Oc
Labarrus lividus							Uc	Uc	
Leptostylopsis argentatus						Oc			Uc
Monoanus concinnulus							Uc	Oc	
Monomorium floricola	Flower ant	Trop. Asia	<1937	Pr	Uc	Pr	Pr		Pr
Necrobia rufipes	Red-legged Ham Beetle					Uc		Pr	Uc
Nialaphodius nigrita							Uc	Oc	
Palorus cerylonoides								Uc	
Papilio demoleus	Lime Swallowtail						Pr	Pr	Pr
Paratrechina longicornis	Longhorn crazy ant			Pr	Pr	Pr	Pr		Pr

Species/species group	common name	habitat	date	Aruba	Bonaire	Curacao	Saba	St Eustatius	St Maarten
Paratrechina pubens	Hairy crazy ant			Po	Po	Po	Po	Po	Po
Pectinophora gossypiella	Pink Bollworm								Uc
Periplaneta americana	American Cockroach			Pr	Pr	Pr	Pr	Pr	Pr
Periplaneta australasiae	Australian Cockroach								Uc
Pheidole megacephala	African-Big headed Ant			Pr		Pr	Pr		
Plagiolepis alluaudi	Alluaud's Little Yellow Ant								Oc
Planuncus tingitanus						Uc			
Pseudoazya trinitatis								Uc	Uc
Rhynchophorus ferrugineus				Uc		Oc	Uc		
Rhyparobia maderae					Uc				
Scymnus coccivora								Oc	
Scyphophorus cf. acupunctatus						Uc			
Solenopsis geminata	Tropical fire ant	Trop. S. America, West Indies	<1936	In	In	In	Pr	Pr	Pr
Solenopsis invicta	South American Fire ant			Pr					Uc
Strumigenys emmae	Emma's Dacetine Ant			Pr	Uc	Pr			
Strumigenys membranifera	Membraniferous Dacetine Ant				Uc				Uc
Supella longipalpa					Uc				
Tapajosa spinata	Sharpshooter leafhopper					Pr			
Tapinoma melanocephalum	ghost ant	old world tropics	<1994	Pr	Pr	Pr	Pr		Pr
Technomyrmex difficilis									

Species/species group	common name	habitat	date	Aruba	Bonaire	Curacao	Saba	St Eustatius	St Maarten
Tetramorium bicarinatum	penny ant,	SE Asia	<2007	Pr					
Tetramorium caldarium						Pr			
Tetramorium lanuginosum	wooly ant	Trop. Asia	<2004	Pr		Pr			
Tetramorium simillimum				Pr		Pr			
Trachyscelis aphodioides								Oc	Uc
Trichobaris bridwelli								Oc	
Trichomyrmex destructor	destroyer ant	old world	<1999	Pr	Pr	Pr			Pr
Typhaea stercorea	Hairy fungus beetle							Pr	
Ulomoides ocularis									Uc
Vespula squamosa	yellow jacket wasp	N America		Po	Po	Po			
Wasmannia auropunctata	Little fire ant	Neotropics	<1972			Pr	Pr		Pr
Animal disease vectors parasites									
Amblyomma variegatum							Po	Po	Po
Cochliomyia hominivorax	New World screw–worm fly			Po	Po	Po			
Ixodes	Lyme disease tiks								
Ornithodoros puertoricensis							Pr	Pr	Pr
Varroa destructor	varroa mite		1996	Pr	Pr	Pr	Pr	Pr	Pr
Geckobiella stamii							Pr	Pr	Pr
Plant disease vectors parasites									

Species/species group	common name	habitat	date	Aruba	Bonaire	Curacao	Saba	St Eustatius	St Maarten
Scyphophorus acupunctatus	Agave weevil			Po	Po	Po		Pr	
Cactoblastis cactorum	Opuntia cactus moth			Po	Po	Po	Uc	Pr	Po
Bemisia tabaci	white fly		1989	Po	Po	Pr			
Toxoptera citricida	Black Citrus aphid, Brown Citrus aphid		1989	Po	Po	Uc		Pr	
Cylas formicarius	sweet potato weevil		1990	Po	Po	Pr		Pr	Pr
Thrips palmi	palm thrips		1994	Po	Po	Pr			
Gynaikothrips ficorum	Cuban Laurel Thrips		1996	Po	Po	Pr			
Phyllocnistis citrella	Citrus miner		1996	Pr	Pr	Pr			
Macconellicoccus hirsutus	Pink/Hibiscus Mealy bug		1997	Pr	Po	Pr			
Paracoccus marginatus	Papaya Mealy bug		2002			Pr			
Rhynchophorus ferrogineus	Red palm weevil		2008	Pr		Pr			
Crypticerya genistae	White partridge pea bug		2009			Pr			
Aenolamia varia	spittle bug		1986			Pr			
Aspidiotus destructor	Coconut scale			Pr	Pr	Pr			Pr
Mionochroma vittatum	longhorn beetle					Pr			
Thrips sp.	Tabebuia plague						Uc	Uc	
Other mites and ticks									
Raoiella indica	red palm mite			Po	Po	Po			
Schizotetranychus hindustanicus	Citrus hindu mite		2000	Pr	Pr	Pr			

Species/species group	common name	habitat	date	Aruba	Bonaire	Curacao	Saba	St Eustatius	St Maarten
Fungi									
Claviceps africana	sorghum erot		2003			Pr			
Fusarium of Palms			2005-2010						
Ganoderma zonatum	Ganoderma butt rot of palms					Oc			
Gliocladium of palms			2005-2010			Pr			
MLO's (Mycoplasma Like Organisms)									
Lethal Yellowing of Palms (LYdisease)				Po	Po	Po			
Papaya Bunchy Top (MLO)			early 1960	Po	Po	Pr			
Papaya Ringspot Virus (PRSV-P)			2002	Po	Po	Pr			
Plants									
Abelmoschus moschatus		Land					Pr		
Abrus precatorius		Land			Uc	Pr	Pr	Pr	Pr
Abutilon hirtum		Land		Pr	Pr	Pr	Uc		
Abutilon indicum		Land					Uc	Pr	Uc
Acalypha indica		Land					Pr		Uc

Species/species group	common name	habitat	date	Aruba	Bonaire	Curacao	Saba	St Eustatius	St Maarten
Acalypha wilkesiana		Land					Pr		
Adonidia merrilii		Land					Uc		
Agave karatto		Land		Pr	Pr	Pr	In	In	
Agave sisalana		Land			Uc	Pr	Uc	Pr	Uc
Ageratina adenophora		Land						Pr	
Albizia lebbeck		Land		Pr	Pr	Pr	Uc	Uc	Uc
Allamanda carthartica		Land					Pr		
Aloe vera		Land		Pr	Pr	Pr	Pr	Pr	
Alpinia purpurata		Land					Uc		
Alysicarpus vaginalis		Land				Pr	Pr	Pr	Pr
Amblovenatum opulentum		Land					Pr	Pr	
Anacardium occidentale		Land					Pr	Pr	Uc
Antigonon leptopus		Land		Pr	Pr	Pr	Pr	Pr	Pr
Aristolochia elegans		Land					Pr		Uc
Artocarpus altilis		Land					Pr	Uc	
Asparagus aethiopicus		Land					Pr		
Asparagus setaceus		Land					Pr	Pr	
Asystatia gangetica		Land					Pr	Pr	Pr
Azadirachta indica		Land		Pr	Pr	Pr	Pr	Pr	Pr
Balanites aeygyptica		Land				Pr			
Bambusa multiplex		Land					Uc		
Bambusa vulgaris		Land					Uc	Uc	

Species/species group	common name	habitat	date	Aruba	Bonaire	Curacao	Saba	St Eustatius	St Maarten
Barleria cristata		Land					Pr		
Bauhinia monandra		Land					Uc	Uc	
Bergia capensis		Land		Pr	Pr				
Bixa orellana		Land					Pr		
Boerhavia diffusa		Land		Pr	Pr	Pr	Pr	Pr	
Bothriochloa ischaemum		Land		Uc	Pr	Pr			
Bothriochloa pertusa		Land		Pr	Pr	Pr	Pr	Pr	Pr
Brassica juncea		Land					Pr	Pr	
Breynia disticha		Land					Uc		
Brugmansia x candida		Land					Uc		
Caesalpinia pulcherrima		Land					Uc	Uc	
Cajanus cajan		Land					Pr	Uc	
Callisia fragrans		Land					Uc		
Calotropis procera		Land		Pr	Pr	Pr	Pr	Pr	Pr
Canna indica		Land					Uc	Uc	
Capsicum frutescens		Land					Uc	Uc	
Cardamine flexuosa		Land					Pr		
Casuarina equisetifolia		Land					Uc	Uc	
Catharanthus roseus		Land		Pr	Pr	Pr	Pr	Pr	Pr
Cenchrus ciliaris		Land		Pr	Pr	Pr			
Cenchrus purpureus		Land					Pr		Uc
Cenchrus setaceus		Land					Pr		

Species/species group	common name	habitat	date	Aruba	Bonaire	Curacao	Saba	St Eustatius	St Maarten
Centratherum punctatum		Land					Pr	Pr	
Chamaecrista absus		Land		Pr	Pr	Pr			
Christella dentata		Land			Pr		Pr	Uc	
Citrus x aurantiifolia		Land					Uc		
Citrus x aurantium		Land					Uc	Uc	
Citrus x jambhiri		Land					Uc		
Cleome rutidosperma		Land					Pr		
Clitoria ternatea		Land		Pr	Pr	Pr	Pr	Pr	
Cnidoscolus aconitifolius		Land					Pr		
Codiaeum variegatum		Land					Pr	Uc	
Coffea arabica		Land					Uc	Uc	
Combretum indicum		Land					Uc		
Commelina benghalensis		Land		Pr	Pr	Pr	Pr	Pr	
Conyza canadensis		Land					Pr	Pr	
Cordia obliqua		Land					Uc		Uc
Cordia sebestena		Land		Pr	Pr			Uc	Uc
Coriandrum sativum		Land					Uc		
Corynandra viscosa		Land		Pr	Pr	Pr	Pr	Pr	Pr
Cosmos sulphureus		Land					Uc	Uc	
Crinum bulbispermum		Land						Uc	
Crotalaria pallida		Land						Pr	
Crotalaria retusa		Land		Pr	Pr	Pr	Pr	Pr	

Species/species group	common name	habitat	date	Aruba	Bonaire	Curacao	Saba	St Eustatius	St Maarten
Crotalaria spectabilis		Land					Pr		
Crotalaria verrucosa		Land					Pr	Pr	Uc
Cryptostegia grandiflora		Land		Pr	Pr	Pr			
Cryptostegia madagascariensis		Land					Pr	Pr	
Cucumis anguria		Land		Pr	Pr	Pr	Uc	Pr	
Cucumis dipsaceus		Land		Pr	Pr	Pr	Pr	Pr	
Cucumis melo var dudaim		Land			Pr	Pr			
Cucurbita pepo		Land					Uc		
Cyanthillium cinereum		Land		Pr	Pr	Pr	Pr	Pr	
Cynodon dactylon		Land		Pr	Pr	Pr	Pr	Pr	Pr
Cyperus croceus		Land						Pr	
Cyperus involucratus		Land					Pr		
Cyperus pelophilus		Land		Pr	Pr	Pr			
Cyperus rotundus		Land		Pr	Pr	Pr	Pr	Pr	Pr
Dactyloctenium aegyptium		Land		Pr	Pr	Pr	Pr	Pr	Pr
Delonix regia		Land					Uc	Uc	Uc
Digitaria bicornis		Land		Pr	Pr	Pr	Pr	Pr	
Digitaria longiflora		Land						Pr	
Digitaria setigera		Land					Pr	Pr	
Echinochloa colona		Land		Pr	Pr	Pr	Uc		
Eleusine indica		Land		Pr	Pr	Pr	Pr	Pr	Pr
Eleutherine bulbosa		Land					Pr	Uc	

Species/species group	**common name**	**habitat**	**date**	**Aruba**	**Bonaire**	**Curacao**	**Saba**	**St Eustatius**	**St Maarten**
Emilia sonchifolia		Land					Pr	Pr	Pr
Epipremnum aureum		Land					Pr		
Eragrostis ciliaris		Land		Pr	Pr	Pr	Pr	Pr	Pr
Eragrostis pilosa		Land		Pr	Pr	Pr			
Eragrostis tenella		Land		Pr	Pr	Pr	Pr	Pr	
Erigeron karvinskianus		Land					Pr		
Erythrina variegata		Land					Uc		
Eucharis amazonica		Land						Pr	
Eugenia uniflora		Land					Pr	Pr	
Euphorbia lactea		Land			Pr	Pr		Pr	
Euphorbia tirucalli		Land			Pr	Uc		Pr	Uc
Ficus benjamina		Land					Uc		
Foeniculum vulgare		Land					Uc		
Gliricidia sepium		Land					Pr	Pr	
Grona triflora		Land			Pr	Pr	Pr	Pr	
Gynandropsis gynandra		Land		Pr	Pr	Pr	Pr	Pr	
Haematoxylum campechianum		Land					Uc	Pr	
Halophila stipulacea		Sea		Pr	Pr	Pr	Uc	Pr	Pr
Heliconia latispatha		Land					Uc		
Heliconia psittacorum		Land					Uc		
Hibiscus rosa-sinensis		Land					Uc		
Hibiscus schizopetalus		Land					Uc		Uc

Species/species group	common name	habitat	date	Aruba	Bonaire	Curacao	Saba	St Eustatius	St Maarten
Hippeastrum puniceum		Land						Pr	Uc
Hymenocallis caribaea		Land						Pr	Uc
Impatiens balsamina		Land					Uc	Uc	
Impatiens walleriana		Land					Uc		
Indigofera tinctoria		Land			Pr	Pr		Pr	Pr
Jasminum fluminense		Land			Pr	Pr	Pr	Pr	Pr
Jatropha curcas		Land					Pr	Uc	Uc
Justicia betonica		Land					Pr		
Kalanchoe laxiflora		Land					Uc		
Kalanchoe pinnata		Land		Pr			Pr		
Kalanchoe x houghtonii		Land		Pr	Pr	Pr	Pr	Pr	
Lablab purpureus		Land					Pr	Uc	
Lantana x strigocamara		Land		Pr	Pr	Pr	Pr	Pr	
Launaea intybacea		Land		Pr	Pr	Pr	Pr	Pr	Pr
Lawsonia inermis		Land		Pr		Pr	Uc		
Leonotis nepetifolia		Land			Pr	Pr	Pr	Pr	
Leonurus japonicus		Land					Pr	Uc	
Lepidium virginicum		Land					Pr	Pr	Pr
Leucaena leucocephala		Land		Pr	Pr	Pr	Pr	Pr	Pr
Livistona chinensis		Land					Uc		
Lonicera japonica		Land					Uc		
Mangifera indica		Land					Uc	Uc	Uc

Species/species group	common name	habitat	date	Aruba	Bonaire	Curacao	Saba	St Eustatius	St Maarten
Manihot carthagenensis		Land		In	In	In		Pr	
Manihot esculenta		Land					Pr		
Maurandya scandens		Land					Uc		
Megathyrsus maximus		Land		Pr	Pr	Pr	Pr	Pr	Pr
Melia azedarach		Land					Uc	Pr	Uc
Melinis repens		Land		Pr	Pr	Pr	Pr	Pr	
Microsorum grossum		Land					Pr		
Mimosa pigra		Land					Pr		
Mirabilis jalapa		Land					Pr	Uc	
Momordica charantia		Land		Pr		Pr	Pr	Pr	Pr
Morinda citrifolia		Land			Pr	Pr	Uc	Pr	Uc
Moringa oleifera		Land			Pr	Pr	Pr	Uc	
Muntinga calabura		Land				Pr			
Murraya paniculata		Land					Uc	Uc	
Neonotonia wightii		Land						Pr	
Nephrolepis brownii		Land		Pr	Pr		Pr	Pr	
Nerium oleander		Land					Pr	Pr	
Nicotiana tabacum		Land		Pr	Pr	Pr	Pr	Pr	
Ocimum africanum		Land			Pr	Pr			
Ocimum gratissimum		Land		Pr	Pr	Pr			
Oeceoclades maculata		Land				Pr	Pr	Pr	
Oldenlandia corymbosa		Land		Pr	Pr	Pr	Pr	Pr	Pr

Species/species group	common name	habitat	date	Aruba	Bonaire	Curacao	Saba	St Eustatius	St Maarten
Opuntia ficus-indica		Land				Uc	Uc	Uc	
Parkinsonia aculeata		Land		Pr	Pr	Pr		Uc	
Passiflora edulis		Land					Uc	Uc	
Pentas lanceolata		Land					Uc		
Pereskia grandifolia		Land					Uc		
Persea americana		Land					Pr	Pr	
Phyllanthus acidus		Land					Uc	Uc	
Phyllanthus urinaria		Land					Pr		
Plantago major		Land					Pr		
Plectranthus amboinicus		Land					Pr	Pr	
Plumbago auriculata		Land					Uc		Uc
Plumbago zeylanica		Land		Pr	Pr	Pr	Pr	Pr	
Pontederia crassipes		Freshwater		Pr	Pr	Pr			Uc
Pseudogynoxys chenopodioides		Land					Pr		
Pteris vittata		Land			Pr	Pr	Pr	Pr	
Rhipidocladum racemiflorum		Land					Uc		
Rhynchosia minima		Land		Pr	Pr	Pr	Pr	Uc	Pr
Ricinus communis		Land		Pr	Pr	Pr	Pr	Pr	Pr
Rubus rosifolius		Land					Pr		
Saccharum spontaneum		Land					Uc		
Salvinia molesta		Freshwater		Pr	Pr				
Samanea saman		Land					Uc	Uc	

Species/species group	common name	habitat	date	Aruba	Bonaire	Curacao	Saba	St Eustatius	St Maarten
Sansevieria hyacinthoides		Land			Pr		Pr	Pr	
Sansevieria trifasciata		Land					Uc		
Scaevola taccada		Land		Pr	Pr	Pr		Pr	Pr
Schefflera actinophylla		Land					Uc	Uc	Uc
Schinus terebinthifolia		Land				Pr	Pr	Pr	
Sechium edule		Land					Pr		
Selenicereus grandiflorus		Land					Uc	Uc	
Selenicereus undatus		Land					Uc		
Senna alata		Land					Uc	Uc	Uc
Senna italica		Land		Pr	Pr	Pr			
Sesbania bispinosa		Land		Pr	Pr	Pr			Uc
Sonchus oleraceus		Land					Pr	Pr	
Sorghum bicolor		Land					Uc	Uc	
Sorghum halepense		Land					Uc		
Spathiphyllum wallisii		Land						Pr	
Spathodea campanulata		Land					Uc	Uc	Uc
Spathoglottis plicata		Land					Pr		
Swietenia mahagoni		Land					Pr	Uc	
Syngonium podophyllum		Land					Pr		Uc
Syzygium jambos		Land					Pr		
Tabebuia caraiba		Land				Pr			
Tabebuia heterophylla		Land			Pr	Pr	In	In	In

Species/species group	common name	habitat	date	Aruba	Bonaire	Curacao	Saba	St Eustatius	St Maarten
Tamarindus indica		Land		Pr	Pr	Pr	Pr	Pr	Pr
Tarenaya hassleriana		Land					Pr		
Tecoma stans		Land		Pr	Pr	Pr	Pr	Pr	Pr
Tecomaria capensis		Land					Pr		
Terminalia catappa		Land			Pr	Pr	Uc	Uc	Uc
Theobroma cacao		Land					Uc	Uc	
Thespesia populnea		Land		Pr	Pr	Pr	Pr	Pr	Pr
Thevetia peruviana		Land					Uc		
Thunbergia alata		Land					Pr		
Thunbergia fragrans		Land					Pr	Pr	
Thunbergia grandiflora		Land					Pr		
Thymophylla tenuiloba		Land		Pr		Pr	Pr		
Tithonia diversifolia		Land					Pr		
Tradescantia zebrina		Land					Pr		
Tragus berteronianus		Land		Pr	Pr	Pr	Pr	Pr	Pr
Tridax procumbens		Land		In	In	In	Pr	Pr	Pr
Triphasia trifolia		Land				Pr	Pr	Pr	Pr
Triumfetta rhomboidea		Land					Pr	Pr	
Turnera subulata		Land				Pr	Pr		
Urena lobata		Land					Pr		
Urochloa distachya		Land					Pr	Pr	
Urochloa mutica		Land		Pr	Pr	Pr	Uc	Pr	

Species/species group	**common name**	**habitat**	**date**	**Aruba**	**Bonaire**	**Curacao**	**Saba**	**St Eustatius**	**St Maarten**
Urochloa reptans		Land		Pr	Pr	Pr			
Vigna luteola		Land					Pr		
Washingtonia robusta		Land		Pr	Pr	Pr			Uc
Xanthosoma sagittifolium		Land					Pr	Uc	
Xanthosoma violaceum		Land					Pr		
Youngia japonica		Land					Pr		
Yucca guatemalensis		Land					Uc		
Zingiber officinale		Land					Uc		
Zinnia elegans		Land					Uc		
Ziziphus mauritiana		Land					Pr	Pr	Uc
Ziziphus spina-christi		Land		Pr	Pr	Pr			
Zoysia matrella		Land					Pr	Pr	Pr

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