Chapter **3**

STATUS OF AND TRENDS IN THE USE OF WILD SPECIES AND ITS IMPLICATIONS FOR WILD SPECIES, THE ENVIRONMENT AND PEOPLE[1]

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THIS CHAPTER SHOULD BE CITED AS:
Barron, E.S., Chaudhary, R.P., Carvalho Ribeiro, S., Gilman, E., Hess, J., Hilborn, R., Katz, E., Kigonya, R., Masski, H., Mesa Castellanos, L.I., Mograbi, P.J., Nayak, P.K., Queiroz, H., Sidorovich, A., Silvano, R.A.M., Zeng, Y, Djagoun, C, and Danner, M.C. (2022). Chapter 3: Status of and trends in the use of wild species and its implications for wild species, the environment and people. In: Thematic Assessment Report on the Sustainable Use of Wild Species of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. Fromentin, J.M., Emery, M.R., Donaldson, J., Danner, M.C., Hallosserie, A., and Kieling, D. (eds.). IPBES Secretariat, Bonn, Germany.
https://doi.org/10.5281/zenodo.6451322

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Schematic and adapted figures can be found in the following Zenodo repository:
https://doi.org/10.5281/zenodo.7009633

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STATUS OF AND TRENDS IN THE USE OF WILD SPECIES AND ITS IMPLICATIONS FOR WILD SPECIES, THE ENVIRONMENT AND PEOPLE

EXECUTIVE SUMMARY

1 **Monitoring of the ecological and social, including economic aspects of uses of wild species is critical for sustainable use *(well established)* {3.2.4, 3.3.3.3.4}. Progress towards achieving the Sustainable Development Goals and the Aichi Biodiversity Targets is assessed using global indicators, however to date, there is not a comprehensive set of global indicators able to monitor status and trends of wild species use *(well established)* {3.2.1}.** Scientific monitoring is limited or lacking for many extractive and non-extractive practices *(well established)* {3.3.1, 3.3.3, 3.3.5} and is identified as a critical knowledge gap for sustainable use {3.5}. The indicators available provide a fragmented view of wild species use in different social-ecological systems across the globe and within each practice. Global indicators on biodiversity status and trends emphasize major fisheries and terrestrial animal harvesting of large mammals, while gathering and non-extractive practices lag behind significantly in global indicator initiatives *(established but incomplete)* {3.2.1.2, 3.2.1.3, 3.2.1.5}. Monitoring is resource intensive and will require more support and investment in all countries to overcome the capacity, financial, technical and institutional challenges that generate strong limitations to monitoring of wild species, which are more pronounced in developing countries. Monitoring efforts that are inclusive of indigenous peoples and local communities, scientific approaches and equitable participation of all key actors can better inform decision-making *(well established)* {3.2.4, 3.3.3, 3.3.5}.

2 **A conservative estimate of approximately 50,000 wild species are used for food, energy, medicine, material, income generation and other purposes through fishing, gathering, logging and terrestrial animal harvesting globally *(well established)* {3.2.1, 3.3.1, 3.3.2, 3.3.3, 3.3.4}.** People all over the world directly use about 7,500 species of wild fish and aquatic invertebrates, 31,100 wild plants (7,400 of which are tree species) 1,500 species of fungi, 1,700 species of wild terrestrial invertebrates and 7,500 species of wild amphibians, reptiles, birds and mammals *(well established)* {3.2.1.3, 3.3, 3.3.2.3.4}. Among the wild species that are used, more than 20% (over 10,000 species) are used for human food, making the sustainable use of wild species critical for achieving food security and improving nutrition, in rural and urban areas worldwide *(well established)* {3.3} Knowledge and skills developed over generations make single species likely to deliver multiple uses. The contribution of wild species to livelihoods is context and situationally specific, ranging from 10% to 80% of household income globally *(well established)* {3.2.2}. An estimated 70% of the world's poor depend directly on biodiversity and businesses it fosters *(well established)* {3.2.1}. Therefore, sustainable use supports subsistence livelihoods, trade, and human well-being, including for indigenous peoples and local communities, and provides options for further economic development linked directly to successful conservation *(well established)* {3.2.1, 3.3.1, 3.3.2, 3.3.3.2.3, 3.3.3.2.4, 3.3.4.3.1, 3.3.4.3.2, 3.3.4.4.2}. While trade in local markets is important, some wild species products are part of long commodity chains and are global commodities {3.3.1, 3.3.2}. In many cases, wild species are considered superior to cultivated alternatives *(well established)* {3.3.1.5.1, 3.3.2.3.4, 3.3.3.2.3, 3.3.3.3.2, 3.3.5.2}. Fishing, terrestrial animal harvesting, logging, and nature-based tourism are vital to regional and local employment and economies in many developing and developed countries and further contribute to public infrastructure, development and provisioning of related goods and services *(well established)* {3.3}. The use of wild species also provides nonmaterial contributions by enriching people's physical and psychological experiences, including their religious and ceremonial lives *(well established)* {3.3.5.2.1}.

3 **Fisheries constitute a major source of food from wild species, with a total annual harvest of 90 million tons over recent decades of which about 60 million tons go to direct human consumption and the rest as feed for aquaculture and livestock *(well established)* {3.2.1.1}. Recent global estimates indicate that approximately 66% are fished within biological sustainable levels and 34% of marine wild fish stocks**

are overfished, but this global picture displays strong heterogeneities *(well established)* {3.2.1.1}. In countries or regions with strong fisheries management, which account for approximately half of the fisheries landings reported by the Food and Agriculture Organization of the United Nations, on average stocks are increasing in abundance and above target levels *(well established)* {3.3.1}. For countries and regions with low intensity fisheries management of large- and small-scale fisheries, the status of stocks is less well known *(well established)* {3.3.1.2}, but generally believed to be below the abundance that would maximize sustainable food production *(established but incomplete)* {3.3.1}. At the same time, small scale fisheries contribute two-thirds of the global fish catch destined for direct human consumption *(well established)* {3.3.1}. In most fisheries, there are large gaps in understanding of life histories for many marine species. For small-scale fisheries that have been assessed around the world, many have been considered to be unsustainable or only partially sustainable, especially in Africa for both inland and marine fisheries and in Asia, Latin America and Europe for coastal marine fisheries *(established but incomplete)* {3.3.1.4.1}. Small-scale fisheries are strongly anchored in local communities' ways of life on all continents and it is known that small-scale fisheries support over 90% of the 120 million people engaged in capture fisheries globally. About half of the people involved in small-scale fisheries (e.g., production, marketing) are women *(well established)* {3.4.3.1}.

4 **Unintentional bycatch fishing mortality of vulnerable, endangered, threatened and/or protected marine species, which is beginning to be assessed and managed, is unsustainable for many populations of marine turtles, sea snakes, seabirds, sharks, rays, chimaeras, marine mammals and some bony fishes *(well established)* {3.3.1.1}. Reducing unintentional bycatch and discards is progressing, but still insufficient *(well established)* {3.3.1.1}.** Some of these species may be unintentionally targeted, but are retained for food as incidental catch (including retention of shark fins and manta and devil ray gill plates and discarding of the remaining carcass), or discarded *(well established)* {3.3.1.3}. Among the 1,250 shark and ray species identified today, 1,199 have been recently assessed and 449 (37.5%) have been assessed as threatened *(well established)* {3.3.1.3}. While fishing of target species may be sustainable, the conservation status of bycatch species and other associated and dependent species is often poorly known. Bycatch is a well-known issue for several large-scale fisheries, such as the shrimp or bottom trawl fisheries, but it is also a concern for several small-scale fisheries *(well established)* {3.3.1.1, 3.3.1.5}. There have been recent advances in monitoring and managing fishing mortality of marketable incidental species and discarded bycatch species, however global uptake of effective bycatch management measures is severely lagging in a majority of marine capture fisheries *(well established)* {3.3.1.5}. For example, nearly all (99%) shark and ray species are officially declared to be taken unintentionally, but are valuable and are retained for food. Consequently, shark species have been declining steeply since the 1970s, especially in tropical and subtropical coastal shelf waters *(well established)* {3.3.1.3}.

5 **Increases in recreational fishing show it is becoming a significant component of marine capture fisheries *(well established)* {3.3.1.5.3} and a potentially significant contributor to fish declines *(established but incomplete)* {3.3.1.5.3} in combination with the commercial fleet.** There have been recent advances in monitoring and managing fishing mortality of marketable incidental species and discarded bycatch species, however global uptake of effective bycatch management measures is severely lagging in a majority of marine capture fisheries. Therefore, stock assessments which do not incorporate recreational fishing do not provide accurate assessments of global uptake and fish mortality. Recreational catch and release fishing can have negative impacts, but can be done sustainably if responsibly practiced *(well established)* {3.3.1.5.3}.

6 **Indigenous peoples and local communities contribute vital knowledge to the sustainable use of wild animals {3.3.3}, wild plants and fungi {3.3.2}, wild timber species {3.3.4.3.1} and small-scale fisheries {3.3.1.4} *(well established)*.** Subsistence uses of wild species are important sources of food, medicine, fuel and other livelihood resources for indigenous peoples and local communities in both developed and developing countries. A key to sustainable gathering, terrestrial animal harvesting, fishing, and logging practices is to work with indigenous peoples and local communities in data collection and knowledge production, which is deemed essential to evaluate and reconstruct temporal trends on resource use, establish participatory monitoring programs and develop locally based co-management systems *(well established)*. Many wild foods have nutritional benefits over processed foods and there may be no culturally acceptable alternative for ceremonial and ritual materials *(well established)* {3.3.1.7.1, 3.3.2.3.4, 3.3.3.3.3, 3.3.3.4.2, 3.3.5.2.1}. Wild species also provide a basis for culturally meaningful employment {3.3.3.2.1, 3.3.5.2.3}. In light of ongoing growth and demand for health and food security, collaboration with indigenous peoples and local communities on wild plants and fungi, genetic resources of crop wild relatives, and small-scale fisheries is an especially urgent need *(well established)* {3.3.1.4, 3.3.2.3.7}.

7 **The gathering and trade of wild fungi, plants and algae for food, medicine and ornamental use is increasing because of public demand *(well established)* {3.3.2}, and continues to be an**

economically and culturally important activity worldwide. An estimated one-fifth of the world's population participates in gathering practices, often irrespective of economic status *(established but incomplete)* {3.3.2}. People in economically disadvantaged urban and rural areas rely on wild plants, algae and fungi as a source of of essential calories, micronutrients and medicine *(well established)* {3.3.2, 3.3.2.2.2}. Gathering is often assumed to be an activity more prevalent in the Global South. However, estimates of individuals and households participating in gathering in Europe and North America range from 4% to 68%, with the highest rates of gathering by households in Eastern Europe *(established but incomplete)* {3.3.2.2.1}, often irrespective of economic status *(established but incomplete)* {3.3.2.2.3}. Nor is gathering is confined to rural areas, with dozens to hundreds of wild plant and fungi species gathered for food, medicine, firewood, decoration, and cultural practices in urban ecosystems worldwide *(well established)* {3.3.2.2.2}. Gathering is often a gendered activity in many parts of the world, with roles depending on cultural rules, on the type of harvested wild plants, algae or fungi and the places where they are harvested. In many countries, women perform the bulk of gathering and processing of wild plants for food, medicine, fuel and handicrafts for subsistence purposes and sale in local markets *(well established)* {3.3.2.2.3}.

Trade of wild plants, algae and fungi is a billion-dollar industry and establishment of supply chains can fuel economic development and diversification *(well established)* {3.3.2.1}. Trade in ornamental plants has increased rapidly over the past 40 years. Although much of the trade is in cultivated plants, poaching of ornamental species from the wild continues to occur, and can threaten the survival of species *(well established)* {3.3.2.3.2}. There is a growing demand for wild foods in the food and aromatics industries including among fine dining and *haute cuisine* establishments, and among urban populations *(well established)* {3.3.2.2.2, 3.3.2.3.4}. There is also a growing demand for products produced at least in part from harvested wild plants and fungi, for example to complement chemical medicines in many developed and developing countries *(well established)* {3.3.2.3.5}.

Unsustainable gathering is one of the main threats for several plant groups, notably cacti, cycads, and orchids *(well established)* as well as other plants and fungi harvested for medicinal purposes {3.2.2, 3.3.2.3.2}. Harvests that have been sustainable in the past due to smaller markets and sustainable harvesting practices may become unsustainable if, for example, harvesting is undertaken without following established techniques and protocols *(well established)* {3.3.2.3.4}, or new technologies are employed which increase the volume of harvest or result in damage to or death of the organism, for example when entire trees are felled rather than climbed to harvest ripe fruits *(established but incomplete)* {3.3.2}. Wild plants, algae, fungi and trees are at risk from land use change, environmental degradation, deforestation, climate change and overharvesting *(well established)* {3.3.2.3.2}, but long-term systematic research on the relative importance and interplay of these factors is lacking *(well established)* (3.6.2). Traditional management practices and cultivation / silviculture are promising approaches to increase the sustainable use of wild species *(established but incomplete)* {3.3.4.3}.

8 Terrestrial animal harvesting takes place in a variety of governance, management, ecological and socio-cultural contexts, which affect the outcomes for sustainable use. Globally, populations of many terrestrial animals are declining due to unsustainable use, but the impacts of use on wild species and society can be neutral or positive in some places *(well established)* {3.3.3}. Terrestrial animal harvesting contributes to the food security of many people living in rural and urban areas worldwide, especially in developing countries *(well established)* {3.3.3.2.3}. The most targeted species for subsistence and commercial hunting (a sub-category of terrestrial animal harvesting) are the largest-bodied (> 30 kg), as these animals provide more meat for consumption and sale and generate more economic benefits for hunters' households *(well established)* {3.3.3.2.3}. Wild meat is an important source of protein, fat and other micronutrients such as calcium, iron, zinc and fatty acids *(well established)* {3.3.3.3.3}.

Large mammals alone comprised 55-75% of total wild meat biomass extracted annually in different regions of the world, although in some traditional small band societies (e.g the San, the Hadza, the Ache, Native American groups) small game as well as wild plant resources are gathered as primary sources of protein and daily nutrition *(well established)* {3.3.3.2.3}. Estimates of wild meat consumption differ greatly – from more than 5 million tons a year globally to around 4.6 million tons in the Congo Basin and 1.3 million tons a year in the Amazon respectively. In tropical forests, exploitation of wild meat increased drastically during recent decades due to large numbers of urban consumers, individual food preferences, change in hunting technologies, and scarcity of alternative sources of protein *(established but incomplete)* {3.2.1, 3.3.3.2.3}. Sustainability of hunting for food, especially in tropical areas, has been negatively affected by profound socio-economic changes, which have resulted in shifts from local-level subsistence towards more intensive wild meat trade *(well established)* {3.3.3.2.3}. The sustainability of wild meat hunting is increasingly driven by socio-economic changes, recreation, entertainment, trade, or trafficking, rather than solely hunting for subsistence *(well established)* {3.3.3}.

9 Many game species with high intrinsic rates of population increase or high ecological adaptability have been used sustainably and tolerate even high

utilization levels *(well established)* {3.3.3.2.4}. The impacts of hunting on the abundance of wild species vary worldwide depending on the biological characteristics of the animals as well as the management systems but are generally lower for species with high population growth rates, or high ecological adaptability, and where hunting is well managed *(well established)* {3.3.3.2.4}. Research suggests that hunting can support sustainable use because it increases the economic value of wild species and the habitats they depend on for local people and communities, providing critical benefit flows that can motivate and enable sustainable management approaches *(established but incomplete)* {3.3.3.3.4}.

Hunting can also create major costs for biodiversity, ecosystem functioning, and animal welfare *(well established)* {3.3.3.2.4}. Selective hunting of particular species or of individuals or of populations which have particular attributes (e.g., large-sized or large horns) can impact ecosystem structure and processes through modifying vegetation composition and structure, including forest succession and regeneration patterns, shifts in ecosystem functions, such as nutrient cycling and carbon capture, declines in carnivore densities, changes of the genetic structure of affected populations {3.3.3.2.4}, changes in predator-prey relationships and shifts in distribution of species and biomass across multiple trophic levels *(well established)* {3.3.3.4, 3.3.3.2.4}. Unsustainable hunting has been identified as a threat for 1,341 wild mammal species, including 669 species that were assessed as threatened, and declines in largebodied species with low intrinsic rates of population increase have been linked to hunting pressure *(well established)* {3.3.3}. Negative impacts of hunting have also been reported on bird species *(well established)* {3.3.3.2.5, 3.3.3.2.6, 3.3.3.3.4}.A long-term holistic approach with consideration of all ecological, evolutionary, economic, and social consequences is required to fully evaluate hunting wild species as a conservation tool and provide appropriate management policies {3.3.3.3.4}.

10 Recreational hunting is highly controversial and has been written about extensively in the scientific literature, however only a limited number of these studies contain well-argued, data-driven evidence and even fewer address recreational hunting with regards to sustainable use and its trade-offs *(well established)* {3.3.3.2.4}. There is considerable variation in the way recreational hunting is governed and administered in different regions, which makes any generalization about its sustainability or unsustainability difficult {3.3.3.2.4}. Some species are recovering from small population sizes under management systems that allow regulated recreational hunting, usually as a way to generate revenue and increase the land area for population expansion *(established but incomplete)* {3.3.3.2.4}. Sustainable use needs to consider the social (including institutional and economic) and ecological factors and is therefore highly context specific. Operationally, sound biological management is contingent on appropriate institutional, social and economic conditions, which include proper regulation of the hunting system by scientific and/or local control and knowledge. Weak tenure, the centralization of revenues derived from hunting and breakdown of community governance without any effective replacement by state officials can result in unsustainable recreational hunting *(well established)* {3.3.3.2.4}.

Large areas of land that are managed for recreational hunting (e.g., ~1.4 million km² in Africa) could contribute to conservation objectives and spatial conservation targets, but their unique biodiversity values as well as their ecological and social durability have mostly not been evaluated *(established but incomplete)* {3.3.3.2.4}. Economically, recreational hunting has been considered an important activity and is credited with generating revenues and creating jobs, as well as providing income and other important economic and social benefits to indigenous and local people in rural, remote and/or otherwise marginal areas. Some recreational hunting activities can generate hundreds to hundreds of thousands of United States dollars, and globally create a substantial revenue flow from developed to developing countries, as well as from urban to rural areas within countries *(well established)* {3.3.3.2.4}.

11 Logging for energy is prevalent globally but reliance on wood for heating and cooking is highest in developing countries *(well established)* {3.3.4}. Logging is also an important source of subsistence resources and income for millions of people worldwide *(well established)* {3.3.4.3}. Logging for energy accounts for 50% of all wood consumed globally, and accounts for 90% of timber harvested in Africa. Fuel wood use is declining in most regions but is increasing in sub-Saharan Africa *(established but incomplete)* {3.3.4.4.2}. Fuel wood demand can be met at global and national scales when comparing supply-demand balances, but localized fuel wood shortages and the associated forest and woodland degradation occur in areas where people have few alternatives for cooking and heating *(established but incomplete)* {3.3.4.4.2}.

Worldwide, 2.4 billion people rely fuel wood for cooking and an estimated 880 million people globally log firewood or produce charcoal, particularly in developing countries *(established but incomplete)* {3.3.4.4.2}. Sustainable fuel wood logging remains a renewable energy opportunity that provides income, heating and cooking in developing countries where 1.1 billion people do not have access to electricity or alternative energy sources *(established but incomplete)* {3.3.4.4.2}, provided air pollution (indoor and outdoor) and climate change emissions are mitigated.

Logging is carried out by smallholders, communities and industrial entities *(established but incomplete)* {3.3.4.3}. For example, logging by smallholders provides thousands of jobs in Central African countries *(well established)* {3.3.4.3.1}. An estimated 15% of global forests are managed as community resources by indigenous peoples and local communities, often with a strong focus on multiple use management *(established but incomplete)* {3.3.4.3.2}, while industrial logging occurs in over one quarter of the world's forests *(well established)* {3.3.4.3.3}.

12 **Wild tree species are currently the major sources for wood and wood products and will continue to be in the coming decades *(well established)* {3.3.4.1}. Globally, wild tree species provide two thirds of industrial roundwood {3.3.4.3.3}. However, destructive logging practices and illegal logging threaten sustainable use of natural forests *(established but incomplete)* {3.3.4} and an estimated 12% of wild tree species are threatened by unsustainable logging {3.2.1.4}.** The outcomes of logging affect forest ecology, as well as other forest-based uses of wild species, such as gathering, terrestrial animal harvesting and observing wild species *(well established)* {3.3.4}.Although there is an expected increase in production of plantation wood, there is also a projected increase in timber demand, which will not be matched by plantation wood *(well established)* {3.3.4.1, 3.3.4.2}. Inventory-based management plans and selective logging could reduce the impacts of logging, but its sustainability depends on the planning, techniques and implementation used to minimize damage to the residual forest stand, as well as forest soils, flora and fauna *(well established)* {3.3.4.2}. About 20% of the world's tropical forests (3.9 million km^2) are currently subject to selective logging *(well established)* {3.2.1.4, 3.3.4.2}.

13 **A geographic shift is observed in illegal logging and related timber trade *(established but incomplete)* (3.3.4.2).** Illegal logging has declined in parts of the tropical Americas, as well as parts of the tropical and mountain regions of Asia due to improved monitoring and collaborative transboundary collaborations. However, illegal logging and trade has increased in other regions, including Southeast Asia, Northeast Asia and parts of Africa *(established but incomplete)* {3.3.4.2, 3.3.4.3.1}.

14 **Non-extractive practices using wild species occur widely in all areas by all cultures, although the nature of the practice differs across cultures and locations *(well established)* {3.3.5}. The benefits of wild species for improving human mental and physical health have been widely documented in both urban and natural settings *(well established)* {3.3.5.2.2}.** Non-extractive practices are core to human identity, support mental and physical well-being, raises awareness and facilitate connection to nature and society *(well established)* {3.3.5.2.2}. Despite the crucial importance of non-extractive practices for human-nature connections, with the exception of recreational tourism, there is extremely limited knowledge on the use, trends or sustainability of these practices *(well established)* {3.3.5, 3.5}.

15 **Nature-based tourism is the most prominent non-extractive practice and demand for wild species media (i.e., documentaries) and *in situ* observing (e.g., wildlife watching tourism) was growing steadily until 2020 and the global COVID-19 pandemic *(well established)* {3.3.5.2.3}.** Wildlife watching generates substantial revenue, contributing US$120 billion in 2018 to global gross domestic product (five times the estimated value of the illegal wild species trade) and sustaining 21.8 million jobs {3.3.4.2.3}. Prior to the COVID-19 pandemic, protected areas, globally, received approximately 8 billion visitors per year, generating 600 billion United States dollars per year, with wild-species rich countries experiencing bigger increases in tourism visitation *(well established)* {3.3.5.2.3}.

Wildlife watching is crucial for local livelihoods, provides employment and promotes development of tourism-related infrastructure, particularly in some remote locations *(well established)* {3.3.5.2.3, 3.4.4.2}. These and additional benefits make positive contributions to conservation, community development, and livelihoods in underdeveloped and remote regions when well-managed, but may also create vulnerability to shocks such as global recessions or pandemics *(well established)* {3.3.5.2.3, 3.3.5.3}. Although non-extractive practices are frequently less directly harmful to wild species and ecosystems than extractive ones, wildlife watching may have unintended detrimental impacts through changes to species behavior, physiology, or damage to habitats *(well established)* {3.3.5.2.3}. Many of the unsustainable impacts of the tourism industry could be mitigated through context-based understanding, implementation of best practice guidelines for observing, communication, education and public awareness of tourists and tour operators, collaborative engagement with all stakeholders and sector-specific regulation *(well established)* {3.3.5.2.3, 3.3.5.2.4}.

16 **Effective management systems that promote the sustainable use of wild species can contribute to broader conservation objectives *(established but incomplete)* {3.3.3.3.4, 3.3.3.4.1, 3.3.4.3.2, 3.3.5.2.3}.** Based on assessment of 10,098 species from 10 taxonomic groups documented for the International Union for Conservation of Nature Red List of Threatened Species, at least 34% of the wild species assessed are used sustainably *(established but incomplete)* {3.2.1, 3.2.2, 4.2.4.3.1}. This includes 172 threatened or near-threatened species. Overall, unsustainable harvest contributes towards elevated extinction risk for 28-29% of near-threatened and

threatened species from 10 taxonomic groups assessed on the International Union for Conservation of Nature Red List of Threatened Species {3.2.1, 3.2.2}.

17 **Trade-offs and synergies among fishing, gathering, terrestrial animal harvesting, logging, and non-extractive practices are inherently linked but often treated exclusively or in isolation from each other *(well established)* {3.4}.** This exclusivity is reflected in the dominant approach of practice specific policies, which leads to significant compartmentalization of rules and regulations. The bifurcation of existing uses alongside the emergence of new uses within a practice area must also be considered; for example, the positioning of capture fisheries vs. aquaculture within fishing practices; or ceremony and cultural expression vs. recreation and nature-based tourism within gathering practices. Considering these uses exclusively has led to an intense reconfiguration of intra-practice trade-offs and synergies with similar effects *(well established)* {3.4.5}. Intensification of existing uses and/or emergence of new uses for wild species have often led to rapid and substantial reconfiguration of trade-offs and synergies within and among practices with negative impacts on sustainable use *(well established)* {3.4}.

3.1 INTRODUCTION

People directly benefit from nature by interacting with and using wild species (see 1.3.2) for definition of wild species), which provide provisioning and material contributions, and cultural and spiritual uses for human well-being (Millennium Ecosystem Assessment, 2005). Furthermore, as discussed in greater depth in Chapter 1, the ability to use wild species is crucial for social and economic justice, and to maintain the livelihoods, well-being and cultural diversity of indigenous peoples and local communities. The use of wild species involves three interconnected dimensions: (i) the wild species itself, (ii) the practices undertaken by people to obtain parts of or the whole organism, and (iii) the uses (both extractive and non-extractive) of the organism (Figure 1.1). Identifying and documenting the status of these dimensions, and their interactions and trends, is the subject of this chapter.

It is important to note that the scoping document for this assessment calls for "an understanding of sustainable use of wild species that are important elements in the present and future functioning of ecosystem and their contributions to people," (p.3 of the sustainbale use of wild species scoping document). Thus, the systematic literature reviews on which much of the current chapter are based specifically focused on those uses considered to be sustainable, rather than reporting on all uses and determining their sustainability. This has clear implications for the status and trends reported in the following pages in terms of which literatures were reviewed and how status and trends are reported.

The scale and scope of the overall use of wild species is needed in order to understand the status and trends of specific uses at a finer scale. This overview is provided in section 3.2, based on an analysis of a subset of global indicators previously used by IPBES and from the Biodiversity Indicators Partnership. Subsistence use includes the use of wild species by individuals or their direct social network, for nutritional, cultural, spiritual and social survival (Emery & Pierce, 2005). Wild species use also includes trade in informal and formal markets. Informal trade is defined as unrecorded trade which may be paid for in currency or in goods and services. Formal trade refers to recorded transactions in legal and illegal markets. These aspects are considered part of sustainable use. This section also provides a global level overview of human-used wild species distributions, practices and purposes.

As discussed in detail in Chapter 1, the IPBES conceptual framework recognizes different types of evidence, including but not limited to scientific knowledge. It aims to include different worldviews and associated knowledge systems equally, as much as possible, in the assessment. Therefore, throughout the chapter every effort was made to augment the systematic review of the scientific literature with knowledge from additional sources. This included drawing from experts' own experiences working with indigenous people and local communities, attending the indigenous people and local communities Workshops organized as part of this assessment, and drawing from non-scientific reputable sources when appropriate.

Reports on the status and trends separated out by practice and uses (as defined in chapter 1) are provided in section 3.3 ((fishing (3.3.1), gathering (3.3.2), terrestrial animal harvesting (3.3.3), logging (3.3.4), and non-extractive practices (3.3.5)). These analyses were conducted following a common standard, but somewhat independently in order to be consistent with the standard approach in the relevant scientific and policy literature. Throughout section 3.3 all authors made every effort to draw from multiple knowledge systems in tandem. Within each sub-section the information is organized, as much as possible, according to the relevant uses: ceremony and cultural expression, decorative and aesthetic, energy, food and feed, medicine and hygiene, recreation, science and education, and materials and shelter. In order to save space, only those uses relevant for the practice, and being undertaken at significant enough levels as to be appropriate for inclusion in a global assessment, are included in the various sub-sections. Within the fishing and terrestrial animal harvesting sections there are separate sections for non-lethal uses. Each practice sub-section concludes with a brief review of emerging issues to highlight complex and novel topics. These vary by practice but all sections include information on the emerging effects of the COVID-19 pandemic.

Nature's contributions to people are discussed throughout the chapter, as it was felt most effective to include the information in the relevant sections rather than sequestered into its own section. Throughout section 3.3 it becomes abundantly clear that the ability to sustainably use wild species is important for people all over the world, in all countries where people eat meat or fish, eat berries or wild vegetables, use paper, and enjoy nature; and it is absolutely critical to indigenous peoples and local communities worldwide who fundamentally rely on wild species for their own subsistence and livelihoods in terms of food and medicinal provisioning, informal and formal trade, and often also cultural and spiritual practices. Furthermore, several sections in 3.3 point out that certain kinds of uses may create new opportunities for upward social and economic mobility for some but simultaneously exclude others, resulting in differential qualities of life and well-being for groups of people, often exaggerating existing inequalities.

A growing trend in the scientific literature is increasing awareness of the trade-offs and synergies among the practices and uses, which is addressed in section 3.4. This includes a discussion of multifunctionality in different sectors. Trade-offs and synergies reflect a host of interactions, connections, relationships and linkages within, between and among practices and uses. This being the case, achieving and maintaining the goal of sustainable use of wild species hinges on the level of understanding of the key trade-offs and possible areas of synergy within and across practice areas. A simple three-pronged approach is used to consider the various trade-offs and synergies by focusing on (i) Trade-offs and synergies at intra-practice and intra-use level; (ii) Trade-offs and synergies between practices and uses; and (iii) Trade-offs and synergies involving the social, economic and environmental aspects of sustainable use.

3.2 SCALE AND SCOPE: A GLOBAL OVERVIEW

Use of wild species varies across space and over time. While use of wild species is often addressed based on local case studies, a global overview on status and trends of wild species use is lacking. In order to provide this global overview, a search was conducted across different global organization websites to select available datasets and global estimates on wild species use (3.2.1) (see the data management report for Chapter 3 systematic literature review at https://doi.org/10.5281/zenodo.6452651). Indicators for sustainable use of wild species were selected based on criteria in the scoping document and Chapter 1; this process is outlined in section 3.2.2. From a high diversity of indicators available (see chapter 2), this section focuses on (i) indicators selected by IPBES experts in the context of the global and regional assessments, (ii) Sustainable Development Goals indicators by the United Nations and (iii) the Aichi Biodiversity Target indicators by the Convention on Biological Diversity, particularly focusing on indicators within the theme "sustainable use" and "species". In addition to searching data and indicators across different institution websites, a literature search was completed in "Google" and "Science Direct" using the following keywords: "wild species" AND "use" AND "indicators" OR "indices" OR "indexes" (accessed in June 2020). Data sources and indicators suggested by reviewers during internal and external reviews were also reviewed and considered. Following the section indicators which focuses on spatial scales and distribution, we include a special section on the importance of the temporal scale in relation to use of wild species (3.2.3). Finally, this we explore the relative importance of different contexts in which wild species are used both for subsistence and trade (3.2.4).

3.2.1 Datasets available and global estimates of wild species used

Estimates on the number of wild species used by humans across the different regions of the globe are scarce and scattered amongst different datasets and organizations. The review of datasets presented below show that there is an uneven distribution of data available across the world documenting the number of wild species and their direct uses by humans. Most of the global datasets reviewed predominantly register and document use of wild species in the Northern Hemisphere (**Figures 3.1** to **3.3**). However, evidence suggests that the greater part of global biodiversity occurs in the tropical and subtropical regions, and in many of these countries local communities depend heavily on direct use of natural resources.

Uses are dynamic and change over time. Traditional knowledge and skills as well as science and technology continue to develop novel techniques and adapt to changing uses (Kersey *et al.*, 2020). The evolving relationships between wild species use and associated knowledge/skills, together with the development of science and technology, drives the creation of novel economies surrounding to and associated with the use of wild species. Unfortunately, the review shows that although traditional and scientific knowledge often highlight that one wild species can have many uses (e.g., food, raw material, cultural expression, etc.), and provide a range of nature's contributions to people (NCP) benefits, the datasets reviewed generally focus on a single use category for a single species.

Table 3.1 summarizes key estimates in order to provide an overview of the total number of wild species and their uses across different taxa and practices of use. About 50,000 wild species are used for food, energy, medicine, material and other purposes through fishing, gathering, logging and terrestrial animal harvesting globally. People all over the world directly use about 7,500 species of wild fish and aquatic invertebrates, 31,100 wild plants, of which 7,400 ON 5 species are trees, 1,500 species of fungi, 1,700 species of wild terrestrial invertebrates and 7,500 species of wild amphibians, reptiles, birds and mammals. Among the wild species that are used, more than 20% (over 10,000 species) are used for human food. The practices are further analyzed in the following sections (3.3.1 to 3.3.5).

The International Union for Conservation of Nature (IUCN) Red List of Threatened Species (https://www.iucnredlist.org/) is one of the most widely used datasets to determine status and trends of wild species use. The list includes assessments of 128,918 species of vertebrates, invertebrates, wild plants, fungi and protists; its major focus is to report their threat categories. In the November (IUCN, 2020b:4) update of the list, the total number of species assessed was: animal: 78,126, wild plants: 50,369, fungi: 408 and Chromista: 15.

The use of wild species is captured by the International Union for Conservation of Nature Red List in two ways: as a threat (under the threats classification scheme) and as a form of use or trade (under the use and trade classification scheme). While the coding of major threats is required (except for species of least concern), the coding of use and trade is only recommended, and is therefore less consistently coded across listed species, including the comprehensively assessed groups. To qualify as a comprehensively assessed group, the taxonomic group must include at least 150 species, of which more than 80% have been assessed (Marsh *et al.*, 2021). The 2020 July (IUCN, 2020b:3) report shows that around 35,765 species (28%) are considered threatened to minor or major degrees. Of these, 20,935 species of animals (26.8% of the total assessed animals), 13,142 species of wild plants (26.1%) and 162 species of fungi (39.7%) were reported as threatened.

Table 3 1 **Number of species and their uses by practice.**

The table shows estimates from different sources. Only estimates corroborated by two or more sources are included. Sources: Flora of China (FOC), The Plant List (TPL), World Flora Online (World Flora Online), State of the World's Plants 2016 (SOTWP-2016), State of the World's Plants and Fungi 2020 (SOTWPF-2020), State of the World's Fungi 2018 (Willis, 2018), Food and Agriculture Organization (FAO), Butchart, (2008), Global Tree Assessment (BGCI, 2021; Global Tree Assessment, 2020, 2021), (Balmford *et al.*, 2015; WTTC, 2019a).

Number of wild species & uses/ practices	Fishing (section 3.3.1)	Gathering (section 3.3.2)	Terrestrial animal harvesting (section 3.3.3)	Logging (section 3.3.4)	Non extractive (section 3.3.5)
Estimates of number of species used Note that estimates range widely depending on the source	• Approximately 7,500 wild fish species (Chordata) used and traded (Fukushima, Mammola, & Cardoso, 2020) • 30% of crustacea species and 38% of Mollusca species are used by humans (FAO, 2020d) • 100% of cone snail species are used by humans (Marsh *et al.*, 2021)	Approximately 31,100 wild plant and 1,500 wild fungi species have documented uses (SOTWP, 2016; TPL, 2020; WFO, 2020)	• Approximately 5,600 terrestrial bird, mammal, amphibian, and squamate reptile species are used and traded globally (Scheffers, Oliveira, Lamb, & Edwards, 2019) • Approximately 2,000 species of invertebrates, amphibians, fish, reptiles, birds and mammals are used as wild meat across the world (Coad *et al.*, 2019) • 4,561 birds are used for food or as pets (Butchart, 2008) • Over 300 mammal species are used for hunting (William J. Ripple *et al.*, 2016), • Vertebrates (Chordata) are the most traded organisms (Fukushima *et al.*, 2020) • ~11% amphibians are used by people (Marsh *et al.*, 2021)	• Logging is reported to be a threat to approximately 7,400 tree species (27%) (Global Tree Assessment, 2021; IUCN, 2020b:3) • Approximately 34,000 tree species are used on a regular basis but not only for logging, (State of the World's Forest Genetic Resources (FAO, 2014c) • One in five tree species are recorded as having a specified human use and many have a variety of different uses (Global Tree Assessment, 2020) • ~6,000 tree species (10%) have medicinal or aromatic use (Global Tree Assessment, 2020) • ~3,716 tree species have timber use (IUCN, 2020) • ~2,500 wild species are documented sources of fuel or bioenergy (SOTWP, 2020) • most common uses for trees as recorded by the International Union for Conservation of Nature Red list (IUCN, 2020b:3): constructuction: 3,716 wild species, medicine:1,951 wild species, horticulture: 1646 wild species, fuels: 1444 wild species, human food 1,382 wild species, household goods:1,302 wild species	Non-extractive uses tend to be based in the whole ecosystem instead of species. For example, worship in sacred groves includes all the species in the grove and its vicinity. Recreational tourism may focus on charismatic species or taxonomic group (e.g. butterfly-watching) but encompasses the whole park/ coral reef experience. Forest therapy uses the whole forest, not single species.

Number of wild species & uses/ practices	Fishing (section 3.3.1)	Gathering (section 3.3.2)	Terrestrial animal harvesting (section 3.3.3)	Logging (section 3.3.4)	Non extractive (section 3.3.5)
Uses (average annual consumption OR trade volume)	• Average consumption of 90 million tons/year (FAO, 2020d) • Food fish consumption grew form 9.0 kg per capita in 1961 to 20.2 kg per capita in 2015 at an average rate of about 1.5 percent per year	• There are between 3.5 and 5.76 billion users of Alagae, fungi and plants globally (Charlie M. Shackleton & de Vos, 2022) • Sales of BioTrade beneficiary companies reached €4.3 billion (2015). • International trade volume for wild edible fungi was estimated at 1.23 million tons in 2017 (de Frutos, 2020) • Around 5 million people worldwide from collectors/fishers/ hunters, workers, among others are involved in BioTrade (UNCTAD, 2017) • 70% of the world's poor depend directly on biodiversity (UNCTAD, 2017) • The number of companies that report on biodiversity in their annual reporting is growing. 36 of the top 100 cosmetic companies and 60 of the top 100 food companies now mention biodiversity. • Medicinal wild plants: 60–90% of medicinal and aromatic plants in trade are wild collected • 14–15 billion United States dollars estimated value of trade in essential oils by 2025 (TRAFIC, 2018) • Global value of wild algae, fungi, plants and animal origin was estimated by the Food and Agriculture Organization of the United Nations as 20.6 billion United States dollars in 2010 (TRAFIC, 2018) • Global value of organic wild collected products to be between EUR 630 to 830 million (base year 2005 (IFOAM/ITC, 2007)	• Very different estimates of use, ranging from 5 million tons/year globally to 4.6 million tons/year in Congo Basin alone) • Central Africa: 1.6 to 11.8 million tons/year Brazilian Amazon: 0.07 to 1.3 million tons/year (Coad *et al.*, 2019)	• Timber trade as a whole (including wild) at over 200 billion United States dollars (TRAFIC, 2018) • 880 million people spending time collecting firewood or producing charcoal • ~1.2% global workforce is engaged in commercial fuel wood activities to supply urban centers (FAO & UNEP, 2020) • Global fuel wood production revenue in 2011: 33 billion United States dollars (FAO & UNEP, 2020) • 2.4 billion people use fuel wood for cooking (FAO & UNEP, 2020) • Over 1,500 tree species are traded internationally (BGCI, 2021) and ~2,400 tree species are actively managed for their products and/or services (State of the World's Forest Genetic Resources (FAO, 2014c) • Selective logging represents 15% of the global timber supply (Poudyal, Maraseni, & Cockfield, 2018) • Over 400 million ha, about 10% of global forests, are subject to selective logging practices (Poudyal, Maraseni, & Cockfield, 2018)	• 120.1 billion United States dollars in gross domestic product to the global economy (2018), including multiplier effects: 343.6 billion United States dollars sustaining 21.8 million jobs (WTTC, 2019a) • 8 billion visits to protected areas annually 600 billion United States dollars per year. (Balmford *et al.*, 2015)

Using Red List data, Marsh *et al.* (2021) analyzed species-level data for 30,923 species from 13 taxonomic groups which have been comprehensively assessed. Results of this study demonstrate widespread use across taxa, of approximately 40% of species (10,098 of 25,009 from 10 taxonomic groups with adequate data). This estimate is an important reminder of the relevance of the current assessment. According to this data source, wild plant groups tend to be used for more purposes than animal groups, including for food and animal feed, medicinal use, household goods and handicrafts /jewelry, fuels and chemicals. For aquatic animals, the top uses were human food (bony fishes and crustaceans), specimen harvest (cone snails), and pets and display animals (corals and bony fishes). Additional uses included handicrafts and jewelry (cone snails and corals) and medicine (cone snails). It should be noted that the majority of the taxa have multiple uses.

McRae *et al.* (2022) using the Living Planet index data (https://livingplanetindex.org/) to show the locations of populations which were coded as utilized (black diamonds) and non-utilized (white diamonds) across the globe for practices such as fishing, gathering and hunting (non-extractive uses were not included) **(Figure 3.1)**. Threat information from the International Union for Conservation of Nature Red List was available for 3,195 populations analyzed (1,694 utilized and 1,501 not utilized) (McRae *et al.*, 2022). Nearly three quarters of overexploitation threats of coded utilized populations were as result of practices such as hunting and gathering. The results show that the global trends for those wild populations analyzed were negative, and in populations where there is no management, decline was more rapid, especially in Africa and the Americas (section 3.2.1.2).

In reports from 91 countries submitted to the Food and Agriculture Organization of the United Nations (FAO) reports on the use of wild species (wild plant, animals and fungi) for food across different taxa, over 60% of responses in Africa, Asia, Latin America and the Caribbean and Near East and North Africa Caribbean (FAO, 2019a) refer to use of wild plants as food sources. The use of wild fish as food is most reported in North America and the Pacific, while the use of wild birds as food sources is most reported in Latin America and the Caribbean **(Figure 3.2)**.

In summary, no comprehensive global dataset was found for reporting status and trends of wild species that are directly used by humans. Furthermore, aggregating

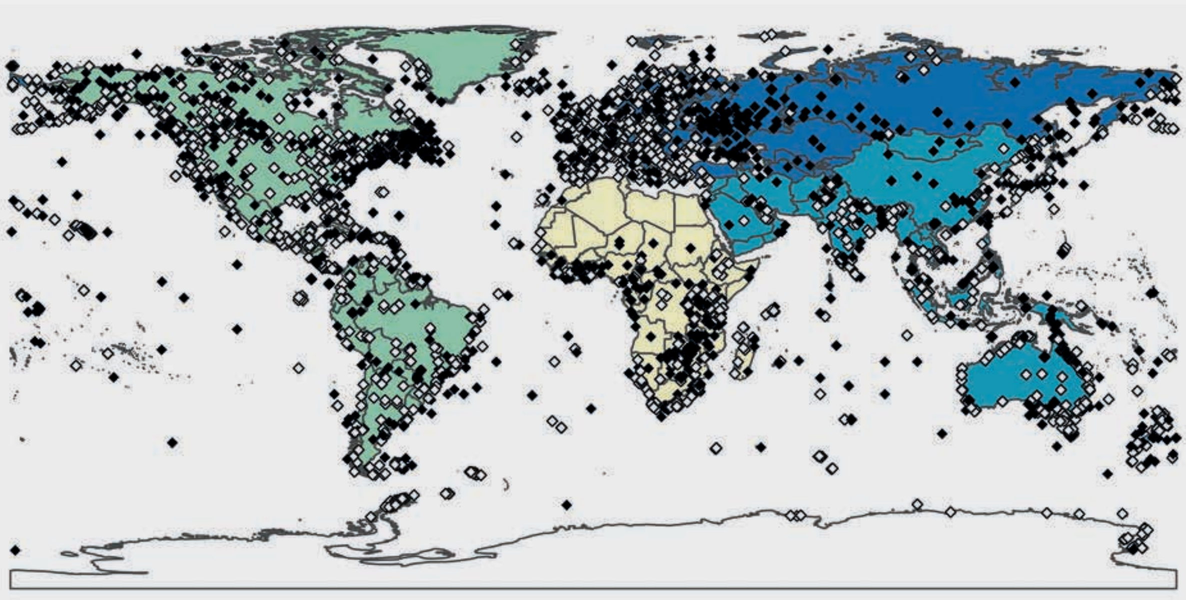


Figure 3 1 **Locations of utilized (black diamonds) and non-utilized populations (white diamonds).**

This map is directly copied from its original source (McRae et al., 2022) and was not modified by the assessment authors. The map is copyrighted under license CC BY-NC-ND 4.0. The designations employed and the presentation of material on the maps used in the assessment do not imply the expression of any opinion whatsoever on the part of IPBES concerning the legal status of any country, territory, city or area or of its authorities, or concerning the delimitation of its frontiers or boundaries. These maps have been prepared or used for the sole purpose of facilitating the assessment of the broad biogeographical areas represented therein and for purposes of representing scientific data spatially.

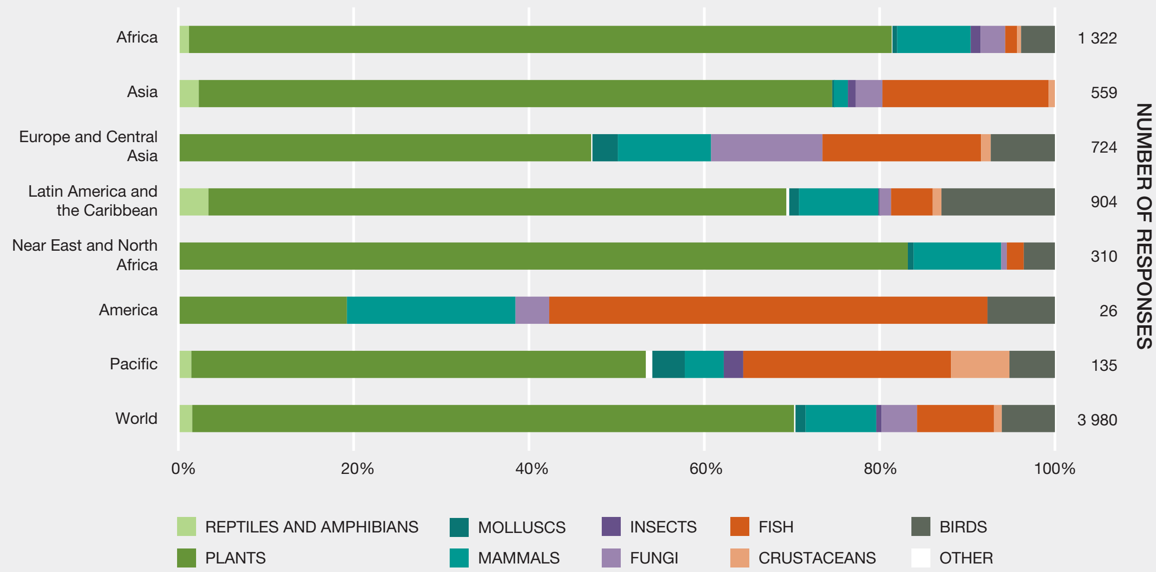


Figure 3 2 **Response on use of wild species for food reported by type and region.**

A "response" is the report of a given wild food species by a given country. Analysis based on 91 country reports. Source: (FAO, 2019a) under license CC BY-NC-SA 3.0.

estimates of species used across taxa would only serve to aggregate the levels of uncertainty within each data set. Therefore, information on use is presented by practice type. The following section details, in the context of these data constraints, the indicators available for assessing the sustainability of use of wild species across different practices. The practices are further analyzed in section 3.3.

3.2.1.1 Fishing

Fish are a valued food source throughout the world contributing both culturally and economically to food security, especially in coastal areas **(Figure 3.3)**. Capture fisheries constitute the largest wild food consumed by humans, with estimates from the FAO of a total capture fisheries harvest of 90 million metric tons per year over recent decades, of which about 60 million metric tons goes to direct human consumption and most of the rest as feed for aquaculture and livestock (FAO, 2020d).

The most widely used data on global fisheries is on fisheries landings from 1950 to present maintained by the FAO. Reporting includes landings by country, region, and taxonomic group (http://www.fao.org/fishery/statistics/global-capture-production/3/en). These data are widely accepted and used, while it is recognized that the landings of small-scale fisheries are almost certainly underestimated.

The FAO also presents a bi-annual report estimating what fraction of these fish stocks are underfished, sustainably fished, and overexploited. As of 2017, 34.2% of global fish stocks were overfished, 59.6% were fished in accordance with maximum sustainable yield guidelines and 6.2% were underfished (FAO, 2020d). The share of fish stocks within biologically sustainable levels (maximally sustainably fished or underfished) declined from 90% in 1974 to 65.8% in 2015 (FAO, 2020d). **Figure 3.3** below shows FAO estimates of capture production and aquaculture. These data are reported by individual countries and include only estimates of landing so do not include non-retained catch that are discarded at sea. Landings estimates for small scale fisheries are widely regarded to be significantly underestimated.

So far, there is no available global estimate of total number of wild fish species (marine and freshwater) used or how this varies across the globe (list of species across regions are incomplete to give an estimate). There are, however, reports that ~7,500 Chordata species traded globally are fish (Fukushima *et al.*, 2020). A wide range of countries and regional fisheries management organizations report the status and trends of individual fish stocks, and the University of Washington maintains a database (www.ramlegacy.org) of these giving abundance trends and status for about 1,200 individual marine species stocks constituting roughly half of global fish landings **(Figure 3.4)**. While there is data for IPBES regions such as Europe (e.g., https://www.eumofa.eu/) and North America, data for other IPBES regions are missing.

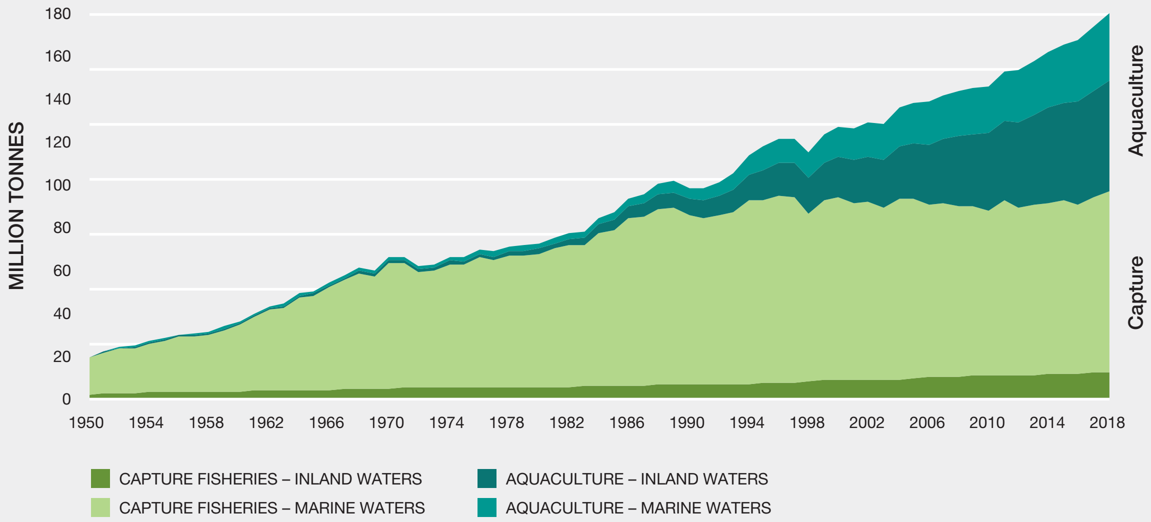


Figure 3 3 **Global trends in world capture fisheries and aquaculture production (excluding aquatic mammals, crocodiles, alligators and caimans, seaweeds and other aquatic plants).**

Source: (FAO, 2020d) under license CC BY-NC-SA 3.0 IGO.

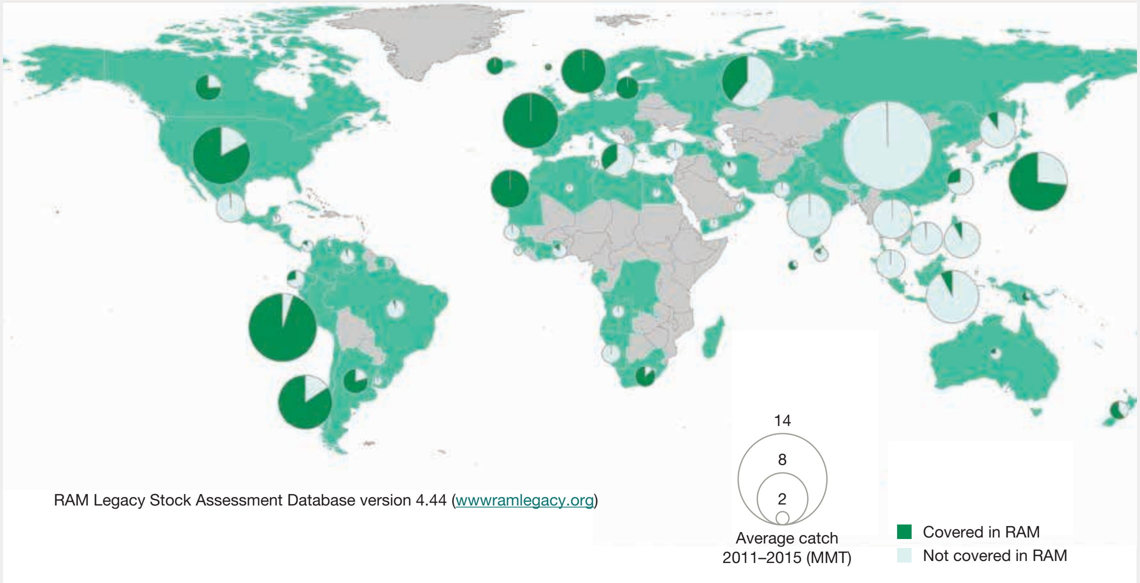


Figure 3 4 **Map showing the amount of total marine fish landings (MMT: millions of metric tons) in a country or region covered by stocks in the RAM Legacy Database.**

The area of circles is proportional to the total landings from the country or region, and the dark green portion represents the fraction of landings from stocks in the RAM Legacy Database. Green-shading of countries (or regions) on the map is applied for the top 50 countries (or regions) in terms of landings in the Food and Agriculture Organization of the United Nations' Capture Fisheries Landings Database. *This map is directly copied from its original source (RAM Legacy Stock Assessment Database, 2018) and was not modified by the assessment authors. The map is copyrighted under license CC-BY 4.0. The designations employed and the presentation of material on the maps used in the assessment do not imply the expression of any opinion whatsoever on the part of IPBES concerning the legal status of any country, territory, city or area or of its authorities, or concerning the delimitation of its frontiers or boundaries. These maps have been prepared or used for the sole purpose of facilitating the assessment of the broad biogeographical areas represented therein and for purposes of representing scientific data spatially.*

3.2.1.2 Gathering

Both the State of the World's Plants (KEW, 2020) and Flora of China (FOC, 2020) estimate that there are approximately 31,100 wild plants with documented use. Although both datasets agree on the number of plant species used, they differ on the typologies of use. Kew royal botanical garden (2020) defines "useful plants" as plant species that have been described as fulfilling a particular need for humans, animals or the environment. This definition of use differs from that produced as parts of this assessment (see Chapter 1), however since it is the definition tied to the Kew data it is used for the remainder of this discussion. According to Kew (2020), the total number of wild plant species used for human food is of 5,538, and 3,649 species are used as animal food. Medicinal use is made of 21,695 plants for medicines (20,313) and social uses (1,321). Wild plants are also used as sources of fuel (1,621) and raw materials (11,365). The flora of China (FOC, 2020) reports economic use of species from 301 plant families: 1,068 species used as food, 3,815 species of medicinal plants, 713 plants for feed (grass/honey source), 531 plant species used for fiber, 1,318 timber species, 1,296 species used for ornamental purposes and 989 species used for oil (essential oils, gums, gels) (accessed June 2020).

Reviews of additional datasets such as "The Plant List "(http://www.theplantlist.org/) (TPL, 2020), "World Flora Online" (http://www.worldfloraonline.org/) (WFO, 2020) and the United States of America "Plant Germplasm System" (https://npgsweb.ars-grin.gov/gringlobal/taxon/taxonomysearch) (GRIN-WEP, 2020) were not possible in the same way because those databases are not searchable in a way that allows identification of wild species and uses across different regions of the globe.

Despite the documentation from Kew (2020) and the Flora of China (2020), it remains challenging to provide an estimate on the number of wild plant species that are used across different regions. There are estimates showing that around 7,000 wild plant species are traded globally (Khoury *et al.*, 2019; UNCTAD, 2017), suggesting that approximately 22% (7,000 out of 31,000) of those collected are destined for formal markets (see section 3.2.3).

Other global estimates on wild plant use and associated gathering practices are from certification bodies such as the International Federation of Organic Agriculture Movements (https://www.ifoam.bio/) and International Trade Centre (https://www.intracen.org/). These provide an overview of organic and other standards that deal with wild gathering (mostly for certified organic) and wild harvested products worldwide. Acknowledging that these datasets likely underestimate gathering areas that are not certified, they are the best that were found at the time of the assessment. The study used certification bodies data (base year 2005) to estimate gathering areas, wild harvested products, harvest quantities, processing, collector households and sustainability across the globe. Results suggest that certified wild products are gathered across approximately 62 million hectares of land worldwide. The total global gathering area is estimated to be much larger than reported, as not all existing organic wild gathering projects were identified. According to this report, the global figure may in fact be between 78 and 104 million hectares. For comparison, the total land area of the planet is estimated at 13,003 million hectares, 4,889 million hectares of which are classified as 'agricultural area' by the FAO (this is 37.6% of the land area) (F.A.O., 2017).

The largest gathering areas were reported to be in Africa (26.8 million ha) and Europe (26.7 million ha). The ten countries with largest registered areas where wild products are gathered include Romania, Kenya, Zambia, Finland, Azerbaijan, China, South Africa, Uganda, Namibia and Bolivia. These countries cover nearly 92% of the total reported registered wild gathering area. In Europe, Finland and Romania were reported to have the largest gathering areas followed by Bulgaria, Iceland and Albania. The two countries in Africa with the largest reported gathering areas (Kenya and Zambia) have only few gathering activities officially recorded.

Globally, the ten products which are harvested in largest quantity are bamboo shoots, Brazil nut, lingonberry, rosehip, tea seed for oil, blueberry, iron walnut, green laver, coconut and white mushroom. These products make up 136,411 of the 223,754 tons (61%) of globally reported harvests (IFOAM/ITC, 2007). The highest quantity (138,426 tons) was reported harvested in Asia, from a relatively small area (6.2 million ha). Approximately 200 different wild plant products were reported harvested in Europe. Wild berries and mushrooms were reported to be the dominant wild harvested products there. The highest amounts were harvested in Romania, Russia and Bulgaria as well as Serbia and Montenegro, Bosnia and Herzegovina and Albania. In Africa, the most important products, in terms of quantity, were reported to be rosehip, argan oil, gum Arabic, shea butter and honeybush (International Finance Corporation (IFC), 2018). The most important wild harvested products in North America are wild rice, maple syrup, wild blueberries and blue green algae. Unlike Canada, organic wild gathering in the United States of America is of less significance. Brazil nuts were reported to be the most important wild harvested product in Latin America, harvested mostly in Bolivia, Peru and Brazil. Other important products are coconut, heart of palm and rosehip. In terms of gathering area Bolivia was reported to be the leading country, followed by Brazil, Peru and Guatemala.

Asia shows the widest variety of harvested products (approximately 241). Products such as bamboo shoots, walnuts, tea seeds, seaweed, berries and mushrooms are harvested in large quantities (International Finance Corporation (IFC), 2018). These products make up more than 80% of the total harvest. China is the leading country in Asia in terms of registered gathering areas (International Finance Corporation (IFC), 2018). An even larger area was reported in Azerbaijan, but the certification status was not clear. China is also the country with largest reported harvesting of organic wild harvested products in terms of weight. In Australia and Oceania, organic wild gathering has little commercial importance. Products include noni, sandalwood, sea weed, kangaroo grass and honey.

There was almost no data provided on registered areas or quantities.

Estimates of wild useful fungi, including those presented in the Kew reports (Willis, 2018), are largely based on a 2004 report from FAO (Boa, 2004) which is now somewhat out of date. The sustainable use assessment presents a comprehensive review of more recent literature on the various uses of wild fungi in section 3.3.2.3.4. A bit of information here demonstrates the complexities and rapid changes in this area. For example, 282 species of

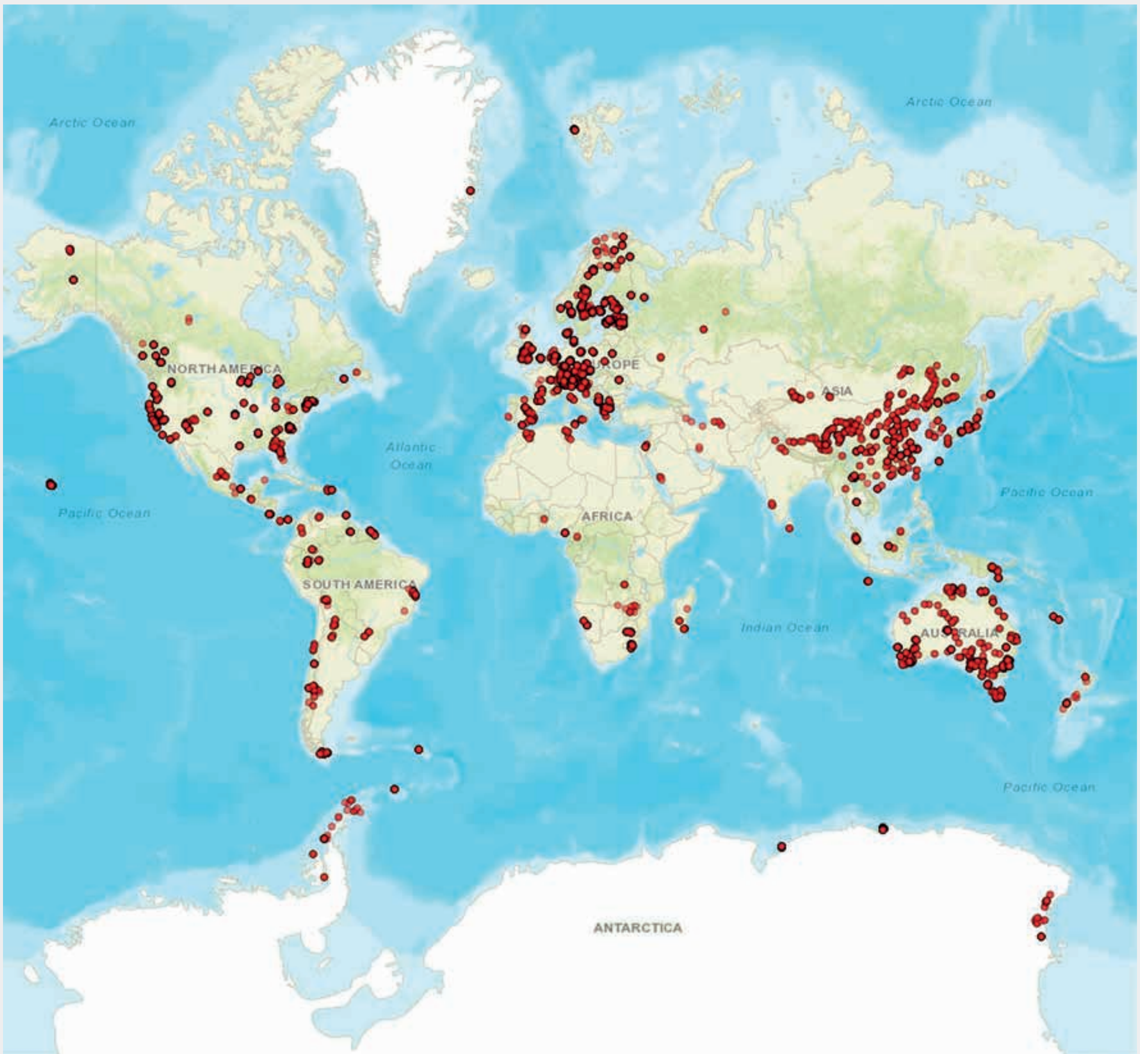


Figure 3 5 **Locations of samples in the global wild fungi database.**

This map is directly copied from its original source (Větrovský et al., 2020) and was not modified by the assessment authors. The map is copyrighted under license CC-BY 4.0 and copyright © 2019 Esri. *The designations employed and the presentation of material on the maps used in the assessment do not imply the expression of any opinion whatsoever on the part of IPBES concerning the legal status of any country, territory, city or area or of its authorities, or concerning the delimitation of its frontiers or boundaries. These maps have been prepared or used for the sole purpose of facilitating the assessment of the broad biogeographical areas represented therein and for purposes of representing scientific data spatially.*

wild fungi are listed on official governmental legislation or guidelines as 'fit for commercialization' in Europe alone (Peintner *et al.*, 2013). Moreover, taxonomic description of fungi is far from complete and an estimated 2 million species are yet to be described (Willis, 2018). In 2019 alone 1,886 species of fungi were scientifically named (SOTWP, 2020). This knowledge gap includes widely-used and internationally traded species. For example, a study of a packet of dried porcini mushrooms purchased at a supermarket contained three species of porcini relatives previously unknown to science (Dentinger & Suz, 2014). Use of fungi as a food source is particularly important in IPBES regions such as Central Asia and Europe. One of the most recent global datasets on fungi is presented in **Figure 3.5**. The global fungi database, from which this figure was generated, contains over 600 million observations of fungal sequences across the world and over 17,000 samples with geographical locations (Větrovský *et al.*, 2020).

3.2.1.3 Terrestrial Animal Harvesting

Humans use terrestrial animals for very different purposes, such as food-feed and pets. In 2013, the United Nations Environment Programme-Worl Conservation Monitoring Centre (UNEP-WCMC), in partnership with the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) secretariat, brought various data-holdings together into one comprehensive data portal to assist Parties to implement biodiversity Multilevel Environmental Agreements using available data. This global dataset documenting use of terrestrial animals is called Species+ (https://speciesplus.net). Species+ contains information on all species listed in the Appendices of the Convention on International Trade in Endangered Species of Wild Fauna and Flora, and other family listings and species included in the annexes to the European Union wildlife trade regulations.

A recent global estimate shows that of 31,500 terrestrial bird, mammal, amphibian, and squamate reptile species, 5,579 species (18%) are used and traded globally (Scheffers *et al.*, 2019). Reptiles, for example freshwater and marine terrestrial turtles, lizards, snakes, and crocodiles are widely used by indigenous people. There are very different estimates on the number of wild animals used as food sources. Estimates suggest that globally, as many as 2,000 species of invertebrates, amphibians, reptiles, birds and mammals are used for food and considered as wild meat (Ingram *et al.*, 2015a; Redmond, 2006; Stafford, Preziosi, & Sellers, 2017). However, Marsh et al (2021) (after Butchart (2008)) reported over 4,500 wild bird species alone are used for food and pets (Butchart, 2008).

Use of mammals for food is reported from North America, Africa, Europe and central Asia **(Figure 3.2)**. The International Union for Conservation of Nature Red List (version 2021.1) has coded 1,248 mammal and 250 reptile species for use as food. Global estimates of hunting (section 3.3.3) highlight regional differences and again challenges with data collection. Estimates of annual offtake rates from forests in Central Africa, for example, range between 1.6 and 11.8 million tons of meat per year. Estimates in the Brazilian Amazon range between 0.07 and 1.3 million tons per year. No similar reviews were found for Asia, where there are still insufficient site-level hunting data to make adequate comparisons. Off-take data are similarly scarce for animal communities in savanna habitats in Africa and South America (Coad *et al.*, 2019). A meta-analysis of 78 hunting studies from sites located in Central America, Amazonia and the Guiana Shield, recorded a total of 90 hunted mammal species including 12 primates, 6 ungulate and 8 rodent genera. As in Africa, ungulates and rodents make up the majority of the wild meat offtake in neotropical communities. In the Amazon Basin, with regional variations, much of the wild meat offtake is medium-sized ungulates such as white-lipped peccary *(Tayassu peccari)*, collared peccary *(Pecari tajacu)*, white-tailed deer *(Odocoileus virginianus)* and various brocket deer *(Mazama* species) and tapir *(Tapirus* species).

3.2.1.4 Logging

Estimates for the total number of tree species used vary somewhat. Both the global tree assessment (Global Tree Assessment, 2020) and estimates from the world flora online (WFO, 2020) list around 60,000 tree species across the world. Estimates differ on the number of wild tree species that are used. The FAO has previously reported 34,000 species, including fruit- and nut-trees and their wild relatives, are used on a regular basis for a range of uses, including logging, environmental, social and scientific purposes, and food (FAO, 2014c). The global tree assessment estimates 12,000 species as having at least one use, and many have a variety of uses (BGCI, 2021). According to the International Union for Conservation of Nature red list (IUCN, 2020b:3), 17,510 tree species (29.9% of all tree species), are considered threatened, 7,400 species (12%) from logging. The 2021 state of the world's trees report also state that "the second major threat to tree species, is direct exploitation, especially for timber, impacting over 7,400 tree species" (Global Tree Assessment, 2021).

Although the amount of timber harvested (volume) from wild and plantation forests is recorded in several global datasets, there is little or no information available about the ways in which those trees were felled and removed from the landscape. In other words, the practice is not recorded, only the result. It is well established that clear felling is prevalent in boreal and temperate forests, and selective logging is the dominant timber harvesting practice in natural tropical forests.

Over 20% of the world's tropical forests have been selectively logged (Bicknell, Struebig, & Davies, 2015). Furthermore, in the majority of the cases the data on selective logging refers only to the minimum felling diameter (normally between 45 and 60 cm diameter at breast height (DBH) only. However, selective logging practices also include other management actions such as type of cutting and regeneration planning. Therefore, the International Tropical Timber Organization estimates that less than 10% of the total permanent forest estate of tropical countries is managed sustainably (Poudyal, Maraseni, & Cockfield, 2018).

3.2.1.5 Non-extractive use

Global estimates on wild species use for non-extractive practices (section 3.3.5) tend to be undocumented, and therefore there are very limited global statistics available. In contrast with fishing, gathering, hunting and logging, non-extractive uses tend to be based in experiencing the whole ecosystem. For example, worship in sacred groves includes all the species in the grove and its vicinity. Recreational tourism may focus on charismatic species but encompasses the whole park / coral reef experience. Forest therapy uses the whole forest. Species-specific tourists, such as bird- / butterfly-watchers, aim to view every species in the taxonomic group and making these observations in their native habitats in part of the experience.

While non-extractive uses are not always directly tied to specific species, they are an important part of wild species use and generate significant amounts of revenue worldwide. Balmford *et al.* (2015) report 8 billion tourist visits per year to protected areas around the world, generating approximately 600 billion United States dollars. It is estimated that tourism revenues generate 120.1 billion United States dollars in gross domestic product to the global economy (WTTC, 2019a). Cultural and economic values from the United Nations Educational, Scientific and Cultural Organization (UNESCO) Word Heritage Sites are also linked to wild species **(Figure 3.6)**. For example, analysis of the data from United Nations on cultural landscapes (http://whc.unesco.org/en/, accessed February 2021) reveals that 23% (25 out of 113) of world heritage sites can be associated directly or indirectly with use of wild species (with different degrees of domestication). However, with the exception of recreational use of wild species, there is limited to no global data on the status and trends of other non-extractive uses such as ceremonial and cultural use, medicinal, and educational use (see section 3.3.5).

In addition of the datasets available for the different practices there are also worldwide repositories such as the Global Biodiversity Information Facility (GBIF) that gather data for different taxa. The Global Biodiversity Information Facility platform (https://www.gbif.org/), which currently houses 1,4 billion records (accessed 15th June 2020), documents the occurrence of a species at a given

UNESCO WORLD HERITAGE LIST

Category of site: Cultural site, Natural site, Mixed site

Sites in danger: Cultural site, Natural site, Mixed site

Figure 3 6 **Locations of UNESCO cultural and natural landscapes around the world.**

Based on (UNESCO, 2021) © 1992–2022 UNESCO/World Heritage Centre.
See data management report for the figure at https://doi.org/10.5281/zenodo.6462960.

time and place, however data on wild species use is not reported systematically.

The results of this review show that while there is a vast legacy of available data on species taxonomy and ecology for different taxa, most datasets do not distinguish wild from domesticated species, making this assessment on the use of wild species (as defined in the sustainable use assessment scoping document) very challenging. Although there is available data on taxonomy and ecology for different taxa, particularly in germplasms/herbariums across the world, lists of wild species available for some taxa are very incomplete. Even for the taxa where there are lists of wild species available, the focus is on biological conservation or economic value related to trade and markets rather than specifically on use as defined here. These reports are framed under different perspectives and goals (see Chapter 2) and on a practice by practice or use by use basis.

Another concern regarding the available datasets is that while the reporting focuses on a use-by-use basis, a single wild species is often used for a variety of purposes. As shown in **Table 3.1**, single species of wild plants, animals and fungi often are used for a variety of reasons (as food source, raw materials, rituals, culture and community identity). Successful and sustainable use is often associated with specialized knowledge and skills of the multifunctional use (Carvalho Ribeiro *et al.*, 2018). Throughout generations indigenous peoples and local communities often cultivate specialized knowledge and maintain skills in ways that support community well-being and maintain nature's contributions to people. These comprehensive uses of single species are not yet captured in global datasets.

In sum, although there have been advances in recent decades, there is not yet a global, harmonized observation system for delivering regular, timely data on species status and trends of biodiversity change, particularly on species that are used by humans. Core elements of this developing data infrastructure have been prototyped. For example, the "Map of Life" website (http://mol.org/) couples raw data on species biology (but not on use) with modelling approaches to inform policy making (Jetz *et al.*, 2019).

3.2.2 Global Indicators

IPBES, in the context of the global and regional assessments, reviewed and systematized a list of 345 global indicators (IPBES Technical Support Unit on Knowledge and Data, 2021). From the list of indicators reviewed by experts in the context of the IPBES global assessment of biodiversity and ecosystem services, the ones likely suitable for measuring status and trends in the sustainable use assessment were selected. In order to update this list, additional indicators from the Biodiversity Indicators Partnership (https://www.bipindicators.net/; accessed October 2020) were included. The Biodiversity Indicators Partnership provides the best available information on status and trends of biodiversity, which helps to monitor progress towards the Convention on Biological Diversity and other multilateral environmental agreements. At the moment, the Biodiversity Indicators Partnership integrates indicators grouped into 8 themes. The themes on sustainable use (22 indicators) and species (42 indicators) are those most likely to apply to the sustainable use assessment.

Most of the indicators reviewed by IPBES are from the Sustainable Development Goals and the Aichi Biodiversity Targets. There are 241 indicators to assess progress towards the achievement of Sustainable Development Goals (https://unstats.un.org/sdgs/indicators/; accessed October 2020). There are 22 indicators as part of the Biodiversity Indicators Partnership. For the current assessment, indicators were selected based on a three-stage process from these three sets (IPBES, SDG – Sustainable Development Goals, and BIP – Biodiversity Indicators Partnership). Indicators were chosen through a set of recurrent stages (initial selection, narrowing, assigning usefulness):

Stage 1 - Initial selection

1. Covers those boxes and arrows of the IPBES conceptual framework that are particularly relevant to sustainable use of wild species,
2. Relevant for different stakeholders and end users (i.e., policy- and/or decision-relevant),
3. Reflects various knowledge systems, diverse worldviews and multiple conceptualizations of values,
4. Relevant at different spatial and temporal scales.

Stage 2 - Narrow the set of indicators (IPBES global assessment used ~30 indicators) taking the following into consideration which indicators

1. Contributes best to the socio-ecological narrative for sustainable use (i.e., reflects both ecological and social aspects),
2. Provides the most useful information on the sustainability of the use of wild species,
3. Need to be developed to reflect the multi-dimensionality of sustainable use of wild species,
4. Provide the most relevance for future monitoring of sustainable use of wild species,

5. Reflect interdependencies and trade-offs (indicators more connected to others provide more nuanced information),

6. Apply across taxa and practices (generic indicators). If not possible, selections should include specific indicators for the major taxa and practices,

7. Are most useful for ongoing assessments.

Stage 3 - Further considerations on the usefulness of the indicator for the sustainable use assessment

1. Are data already available (X) or under active development (Y)?

2. Is the indicator suitable for communication?

3. Is there a possibility for aggregation or disaggregation of data used elsewhere (e.g., National data aggregated to form a global indicator)?

4. Is it an indicator for the Sustainable Development Goals?

This review resulted in a total of 47 meaningfully useful indicators for the assessment as defined in this assessment. Fifteen indicators were likely to meaningfully contribute to estimating status and trends of (sustainable) use of wild species **(Table 3.2)**. Thirty-two indicators relate specifically to the sustainable use assessment practices/uses **(Box 3.1)**. This is a notably small number of indicators given that we reviewed approximately 1000 possible indicators against these criteria (including 245 indicators used for the Sustainable Development Goals, 345 from IPBES, and 300 from the Biodiversity Indicators Partnership).

The analysis of the selected indicators started by associating each indicator to the list of key elements of sustainable use reviewed in Chapter 2 (section 2.2.6):

1. Respect laws, policies and institutions
2. Respect local community rights and access
3. Effective interlinkages among levels of governance
4. Local communities empowered
5. Respect customary law
6. Management and monitoring plans in place
7. Adaptive management specified

Table 3 2 **Selected status and trends indicators.**
Abbreviations: SDGs: Sustainable Development Goals.

	Source	Name and brief description of the indicator
1	McRae *et al.*, (2022)	A global indicator of utilized wild species populations: regional trends and the impact of management
2	Marsh *et al.*, (2021)	Prevalence of sustainable and unsustainable use of wild species inferred from the International Union for Conservation of Nature's Red List
3	Biodiversity Indicator Partnership, Tierney *et al.*, (2014)	"Use it or lose it"
4	Biodiversity Indicator Partnership, Khoury *et al.*, (2019)	Comprehensiveness of conservation of socioeconomically as well as culturally valuable species
5	IPBES-SES	Species Habitat Index (wild species) Species Status Information Index
6	Aichi Biodiversity Target 13	Red List Index (impacts of utilization/wild relatives of domesticated animals)
7	Biodiversity Indicator Partnership, IPBES	Proportion of local breeds (wild species) classified as being at risk, not-at-risk or at unknown level of risk of extinction
8	Biodiversity Indicator Partnership	Biodiversity Intactness Index
9	Sustainable Development Goal 2	Indicator 2.5.1 Number of plant and animal genetic resources for food and agriculture secured in either medium or long-term conservation facilities (wild species)
10	Biodiversity Indicator Partnership	Number of species extinctions (birds and mammals) (used species)
11	Faitrade International	Trade volume in Fairtrade certified goods (wild)
12	Sustainable Development Goals/ Organization for Economic Cooperation and Development	Number of intangible cultural heritage practices – under the category of 'knowledge and practices concerning nature' per country (sustainable use)
13	Sustainable Development Goals /Organization for Economic Cooperation and Development	Number of countries with national instruments on biodiversity-relevant taxes, charges and fees
14	IPBES	Biodiversity Engagement Indicator
15	Biodiversity Indicator Partnership	Living Planet Index (utilized/non utilized species)

8. Participatory approach to decision-making
9. Use of multiple knowledge systems
10. Minimize ecological impacts
11. Minimize waste
12. Restore/improve ecological context
13. Foster socioeconomic benefits
14. Provide local capacity building
15. Fair and equitable sharing of benefits
16. Additional community benefits
17. Raise understanding and awareness

These key elements were included into broad categories such as governance (1 to 5), management and monitoring (6 to 9), ecological impacts (10 to 12), socio economic benefits (13 to 16) and education (17) (see Chapter 2). Although these global indicators tend to fill in one (or in the maximum two) principles there are no indicators that cover all principles: governance, management & monitoring, ecological impacts, socioeconomic benefits and education.

This suggests a need to adapt these global indicators sets to better represent the holistic dimensions of sustainable use of wild species.

Our review of the global indicator sets also shows that most of the indicators developed and used by the Sustainable Development Goals, the Convention on Biological Diversity (Biodiversity Indicators Partnership) and IPBES would need to be adapted in order to target wild species that are used. **Table 3.2** lists indicators that can currently be used to assess status and trends in the use of wild species. The ones in bold are described in more detail below the table. The additional sources are included here for reference, but are not included in the more detailed analysis.

The following text provides details regarding the key sources listed in **Table 3.2**. Those presented in more detail are those that report on the most recent global indicators from the peer-reviewed literature.

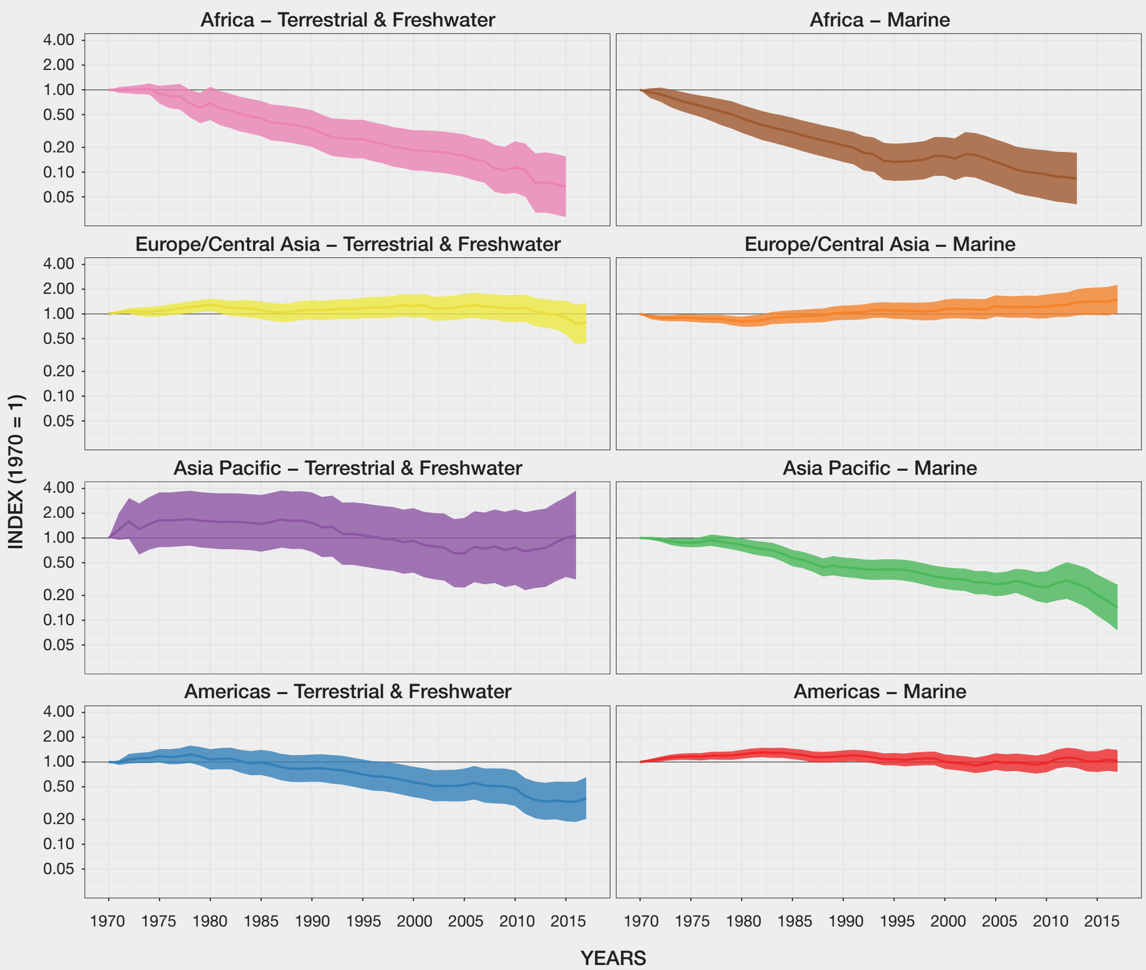


Figure 3 7 **Index of utilized populations for IPBES Regions.**

Abbreviations: TFW: Terrestrial and Freshwater, M: Marine Source (McRae *et al.*, 2022) under license CC BY-NC-ND 4.0.

Source 1: The global indicator developed by McRae *et al.* (2022) follows the method used to calculate the Living Planet Index (https://www.bipindicators.net/indicators/living-planet-index). McRae *et al.*, (2022) used a global data set of over 11,000 time-series to derive indices of 'utilized' and 'not utilized' wild species and assess global and regional changes, principally for mammals, birds and fish. Their work also explored the role that targeted management has in predicting population trends in utilized populations. The results of this work show that from 1970–2016 wild species population trends globally, both used and non-used, are negative **(Figure 3.7)** for both terrestrial and freshwater (TFW) and marine (M) species for all IPBES regions. Note that the trends being shown here are for populations, not for sustainable use.

On average, utilized populations declined by 50% over the 46-year period (0.41-0.62) and non-utilized populations declined by 3% (0.80-1.18). **Figures 3.7** and **3.8** show the estimated total change from the best linear mixed-effect model (binomial and location as random effects). Coefficients show the estimated overall change (log10) for mammals, fish and birds. This work found no significant interaction between taxonomic group and utilization; however, it does show utilized populations tend to decline more rapidly, especially in Africa and the Americas (McRae *et al.*, 2022). However, where utilized populations are managed, there is a positive impact on the trend. This work corroborates that use of species can either be a driver of negative population trends, or a driver of species recovery, with numerous species and population specific case examples making up these broader trends (see section 3.3 for more details and case studies).

The role of management, especially with regards to trade, has been controversial. A considerable body of research on vertebrate species in international trade reports an overall perverse trend in use, with management having a limited mitigating effect sustainability species (Morton, Scheffers, Haugaasen, & Edwards, 2021). International trade databases such as the Convention on International Trade in Endangered Species of Wild Fauna and Flora (https://trade.cites.org/) can shed light on this by reporting both negative and positive effects on population status. In some cases, economic incentives to use a listed species can be directly linked to facilitating recovery and demonstrating non-detrimental use. The role of the Convention on International Trade in Endangered Species of Wild Fauna and Flora can therefore be pivotal for linking use of species with its management and recovery plans.

Source 2: the International Union for Conservation of Nature red list data assess species risk of extinction in relation to threat categories: Least Concern (LC), Near Threatened (NT), Vulnerable (VU), Endangered (EN), Critically Endangered (CE). Red list data show that while use is considered a threat for some species, for others use is not associated with red list threat categories. For example, the work by Marsh *et al.* (2021) shows that for the 10,000 wild species where use and trade data are reported, use

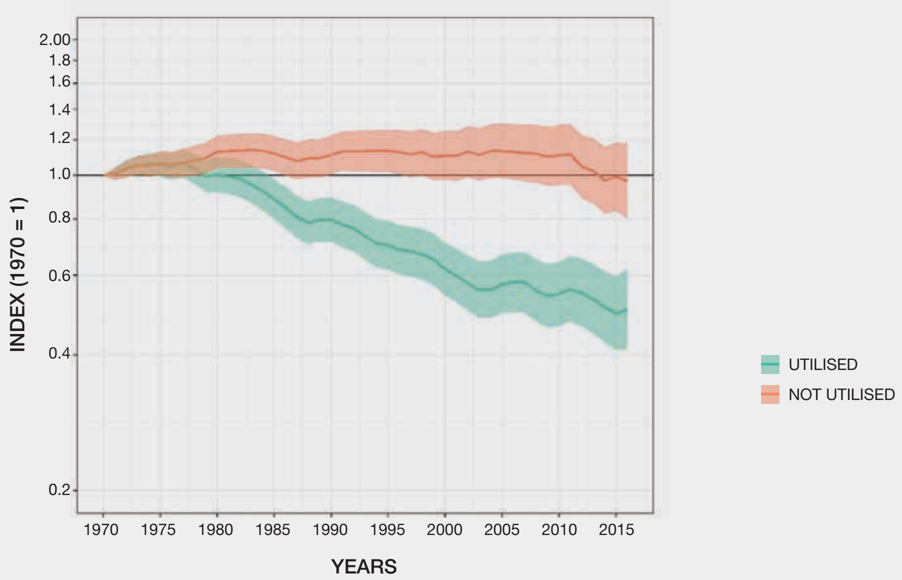


Figure 3 8 **Global trends in utilized vs non utilized species for species of bird, mammal and fish.**

Source (McRae *et al.*, 2022) under license CC BY-NC-ND 4.0.

is likely unsustainable for 16% of species. However, the majority (72%) of species that are used are not threatened, with 34% of used species having stable or improving population trends. Marsh and colleagues (2021) suggest that use is likely to be sustainable for the majority of the species analyzed.

Across Near Threatened (NT) and threatened species, a higher overall proportion of aquatic species than terrestrial species have intentional biological resource use coded as a threat. Among aquatic groups, the taxa with highest prevalence are corals (388 species) and almost all cartilaginous fishes (314 out of 318 species), with fishing the predominant threat. In the terrestrial groups, cycads appear most affected (147 –152 of 255 species), largely due to gathering (147 species) **(Figure 3.9)**. For 48% of the total number of species assessed it was not possible to determine the associations between use and status and trends of the species (Marsh *et al.*, 2021).).

Source 3: The "use or lose it" by Tierney *et al.* (2014) measures trends in the use of wild species, with a focus on both terrestrial and aquatic vertebrate arctic species using two indicators which they developed: the Utilized Species Index (applied at the global scale based on the Living Planet Index) and the Harvest Index (applied in the Arctic region only). The examined data is on amphibian, bird, fish, mammal and reptile species from freshwater, marine and terrestrial realms.

The results of the utilized species index show that between 1970 and 2007 populations of utilized freshwater species declined by 3% and utilized marine species declined by 17%. The populations of utilized terrestrial species decreased 21% over the same period. However, according to this study since the early 2000s the rate of decline of utilized marine and terrestrial species indices has slowed or stabilized. The utilized freshwater species index has been increasing steadily since 2000. The index for species used

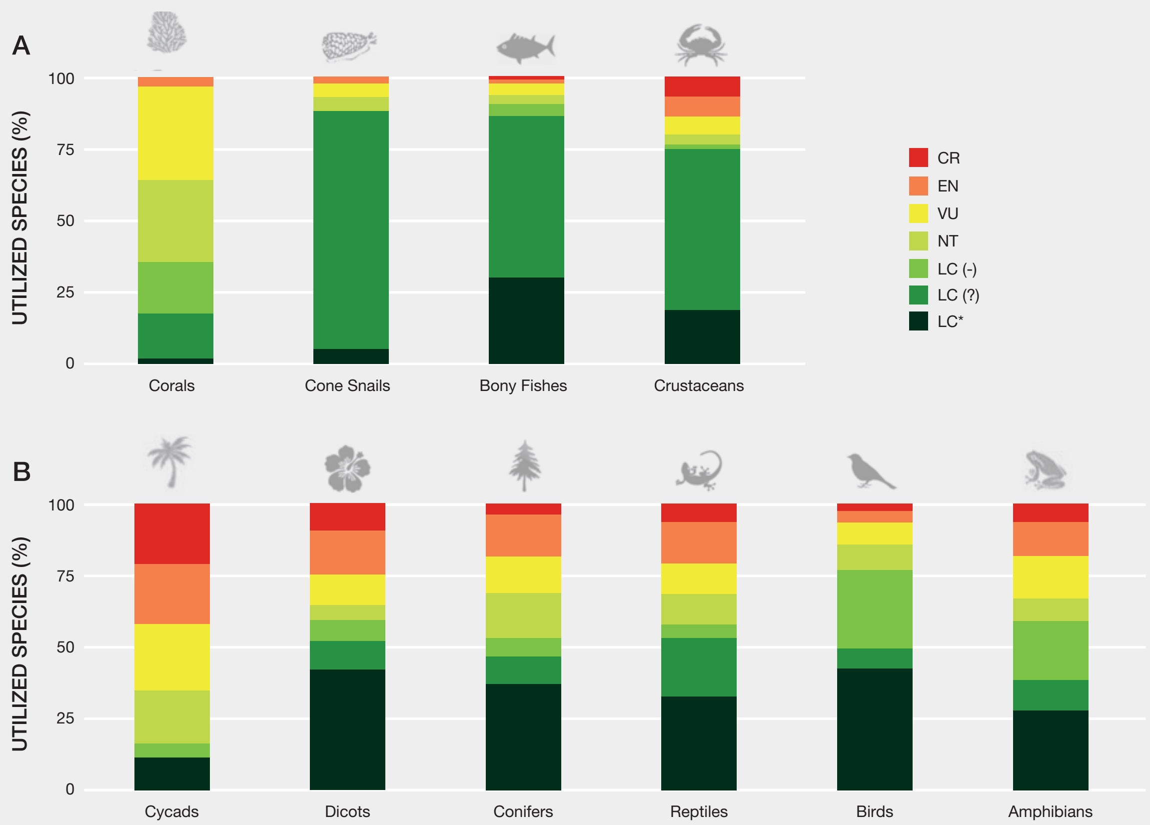


Figure 3 9 **Percentage species by the International Union for Conservation of Nature Red List Category in (A) aquatic and (B) terrestrial groups that are subject to use and trade.**

Legend: LC(-) = Least Concern species with declining population trend; LC(?) = Least Concern species with unknown population trend; LC (*) = Least Concern species with stable or increasing population trend. Note that being LC and having a declining population trend, or being threatened and being subject to use and trade, does not imply that use is a major threat. NT =Near Threatened, Vu =Vulnerable, EN= Endangered, CR=Critically Endangered. Source: (Marsh *et al.*, 2021) under license CC-BY.

reports that from 1997 to 2007 wild species population used for food and species used as pets declined by 17% and 9%, respectively. However, like with terrestrial species, after 2000 this trend inverted **(Figure 3.10)**.

Despite these initial reviews, it remains challenging to undertake comparisons of population trends between utilized and non-utilized species. Most of the datasets available lack the detail needed to do meaningful comparisons amongst utilized vs non utilized species. In this context it is difficult to account for the range of influential factors that could be influencing these trends. Without the ability to account for additional factors and correlated them with these datasets, it is incorrect to assume that use is the primary driving factor of decline or increase in population size.

Source 4: The "Comprehensiveness of conservation of useful wild plants" by Khoury *et al.*, (2019) was included in the Biodiversity Indicators Partnership. In developing the indicator, 6,941 wild plants native to different countries were selected from the United States of Amercia dataset "GRIN-WEP" (2020). The resulting *in situ* indicator shows the extent to which wild species economically used across

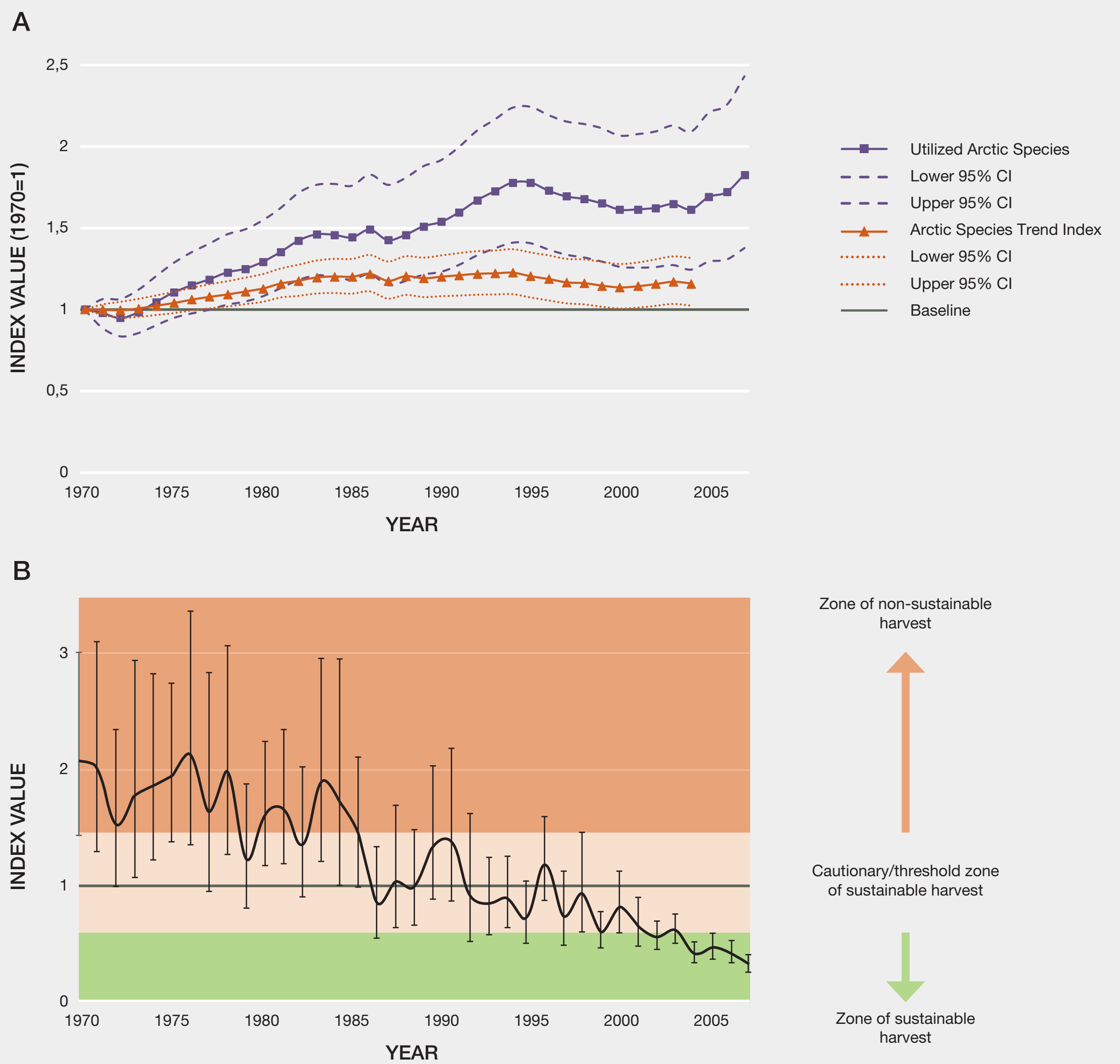


Figure 3 10 **Trends (-/+ 95% CI) in (A) utilized Artic species compared to the Artic Species Trends Index between 1970 and 2007 and (B) Harvest Index of Artic Species between 1970 and 2006, with zones of unsustainable, cautionary and sustainable harvest levels shown.**

Source (Tierney *et al.*, 2014), © Cambridge University Press, license number 5146950749639 CC-BY NC.

the world are conserved *in situ* (through conservation areas). This indicator ranges from 1 (Andorra, Falkland Islands, Gibraltar, Kiribati, Niue, Palestinian Territory, St. Helena, Timor-Leste, and United States of America minor outlying islands) to 642 (Turkey). The mean number of species used across countries is 141; the median is 86. An interactive version of this indicator is available at https://ciat.cgiar.org/usefulplants-indicator/. Areas where *in situ* conservation is likely low are concentrated in Asia and to a lesser extent in Sub-Saharan Africa **(Figure 3.11)**.

In addition to the 16 indicators listed above **(Table 3.2)** there are indicators which can be used to characterize the specific practices detailed in section 3.3 **(Box 3.1)**. The review of indicators highlights that while information on harvesting of terrestrial species is limited, there is a diversity of indicators tracking the off-take and use of marine species. Although there are several indicators used for fisheries, they do not capture the specificities of small-scale fisheries and inland fisheries. These topics are discussed in greater detail in section 3.3.

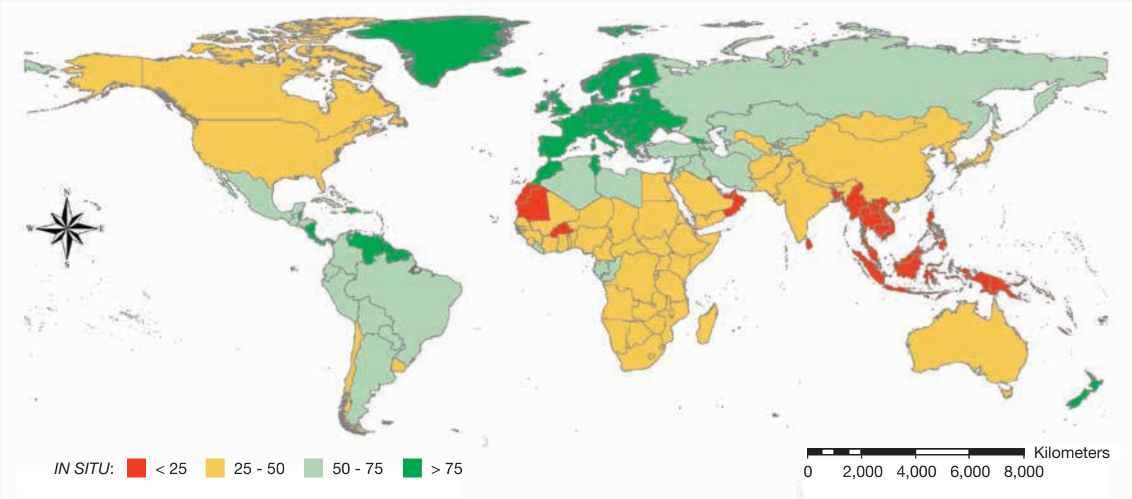


Figure 3 11 ***In situ* conservation indicator.**

This map is directly copied from its original source (Khoury et al., 2019) and was not modified by the assessment authors. The map is copyrighted under license CC-BY 4.0. The designations employed and the presentation of material on the maps used in the assessment do not imply the expression of any opinion whatsoever on the part of IPBES concerning the legal status of any country, territory, city or area or of its authorities, or concerning the delimitation of its frontiers or boundaries. These maps have been prepared or used for the sole purpose of facilitating the assessment of the broad biogeographical areas represented therein and for purposes of representing scientific data spatially.

Box 3 1 **List of possible indicators by practice (selected from indicators developed for the Sustainable Development Goals, Biodiversity Indicators Partnership & IPBES).**

FISHING (13)

1. Sustainable Development Goals indicator 14.4.1 Proportion of fish stocks within biologically sustainable levels
2. Sustainable Development Goals indicator 14.6.1 Degree of implementation of international instruments aiming to combat illegal, unreported and unregulated fishing
3. Sustainable Development Goal indicator 14.b.1 Degree of application of a legal/regulatory/policy/institutional framework which recognizes and protects access rights for small-scale fisheries
4. Number of Marine Stewardship Council's (MSC) chain of custody certification holders by distribution country
5. Number and volume of Marine Stewardship Council certified, consumer-facing products by distribution country
6. Marine Stewardship Council certified catch, Ocean Health Index
7. Red List Index (impacts of fisheries) & number of species listed in the Appendices of the Convention on International Trade in Endangered Species of Wild Fauna and Flora
8. Cumulative human impacts on marine ecosystems
9. Living Planet Index (trends in target and bycatch species)
10. Large reef Fish, policies make adequate provisions to minimize impacts of fisheries on threatened species
11. Illegal, unreported and unregulated fishing
12. Full access to marine resources
13. Inland fishery production

Box 3 1

GATHERING (5)

1. Quantity of mushrooms and truffles, yield (hectogram/hectare) per country
2. Species richness of medicinal plants per country
3. Indigenous and local knowledge trends associated with medicinal plants
4. Number of contracting Parties to the International Treaty on Plant Genetic Resources for Food and Agriculture (adapted for wild species)
5. Red List Index (wild species used for food and medicine) & number of species listed in the Appendices of the Convention on International Trade in Endangered Species of Wild Fauna and Flora

TERRESTRIAL ANIMAL HARVESTING (6)

1. Agreement on International Humane Trapping Standards (AIHT) database
2. Red List Index (internationally traded wild species) & number of species listed in the Appendices of the Convention on International Trade in Endangered Species of Wild Fauna and Flora
3. The Living Planet Index (a measure of the state of global biological diversity based on population trends of vertebrate species from around the world)
4. The species abundance per country (for selected species)
5. Animal individuals hunted yearly per country (for selected species)
6. Proportion of traded wild species that was poached or illicitly traded

LOGGING (5)

1. Sustainable Development Goal indicator 15.2.1 Progress towards sustainable forest management (wild species)
2. Area of forest under sustainable management (wild species): total forest area under management certification (Forest Stewardship Council and Programme for the Endorsement of Forest Certification)
3. Red List Index (forest tree specialist species) & number of species listed in the Appendices of the Convention on International Trade in Endangered Species of Wild Fauna and Flora
4. Timber trade volume in Fairtrade certified goods
5. Total wood removals

NON-EXTRACTIVE PRACTICES (3)

1. Sustainable Development Goal indicator 12.b. Number of sustainable tourism strategies or policies and implemented action plans with agreed monitoring and evaluation tools
2. Importance of protected areas for stimulating eco-tourism and nature-related leisure activities
3. Proportion of jobs in sustainable tourism industries out of total tourism jobs

These global indicators cover different dimensions associated with sustainability and sustainable use **(Box 3.1)**. For example, while most of the indicators for gathering focus on the extent of harvest as a function of area per country, indicators for terrestrial animal harvesting tend to focus on trends in use (overall use increasing or decreasing). Terrestrial animal harvesting indicators also tend to emphasize trade data sources such as the Convention on International Trade in Endangered Species of Wild Fauna and Flora, which may exclude a large number of species which are harvested but not formally traded.

In summary, despite the importance of wild species to economies and livelihoods, relatively few global datasets and indicators have been developed specifically to monitor the status and trends of wild species that people use, except for economically valuable fish species reported on by the biannual reports "State of world fisheries and aquaculture" prepared by the FAO. Particularly lacking are attempts to examine how indicators of species use and sustainable harvest could be linked to provide a broader picture of what, where and how people are using wild species (Tierney *et al.*, 2014).

3.2.2.1 Indigenous Indicators

The importance of wild species in a diversity of livelihood strategies is well recognized, particularly for indigenous peoples and local communities. However little attempt has been made in the available global indicator sets to comprehensively quantify the spatial and temporal scales of sustainable use of wild species occurring specifically in indigenous and local communities across the globe. The United Nations are aware of this gap. The permanent forum requested the inter-agency support group on indigenous peoples' issues, specifically those agencies working on land tenure and changes in land use, to step up cooperation in order to operationalize indicators on these topics as they pertain to traditional territories (lands and waters) of indigenous peoples. The goal was to create a global multipurpose indicator in order to report on status and trends, in line with the Convention on Biological Diversity, the 2030 Agenda for Sustainable Development and the United Nations Declaration on the Rights of Indigenous Peoples (UNDRIP).

The permanent forum recommended that the inter-agency and expert group on Sustainable Development Goal indicators provide support for the inclusion and methodological development of core indicators for indigenous peoples in the global indicator framework (https://www.un.org/development/desa/indigenouspeoples/mandated-areas1/data-and-indicators/recs-data.html). In particular, the inclusion of an indicator on the legal recognition of land rights of indigenous peoples for the targets under Sustainable Development Goals 1 and 2 was requested (United Nations Department of Economic and Social Affairs, 2020).

A key data point for indigenous indicators is regarding spatial patterns of occupancy of indigenous communities around the globe, including estimates for total area and sizes of land plots for habitation and a range of traditional livelihood practices. A recent effort to map the occupancy patterns of indigenous lands at the global scale was undertaken by Garnett *et al.* (2018). In this study, the authors show that indigenous lands seem to have the appropriate scale to support the implementation of several global conservation and climate agreements while also maintaining sustainable local use and local governance institutions. However, details on the scale of sustainable use (both spatial and temporal) were not explicitly presented in this study.

Contrary to the large size of most indigenous lands (large extents that can be mapped at coarse resolutions), identifying the spatial patterns of occupancy of "other" traditional livelihoods, (plots with smaller sizes than can only be mapped at finer grained resolutions) is very challenging. Yet, actors at smaller scales are active natural resource users within many social-ecological systems. The failure to so far comprehensively map and measure the multi-scaled and interwoven distributions of traditional communities' and livelihoods' diverse spatial occupancy patterns likely make these users of wild species invisible to policy makers. For example, in order to estimate scale of use of wild species supporting different types of livelihoods one can, to some extent, explore the spatial scale (grain and extent) of the land consigned by law to different communities and map their rights of use and land tenure regimes. Indeed, traditional communities and their rights are defined by law (including international agreements).

Recognizing and identifying these diverse legal frameworks and the associated spatial occupancy patterns of their territories can be a way forward to estimate the spatial scale of use of wild species globally. However, territoriality and tenure clarification are highly complex, politically driven and often a very slow process. Moreover, while *de jure* standards may be defined, the *de facto* realities might show evidence of positive long-term care and stewardship or negative effects such as failed law enforcement, denied constitutional protections, and in some cases a weak and indiscriminate rule of law. Other data/indicators can then be used, that can complement land ownership datasets in order to provide the best estimate available for different types of uses of wild species.

The next section provided a brief review of key aspects of the temporal scale of use (3.2.3) and economic, ecological and social contexts for sustainable use across the globe (3.2.4). Section 3.3 goes into detail on a practice-by-practice basis.

3.2.3 Temporal scale and use

Use of wild species varies over time. Although there is evidence that temporal scale influences sustainability of use of wild species, based on the review above the temporal dimension has been overlooked in global datasets (section 3.2.1.1) and the global indicator system (section 3.2.1.2). The dedicated attempt here to introduce longer-term temporal indicator dimensions to sustainable use indicators is therefore very much a step forward. There are many insights to be gathered from longer term perspectives, many of which directly challenge more temporally constrained conclusions. Correlative reasoning is sometimes entirely displaced through longer term trend reviews. Another important reason to consider temporal scale for assessing sustainable use is that harvesting intervals vary widely across species. Some species may be subject to seasonal, periodic or annual harvests, others may be biennial or triannual. Timber is often harvested according to a decadal cycle. Other species, such as some wild edible fungi, may be harvested sporadically when they fruit abundantly.

Perception and organization of time is very basic to the internal ordering of all cultures and there are strong evidences of such calendars from all continents across the globe (Dhyani, 2018; Dhyani, Maikhuri, & Dhyani, 2011). Seasonal calendars reveal a body of knowledge about the relationship between people and the environment and underpin local Natural Resource Management (NRM) strategies. These knowledge systems have been built through strong observational, practice-based methods that have been used for centuries. They continue to be enacted and tested, and have sustained consecutive generations by adapting continually, if incrementally, to the local context over time (Woodward & Marrfurra McTaggart, 2019).

Seasonal calendars have been used by indigenous peoples and local communities for generations for monitoring and adaptive management of natural resources, agricultural systems (Bhagawati, Sen, & Shukla, 2017; Jiao *et al.*, 2012; Saylor, Alsharif, & Torres, 2017), climate change (Balehegn, Balehey, Fu, & Liang, 2019; Cochran *et al.*, 2016; Fu *et al.*, 2012; D. Yang & Pomeroy, 2017), water (Woodward, Jackson, Finn, & McTaggart, 2012), and to guide eco-health decision making (SantoDomingo, Castro-Díaz, González-Uribe, Wayúu Community of Marbacella and El Horno, & Barí Community of Karikachaboquira, 2016).The temporal scale of use is also important for measuring the nutritional value and food availability across the "seasonal evenness": when there are many species in the system, the likelihood is increased that one species or another is "in season" at all times (FAO, 2017a; Powell *et al.*, 2015). Seasonal calendars have been relevant in cross-cultural interpretation of indigenous ecological knowledge and a relevant communication tool. While seasonal knowledge for temporal scale of use of wild species has not been sufficiently utilized in sustainable use of wild species or natural resource management (Franco, 2015; Prober, O'Connor, & Walsh, 2011; Woodward *et al.*, 2012), this is beginning to change.

One of the best-known representations of indigenous seasonal calendars in Australia is the poster series developed by multiple researchers and indigenous communities, supported and collated by the Commonwealth Scientific and Industrial Research Organization (CSIRO https://www.csiro.au/en/). This series started with the Ngan'gi seasons calendar in 2009 (on the Daly River, Northern Territory, see **Figure 3.13**) and includes the Tiwi seasons calendar (Prober *et al.*, 2011) (see **Figure 3.14**). The main focus is ecological knowledge and customary activities of resource use.

Examples:

1. **Fodder and leaf litter removal calendars.** Fuelwood and fodder gathering from nearby forests is a frequent and year-round activity, but leaf litter removal is a seasonal practice occurring only during dry winter months (November-March) when there is a lot of leaf litter available on the forest floor (Dhyani, 2018; Dhyani *et al.*, 2011) **(Figure 3.12)**. Seasonal harvesting is enforced by local village forest management committees to ensure sustainable harvest of fodder, fuelwood, litter and other wild plants and fungi (Misra, Maikhuri, Kala, Rao, & Saxena, 2008).

2. **Seasonal migratory calendars of Tibetan pastoralist communities in Tibet** (an autonomous region of China) **and Western Himalayas, India.** Pastoralists in high mountains of Tibet and Indo-Mongoloid Bhotiya tribal sub-communities (Tolchha, Marcha and Jad) primarily adopt centuries old ancestral seasonal migratory livestock raising as a key mechanism for enhancing their ecological sustainability and use first hand observations as ecological indicators

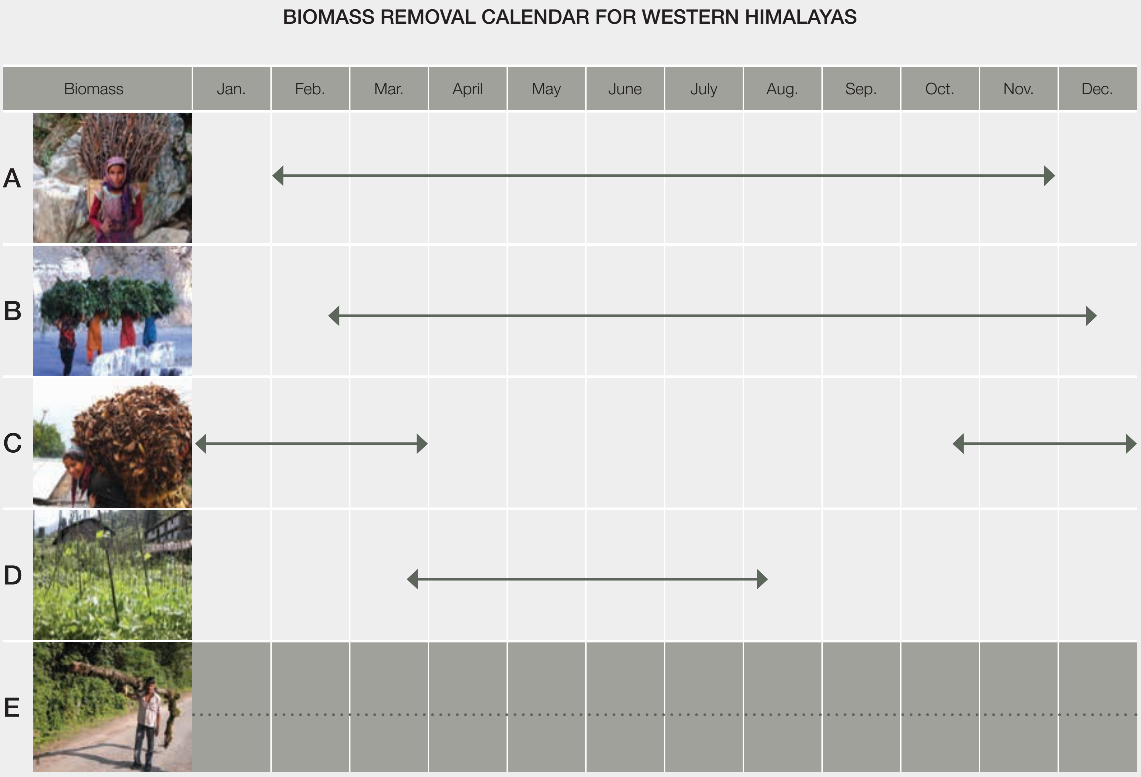


Figure 3 12 **Annual local biomass removal calendar for Western Himalayas.**

A: Fuelwood, B: Fodder, C: Leaf litter, D: Crop support and other wild algae, fungi and plants, E: Timber.
Source: (Dhyani, 2018) © International Society for Tropical Ecology CC-BY NC.

to decide the timing of seasonal activities (Fu *et al.*, 2012; Maikhuri *et al.*, 2011; H. Yang *et al.*, 2019).

3. **Calendar of Tajik community, Xinjiang,** China. Tajik people perceive indicators, including the appearance of migratory birds (*Motacilla alba* and *Motacilla citreola*), the height of grass and the conditions of farmland for conducting their activities for the aims of food production, livestock keeping, and fodder and gathering medicinal plants. They have also developed strategies to keep themselves protected from firewood shortage due to high elevations. These indicators are recognized by local people, associated with their seasonal activities, and passed down through generations.

4. **The Ngan'gi seasons calendar.** This is an indigenous temporal management approach practiced by remote indigenous communities of Pine Creek and Naiuyu Nambiyu in the Daly River catchment, Australia. The Ngan'gi Seasons calendar has informed the scientific understanding of patterns of resource use and relationships between people, subsistence use and river flows in the Daly River catchment (Woodward *et al.*, 2012) **(Figure 3.13)**. The calendar is a relevant guidance approach for sustainable and rotational gathering, hunting and fishing of wild resources. Hunting and gathering of resources start towards the end of the Wet season, known as Wudupuntyurrutu in the calendar, with the harvest of fruits. Saltwater crocodile *(Crocodylus porosus)*, echidna *(Tachyglossus aculeatus)* and rock python *(Liasis olivaceus)* are also actively hunted during this period. The dry season, known as Wurr wirribem filgarri, brings active hunting for freshwater prawn *(Macrobrachium rosenbergii)* in the river and creeks. Indicators of the start of dry season are wind flow from the east and presence of dragonflies that indicates fishing time for barramundi *(Lates calcarifer)*. Wurr bengin derripal, a late wet/early dry season, is a good time to harvest the eggs of magpie goose *(Anseranas semipalmata)* and catfish (*Arius* spp.), but is not yet time for hunting other fish. Resource gathering increases in Wirirr margu with hunting for turtles (*Carettochelys insculpta*; *Chelodina rugosa*; *Emydura* spp.; *Elseya* spp.) and also fish (black bream, *Hephaestus fuliginosus*; archer fish, *Toxotes chatereus*; mullet *Liza* spp.; and freshwater species). During the beginning of the wet season a range of lilies and other water-dependent plants are gathered from swampy areas that include waterlily, red lotus lily, and water chestnut. At this time, native peanut, and bush banana are also harvested. With lower water levels it is easier to harvest mussels, and crabs from creeks and springs.

5. **Urban foraging calendars.** Urban foraging as modern gathering practice has received attention around the world (Friedlander, Stamoulis, Kittinger, Drazen, & Tissot, 2014). Urban foragers make and share foraging calendars that guide them on what to gather in urban landscapes, where and in what seasons. National Geographic developed a guide for the United Kingdom (https://www.nationalgeographic.co.uk/travel/2020/07/a-year-round-foraging-calendar-what-to-pick-and-where-in-the-uk).This not only informs foragers about better foraging approaches but also promotes more sustainable harvesting of wild species from urban spaces that already have a lot of pressure on natural urban green spaces.

6. **Tiwi seasons calendar.** Traditional owners from the Tiwi Islands and the Tiwi land council collaborated with the Commonwealth Scientific and Industrial Research Organization to develop two calendars, a calendar of Tiwi seasonal ecological knowledge and a calendar of wild plants and animals of Tiwi significance **(Figure 3.14)**. The development of the calendars came from a desire to document seasonal-specific knowledge and ecological knowledge of the Tiwi Islands in an appealing format accessible to both students and the broader community, as well as a strong concern about the loss of knowledge as older people pass away.

7. **Seasonal round of harvest activities in Fort Yukon.** The Gwich'in Athabaskans of Fort Yukon, Alaska, follow a strict seasonal round established by their ancestors over centuries. Their calendar of activities has evolved in response to northern environmental conditions such as animal migrations which make them seasonally abundant or absent, ice and snow cover which affect travel and access to resources, and preferences for certain qualities found in resources at specific times of the year (https://www.culturalsurvival.org/publications/cultural-survival-quarterly/wild-food-its-season-seasonal-round-harvest-activities).

8. **Hawaiian moon calendar for responsible fishing practices.** The community in the Ho'olehu Hawaiian Homesteads on the island of Moloka'i is strengthening community influence and accountability for the health and long-term sustainability of their marine resources through revitalization of local traditions and resource knowledge. The traditional system in Hawai'i emphasized social and cultural controls on fishing with a code of conduct that was strictly enforced. Local resource monitors, in conjunction with visiting scientists, are creating a predictive management tool based loosely on the Hawaiian moon calendar to guide responsible fishing practices. Community-sanctioned norms for fishing conduct are being reinforced through continual feedback based on local resource monitoring, education, and peer pressure. Hawaiian community

Figure 3 13 **The Ngan'gi Seasons calendar.**

Source: (CSIRO, 2021). © Tiwi Land Council and CSIRO. CC-BY NC.

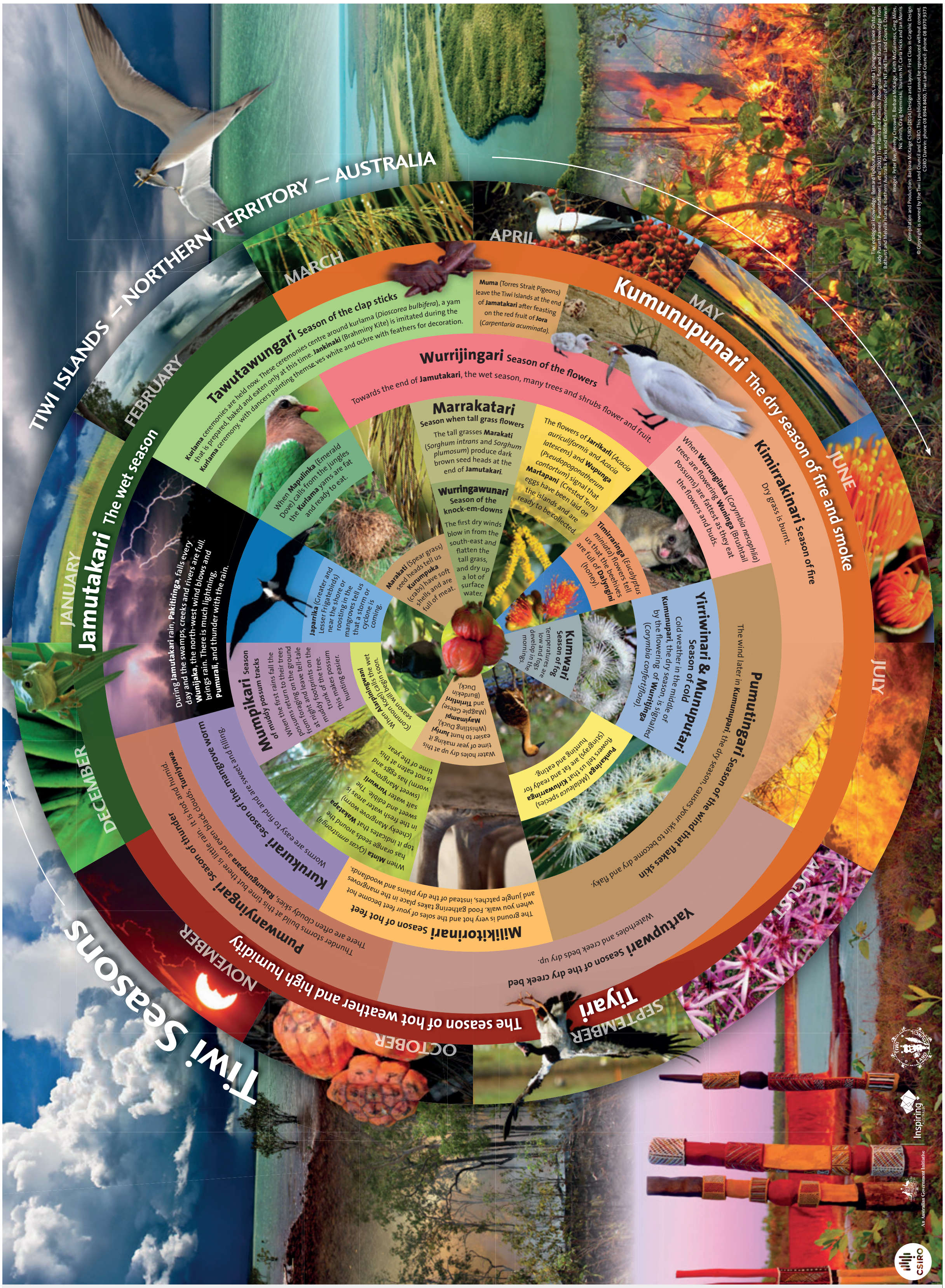


Figure 3 14 **Tiwi seasons calendar.**

This calendar show month of year in the outermost ring, then three “major” Tiwi seasons recognized by weather. Note that Kuwunupunari does not have a shar boundary with Tiyari. Within this ring are smaller seasons recognized by weather or ecological and associated with particular activities. Source: (CSIRO, 2014) © Tiwi Land Council and CSIRO. CC-BY NC.

building and proper cultural protocols are essential to understand and revitalize marine conservation traditions (Friedlander *et al.*, 2014).

9. **Seasonal calendar of Manangis in the Trans-Himalayas, Nepal.** The Manangis, a group of indigenous peoples and local communities, have maintained a dynamic cultural landscape of the trans-Himalayas, Nepal through different socio-economic activities that are reflected in the seasonal calendar of Manang. The seasonal calendar clearly exhibits the typical lifestyle of people influenced by the cold climate: longer photoperiod for agricultural crops, inadequate food materials, important forest and water resources, high tourism activities, skilled trading activities, and topographic obstacles. The Managis sustainably collect the wild resources from common lands only during specified periods. Species include vegetables (*Allium* species), mushrooms including caterpillar fungus *(Ophiocordyceps sinensis)*, and winter fodder grass (Chaudhary, Aase, Vetaas, & Subedi, 2007). The seasonal calendar including harvest of wild species is regulated by traditional knowledge of the indigenous peoples and local communities and social norms monitored by community leaders.

These calendars also reflect seasonal circumstances of access to different areas to hunt and gather. In some areas, the wet season results in tall, matted grasses, which need to be burned when the dry season arrives before people can walk to different areas to hunt and gather. It is a selective rotational system associated with discrete wet (flooding, rain, long grass) and dry seasons (drying, floodwaters abate, grasses are burned, isolated billabongs reappear) in both Day and Tiwi areas – across the whole of the wet-dry tropics – like Llanos and Pantanal in Latin America.

These and other seasonal calendars (e.g celtic tree calendar) are well known amongst indigenous indicators. Indigenous indicators have been recently evolving in the literature, challenging more technocratic views and highlighting that there is an alternative way of including values for guiding indicator development and selection. This work recognizes areas where conventional sustainability indicators cannot be developed for measuring crucial socioecological functions (J. Reid & Rout, 2018, 2020).

3.2.4 Economic, ecological, and social contexts of sustainable use

Wild species are used by billions of people in very different socioecological systems and circumstances around the world. Subsistence gathering, hunting and fishing occur worldwide, as documented in previous IPBES assessments for Africa (IPBES, 2018d), the Americas (IPBES, 2018c); Asia and the Pacific (IPBES, 2018a), and Europe and Central Asia (IPBES, 2018b). Estimates on the number of people who use nontimber forest products, for example, range from 3.5 billion to 5.76 billion globally (Charlie M. Shackleton & de Vos, 2022). FAO also estimated 18% of respondent countries (65% of nation-members of the Organization for Economic Cooperation and Development (OECD) and 4% of countries outside the Organization for Economic Cooperation and Development) are engaged in recreational harvesting of wild foods. Activities commonly undertaken include hunting, angling, mushroom gathering and berry picking (FAO, 2019b). One of the reasons these and the following data range so widely is that many products are used by the harvester themselves or informally traded in small quantities in small village markets, neighborhood exchanges, or amongst kin (see section 3.1 for explanation of informal vs. formal grade).

Individuals, groups, and even companies engage in informal trade. The state of world's forests (FAO, 2014) is one of the few sources available for estimating the value of informal markets across the globe. For the year 2011, FAO estimated the value of global informal trade to be 88,013 million United States dollars. Estimates of informal trade value were higher for Asia and Oceania (FAO, 2014b). Wild species contributions to household income are highly variable ranging from 17% in Acre and Amazonas states in Brazil (Carvalho Ribeiro *et al.*, 2018) to 28.6% of average household income across Latin America, whereas in Asia and Africa forest income shares are 20.1% and 21.4%, respectively (Angelsen *et al.*, 2014). In general, roughly 25-30% of household income in tropical forest countries was from wild forest products in the early 2000s, a percentage almost as high as agriculture (Wunder, Angelsen, & Belcher, 2014).

The same level of market informality is also present in fisheries; especially in developing countries where there are informal markets for small-scale coastal and freshwater fisheries. Although informal and largely unreported, the catch from small-scale fisheries may be large and this informal trade is important to local economies (e.g., in villages or small cities) and to the food security and nutrition of impoverished peoples living in remote areas. Small scale fishing is discussed extensively in section 3.3.1. The lack of monitoring may render the importance of these activities to local communities and some of their environmental impacts, invisible to decision makers (Bartley, De Graaf, Valbo-Jørgensen, & Marmulla, 2015; Doria, Athayde, Lima, Carvajal-Vallejos, & Dutka-Gianelli, 2020).

While subsistence uses often occur somewhat "under the radar" in the informal economy, there is a very large formal economy surrounding wild species. This formal economic activity is collectively referred to by the United Nations as BioTrade (UNCTAD, 2017): the collection, production, transformation and commercialization of goods and services derived from native biodiversity (species and ecosystems) under environmental, social and economic sustainability criteria. Under the Nagoya Protocol on Access to Genetic Resources and the Fair and Equitable Sharing of Benefits Arising from their Utilization, parties are to issue internationally recognized certificates (IRCC) of compliance evidencing that access to genetic resources was based on prior informed consent and that mutually agreed upon terms were established between local communities and research and industry stakeholders. India leads by far in the number of internationally recognized certificates of compliance worldwide (accessed June 2020).

BioTrade subscribes to the objectives of biodiversity-related multilateral environmental agreements including the context of sustainable development and responsible business. It stresses that 70% of the world's poor depend directly on biodiversity and businesses it fosters. BioTrade partners estimate that 86% of species (and their potential uses) are still unknown (UNCTAD, 2017). There are seven established BioTrade Principles and Criteria (BT P&C) as follows: (P1) Conservation of biodiversity, (P2) Sustainable use of biodiversity, (P3) Equitable benefit-sharing, (P4) Socioeconomic sustainability, (P5) Compliance with international legislation and agreements, (P6) Respect for actors' rights, and (P7) Clear land tenure and resources access. These, combined with the four distinctive approaches described within BioTrade (value chain, sustainable livelihoods, ecosystem and adaptive management), greatly contribute to the sustainability of trade in wild species.

While only 20 countries officially participate in BioTrade partnerships, over 12,000 companies worldwide in more than 70 countries have signed up to the United Nations Global Compact, committing to greater environmental responsibility. The number of companies that report on biodiversity in their annual reporting is growing. For example in 2015, thirty-six of the top 100 cosmetic companies and 60 of the top 100 food companies mentioned biodiversity. Sales of BioTrade beneficiary companies reached 5.1 billion United States dollars (UNCTAD, 2017). Approximately 5 million people worldwide from collectors/fishers/ hunters to workers, among others are involved (UNCTAD, 2017).

3.3 PRACTICES AND USES

The use of wild species includes three interacting systems: the wild species themselves, the human practices by which they are obtained from nature, and the uses for which they are intended (Chapter 1, Figure 1.6). Here the status and trends of the use of wild species are reported, organized according to the practices defined at length in Chapter 1: fishing (including lethal and non-lethal use), gathering, terrestrial animal harvesting (including lethal and non-lethal use), logging, and non-extractive practices. These practices are somewhat intuitive, but not always. Thus, readers should be attentive to the definitions and explanations of the practices and why certain organisms (e.g., living shellfish vs. shells) or certain parts of organisms (e.g., tree branches vs. tree fruits, leaves and sap) are discussed in a particular practice category.

Each section begins with an overview presented in a format consistent with ways of thinking most prevalent in that field. This is followed by specific information relevant for the practice. The following section reviews uses according to the structure detailed in Chapter 1: ceremonial/cultural, decorative/aesthetic, energy, food/beverage, medicine/hygiene, recreation, science/education, shelter/construction, and other (see Chapter 1, Figure 1.6). Only the relevant uses are reported upon in each practice section. These categories are not exclusive, and many species have more than one use depending on a range of variables including their biology, habitat, life cycle, knowledge on utilization, existing rules, and regulations. There may thus be some overlap in the reporting. A selection of cases of multiple and complex use systems is discussed in section 3.4 to demonstrate some of the complexities of reporting on status and trends at national and international scales.

When possible, the use categories have structured the reporting in this section. However, in many cases the knowledge about the sustainability of use is not organized according to these use categories. Therefore, in order to increase accessibility to policy makers, in sections where the bulk of knowledge is reported using a different system, hybrid organizing structures were created as an attempt to be attentive both to the organizing structure of this assessment, and the expectations of the readers.

3.3.1 Fishing

3.3.1.1 Introduction

Prior to 1950 large-scale motorized fishing was mainly confined to the North Atlantic and Japan. Marine capture fishery has substantially expanded in the last 70 years in terms of geospatial and vertical distribution, and intensity of catch effort. Automatic identification systems data indicates that industrial fishing currently occurs in over 55% of the global ocean (Kroodsma *et al.*, 2018) although a much smaller footprint is estimated from the same data when a spatial grid of finer resolution is used in the calculation (Amoroso, Parma, Pitcher, McConnaughey, & Jennings, 2018). Relative to coastal ecosystems, high seas ecosystems are much less affected (Halpern *et al.*, 2008; Jackson, 2001). However, reported landings from the high seas has been accelerating since the mid- 20th century with under two million tons in 1950 to over ten million tons in 2008 (FAO, 2010c).

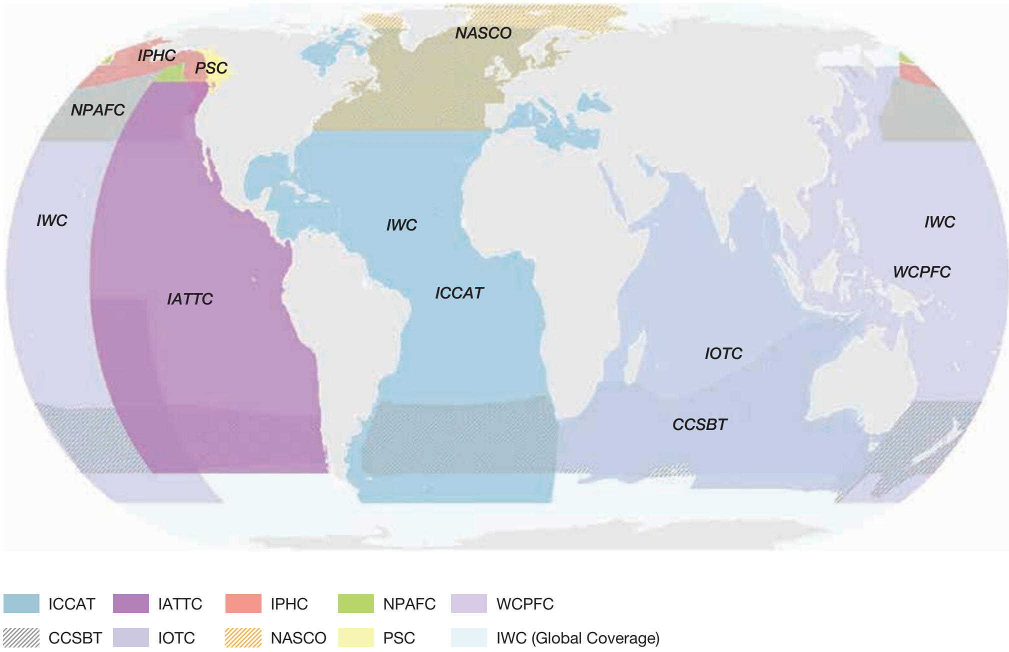
The history of sustainable use of capture fisheries is closely tied with several key events and international agreements. Prominent among those is the United Nations Convention on the Law of the Sea ratified in 1982 by 157 parties. One of its most significant provisions was the establishment of 200-mile exclusive economic zones and introducing the concept of maximum sustainable yield as the default goal of fisheries management. The 200-mile exclusive economic zone allowed countries to exclude wide ranging foreign fishing fleets that earlier were able to legally fish within 12 miles of the national coastline. As a result, several countries established fisheries management systems (e.g., scientific assessment, regulation of harvest) for the newly expanded waters under their jurisdiction. This also led to expansion of many domestic fishing fleets.

The legal framework of the United Nations Convention on the Law of the Sea did not include several fish stocks across multiple exclusive economic zones or in the high seas. The United Nations fish stocks Agreement from 2001 provided international protocols for managing these "straddling stocks" (G. R. Munro, 2000). It mandated the formation of Regional Fisheries Management Organizations (RFMO) to sustainably manage high seas and the straddling stocks. Following the Agreement, there are now 17 Regional Fisheries Management Organizations that cover almost all the high seas fisheries and associated straddling stocks outside national exclusive economic zones. Regional Fisheries Management Organizations are competent and mandated to establish binding conservation and management measures. They provide a formal mechanism for fishing states and states in whose jurisdiction fishery resources occur to meet their international obligation to cooperate to sustainably govern shared living marine resources throughout their distributions (the United Nations Convention on the Law of the Sea Articles 63, 66(5), 118; Code Articles 7.1.5, 6.12 (FAO, 1995a); Agreement on Port State Measure (PSMA) Article 4(1)(b) (FAO, 2010a). Regional Fisheries Management Organizations have played a critical role in multilateral fisheries governance of stocks that straddle or occur beyond national jurisdictions and are highly migratory. While spatial and taxonomic gaps remain, a large proportion of global marine fisheries are now managed by one or multiple Regional Fisheries Management Organizations, and they cover most areas of the high seas **(Figure 3.15)**.

Fishing has impacts on marine ecosystems other than the target species. A range of international agreements have evolved to provide guidance on managing non-target species and vulnerable marine ecosystems (VMEs). Legal instruments establishing international responsibility to conserve associated and dependent species are relatively recent, which first became an obligation under the 1982 Law of the Sea Convention, and was reiterated and clarified further in subsequent United Nations resolutions (United Nations 1982 [Article 119], 1995 [Article 5(f), Article 10(d), and Annex 1]; 2006a, b). These provisions were elaborated further in subsequent instruments and guidance from other multilateral organizations. This includes the 1995 code of conduct for responsible fisheries of the FAO, which calls for the sustainable use of aquatic ecosystems and promotes the conservation of biodiversity and ecosystems by minimizing fisheries impacts on non-target species and the ecosystem in general (FAO, 1995a). FAO has also produced a voluntary international plan of action on reducing the incidental capture of seabirds in longline fisheries (FAO, 1999), an international plan of action on the conservation and management of sharks (FAO, 1999b), international guidelines on reducing marine turtle fishing mortality (FAO, 2009), and broad guidelines on managing fisheries bycatch (FAO, 2011). These new instruments and international guidelines broadened the mandate of pre-existing Regional Fisheries Management Organizations, expanding their mandates from one target species to meet newer expectations for ecosystem-based management and precautionary approaches, i.e., establishing explicit limits of acceptable impacts on fish and non-fish bycatch species, associated or dependent and threatened species (Fisheries Agency of Japan, 2007; Lodge, Anderson, & Lobach, 2007; United Nations, 2006b, 2006a).

Fisheries targeting relatively fecund species can have profound impacts on co-occurring incidentally caught or bycatch species with delayed maturation, low fecundity and other life history traits that make them vulnerable to anthropogenic causes of mortality. While target stocks may be sustainable, the conservation status of bycatch species and other associated and dependent species is often not known. For instance, 47 of 68 fisheries that catch marine resources managed by Regional Fisheries Management Organizations have no observer coverage (Gilman, Passfield, & Nakamura, 2014); for the vast majority of the ca. 4.6 million fishing vessels globally, information on non-retained catch is absent. In most fisheries, there are large gaps in understanding of life histories for many marine species. Information on total cumulative anthropogenic levels of fishery removals from an individual population, knowledge of the conservation status of individual populations, and deficits in monitoring are all unknown. Data

A SPECIES-SPECIFIC RFMOS



B GENERIC REGIONAL FISHERIES MANAGEMENT ORGANIZATIONS (RFMOs)

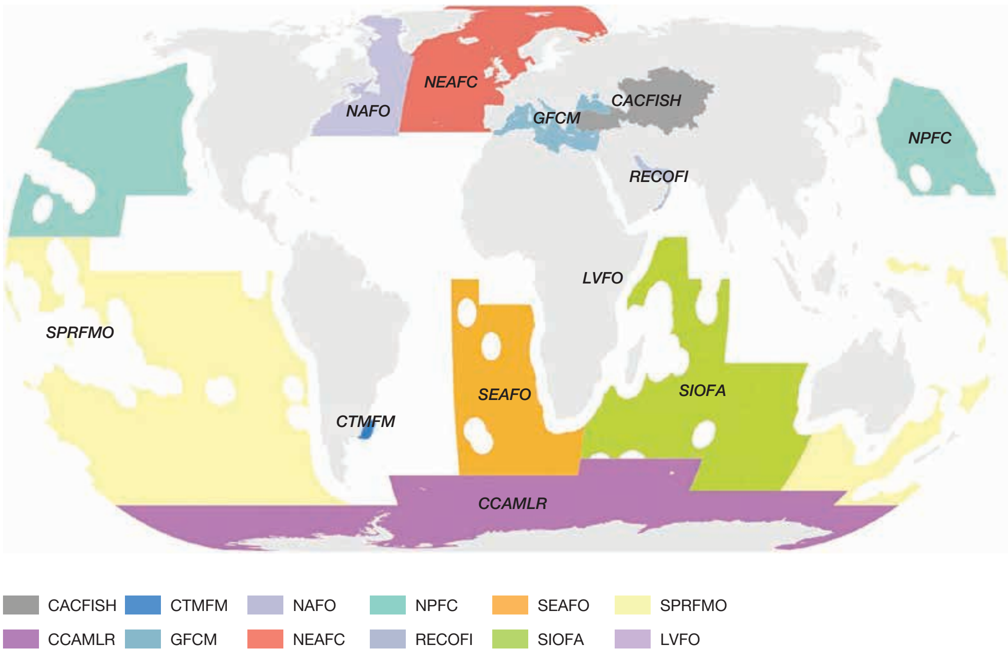


Figure 3 15 **Species-specific regional fisheries management organizations (top) and other regional fisheries management organizations (bottom).**

Abbreviations: ICCAT: International Commission for the Conservation of Atlantic Tunas; IATTC: Inter-American Tropical Tuna Commission, IPHC: International Pacific Halibut Commission; NPAFC: North Pacific Anadromous Fish Commission; WCPFC: Western and Central Pacific Fisheries Commission; CCSBT: Commission for the Conservation of Southern Bluefin Tuna; IOTC: Indian Ocean Tuna Commission; NASCO: North Atlantic Salmon Conservation Organization; PSC: Pacific Salmon Commission;

IWC: International Whaling Commission; CACFISH: Central Asian and Caucuses Regional Fisheries and Aquaculture Commission; CTMFM: Joint Technical Commission of the Maritime Front; NAFO: Northwest Atlantic Fisheries Organization; NPFC: North Pacific Fisheries Commission; SEAFO: South East Atlantic Fisheries Organization; SPRFMO: South Pacific Regional Fisheries Management Organization; CCAMLR: Commission for the Conservation of Antarctic Marine Living Resources; GFCM: General Fisheries Commission for the Mediterranean; NEAFC: North-East Atlantic Fisheries Commission; RECOFI: Regional Commission for Fisheries; SIOFA: Southern Indian Ocean Fisheries Agreement; LVFO: Lake Victoria Fisheries Organization. These maps are directly copied from its original source (Løbach, Petersson, Haberkon, & Mannini, 2020) and was not modified by the assessment authors. The maps are copyrighted under license CC BY-NC-SA 3.0 IGO. The designations employed and the presentation of material on the maps used in the assessment do not imply the expression of any opinion whatsoever on the part of IPBES concerning the legal status of any country, territory, city or area or of its authorities, or concerning the delimitation of its frontiers or boundaries. These maps have been prepared or used for the sole purpose of facilitating the assessment of the broad biogeographical areas represented therein and for purposes of representing scientific data spatially.Salmon Commission; IWC: International Whaling Commission; CACFISH: Central Asian and Caucuses Regional Fisheries and Aquaculture Commission; CTMFM: Joint Technical Commission of the Maritime Front; NAFO: Northwest Atlantic Fisheries Organization; NPFC: North Pacific Fisheries Commission; SEAFO: South East Atlantic Fisheries Organization; SPRFMO: South Pacific Regional Fisheries Management Organization; CCAMLR: Commission for the Conservation of Antarctic Marine Living Resources; GFCM: General Fisheries Commission for the Mediterranean; NEAFC: North-East Atlantic Fisheries Commission; RECOFI: Regional Commission for Fisheries; SIOFA: Southern Indian Ocean Fisheries Agreement; LVFO: Lake Victoria Fisheries Organization. *These maps are directly copied from its original source (Løbach, Petersson, Haberkon, & Mannini, 2020) and was not modified by the assessment authors. The maps are copyrighted under license CC BY-NC-SA 3.0 IGO. The designations employed and the presentation of material on the maps used in the assessment do not imply the expression of any opinion whatsoever on the part of IPBES concerning the legal status of any country, territory, city or area or of its authorities, or concerning the delimitation of its frontiers or boundaries. These maps have been prepared or used for the sole purpose of facilitating the assessment of the broad biogeographical areas represented therein and for purposes of representing scientific data spatially.*

collection protocols, observer coverage rates, and sufficient time-series to detect the response in absolute population abundance of long-lived species to this anthropogenic mortality source are also knowledge gaps in various global fisheries (Gilman *et al.*, 2020; Lewison, Crowder, Read, & Freeman, 2004; Musick, 1999a; Pérez Roda *et al.*, 2019).

United Nations Resolution 61/105 (UNGA, 2006) provides for responsible management of vulnerable marine ecosystems and non-target species as a legally binding instrument. It provides for collection of data on the impacts of fishing on vulnerable marine ecosystems and specific actions to protect them. Another important international protocol is the Agreement on Port State Measures (FAO, 2016a) aimed at preventing, deterring and eliminating illegal, unreported and unregulated fishing by preventing vessels engaged in illegal, unreported and unregulated fishing from using ports and landing their catches (FAO, 2021a).

Outside of formal international agreements, there have been many efforts to improve management both by non-governmental organizations and national governments. The 1990s were an era of greatly expanding concerns about overfishing, in many ways stimulated by the highly publicized collapse of the northern cod fishery in Canada (Finlayson, 1994; Kurlansky, 1997; Rice, Shelton, Rivard, Chouinard, & Fréchet, 2003). The Marine Stewardship Council was formed in 1997 with the goal to use market pressure to improve fisheries sustainability, and now is a major force in market access, particularly in Europe (MSC, 2021). Many environmental non-governmental organizations formed marine conservation divisions, and entirely new non-governmental organizations appeared with a focus on marine ecosystems. These were, to a great extent, funded by United States of America foundations with amounts up to 500 million United States dollars per year spent by environmental non-governmental organizations and foundations on marine conservation (Hilborn & Hilborn, 2019).

Since the 1990s national governments have expanded the science and management efforts through changes in legislations such as the United States of America Magnuson-Stevens act and revisions, and the creation of the Common Fisheries Policy in the European Union.

Finally, there has been increasing attention paid to consider impacts on fishing dependent coastal communities in almost all countries. As examples, Canada guarantees the first 90,000 tons of cod quota to small-scale inshore fishers, the United States of America allocates 8% of the allowable catch in the large industrial fisheries of the Bering Sea to local communities, and in Chile fishing cooperatives can apply for and be granted exclusive ownership of local inshore resources.

3.3.1.2 Status and trends in global marine capture fisheries

For the purposes of this assessment, in accordance with Chapter 1, fishing is defined as the harvest of entire organisms or parts of organisms that result in mortality of the aquatic animals, for example commercial fisheries or shark finning. Non-lethal fishing is defined as harvesting of entire or parts or products of organisms without intended mortality. Examples of non-lethal fishing include harvesting fish for the aquarium trade, catch and release fishing, or the extraction of blood from horseshoe crabs.

The status and trends of wild fish are estimated by a range of methods including scientific surveys, size or age distribution, catch per boat day and other estimates based on catch rate / fishing gear. A sophisticated method, known as "stock assessments", combines all these types of data to provide scientific estimates of the trend in abundance and harvest rate for fish stocks. The most robust approaches now involve multispecies and ecosystem-level assessments, an improvement over conventional single stock assessments, even though single stock assessments remain the dominant approach. Produced by national fisheries agencies and international regional fisheries management organizations, these scientific assessments are publicly available for roughly half of the global fish catch. Considerable effort in recent years has been towards increasing understanding of the status of stocks that produce the other half of global marine catches. This effort is ongoing.

The most cited stock status assessment comes from the state of world fisheries and aquaculture of the FAO(2020d), which uses a sample of roughly 500 fish stocks from around the world to describe the status of stocks. When scientific assessments are not available, expert knowledge is often used to make some sort of assessment. The material presented below follows this approach.

The status of fish stocks can be described in many ways. The most common approach is to compare the current abundance of the fish stock to target abundance, usually a target based on maximizing the long-term harvest, often called "maximum sustainable yield". In FAO terminology, stocks that are above this target level are called "underfished", stocks below the target are "overfished" and stocks with abundance close to the target are called "maximally sustainably fished." FAO uses the range 0.8 to 1.2 of the abundance as an indicator of maximum sustainable yield. Because fish stocks fluctuate naturally, sometimes over orders of magnitude of abundance, a better evaluation of the status of the fishery is to look at the fishing pressure relative to the targets. Fishing harder than the target rate is called "overfishing". Some assessments of stock status are based solely on the trends in catch. When catch declines it is assumed that the stock is in poor shape. Comparisons may also be made between the current abundance of fish stocks to estimates from before significant fishing began, which is most commonly done using various kinds of ecosystem models **(Figure 3.16)**.

A common misinterpretation of the above data is that stocks that are "maximally sustainably fished" are somehow being pushed to the limit and this is an undesirable state. In fact, "maximally sustainably fished" means that stocks are at an abundance level that will provide long-term maximum sustainable yield. Another misinterpretation is that stocks that are overfished are headed towards extinction or necessarily declining. "Overfished" simply means an abundance lower than would produce maximum sustainable yield, and many stocks remain at this level for decades; if fishing pressure is reduced these stocks can rebuild. Despite this common understanding, there is no agreed upon definition of what is overfished. The FAO defines overfished as the stock biomass being below 80% of the

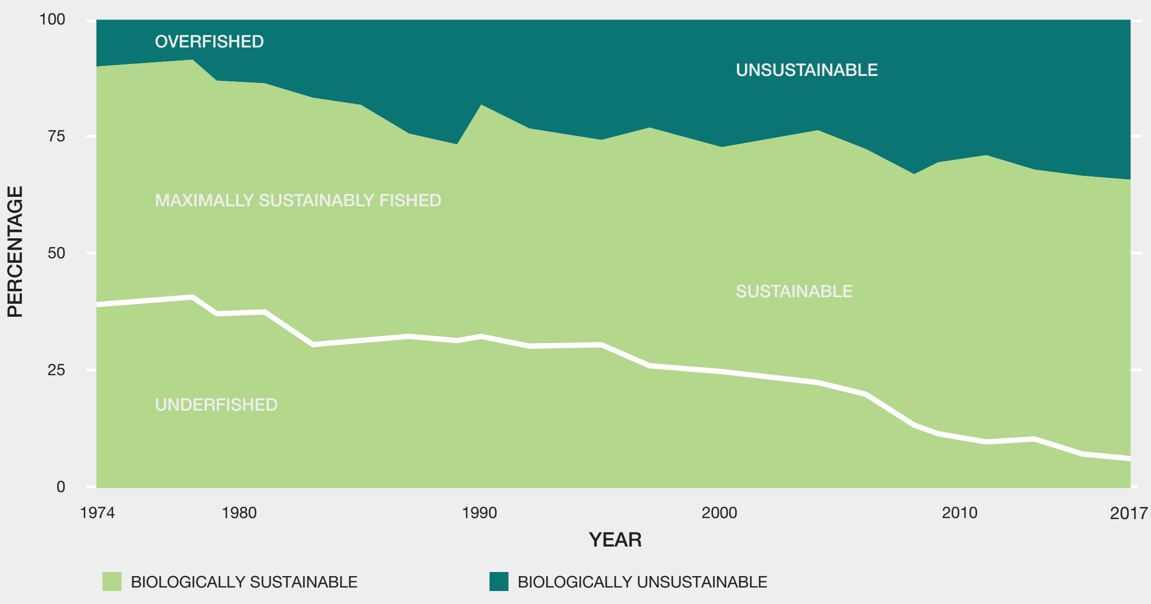


Figure 3 16 **Global trends in the state of the world's marine fish stocks, 1974–2017.**

Source: (FAO, 2020d) under license CC BY-NC-SA 3.0 IGO.

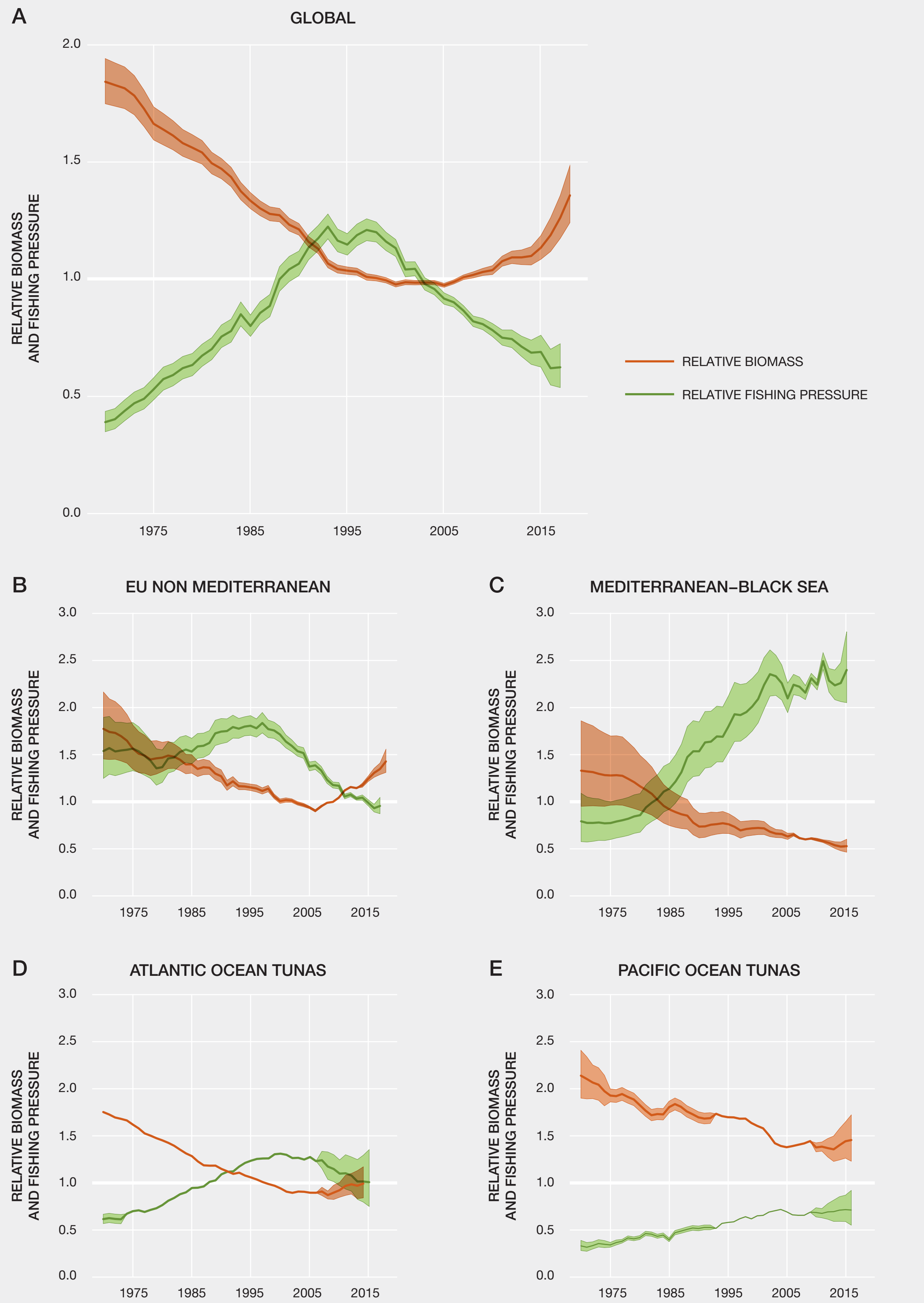


Figure 3 17 **Estimated abundance of global fish stocks 1970–2016.**

(A), European Union non-Mediterranean fish stocks **(B)**, Mediterranean-Black Sea fish stocks **(C)**, and Atlantic **(D)** and Pacific **(E)** ocean tuna fish stocks. Estimated abundance (B/BMSY -Biomass relative to the biomass that produces Maximum Sustainable

Yield- in orange) and fishing pressure (U/UMSY -Fishing pressure or mortality relative to the fraction of the population harvested- in green) are shown for the stocks that are scientifically assessed around the world from 1970 to 2016 – shaded area is the confidence intervals. The biomass and fishing pressure are scaled to the level that would produce maximum sustainable yield. See data management report for the figure at https://doi.org/10.5281/zenodo.6452917.

abundance that would produce maximum sustainable yield; the United States of America and New Zealand use a 50% cutoff, while many tunas' Regional Fisheries Management Organizations define overfished as being below the target level.

From a conservation perspective, stocks that are fished to very low abundance, where recovery is often very slow, are a concern due to lack of knowledge of potential recovery. Neubauer *et al.* (2013) conclude that "prolonged intense overexploitation, especially for collapsed stocks, not only delays rebuilding but also substantially increases the uncertainty in recovery times. Furthermore, when stocks become depleted, catch rates are lower and therefore the effort needed to catch a given volume of fish is higher and so is its environmental footprint.

For those fisheries that produce half of the world's marine catch for which good data is available, on average fish stocks are increasing because fishing pressure is lower than levels that would produce maximum long-term yield, and abundance is above target levels **(Figure 3.17)** (Hilborn *et al.*, 2020).

Figure 3.17A shows the estimated abundance (B/BMSY -Biomass relative to the biomass that produces Maximum Sustainable Yield- in orange), fishing pressure (U/UMSY -Fishing pressure or mortality relative to the fraction of the population harvested- in green), and catch (in blue) for the stocks that are scientifically assessed around the world from 1970 to 2016. The biomass and fishing pressure are scaled to the level that would produce maximum sustainable yield. Abundance declined from 1970 to 1995, then leveled off for 10 years and about 2005 began to increase. This is consistent with increased fishing pressure from 1970 to the mid 1990s, then declining pressure since that time **(Figure 3.17)**. When looking at different regions where there is good scientific understanding of stock status, one notes contrasting trends **(Figure 3.17 B-E)**. The European Union **(Figure 3.17B)**, Atlantic and Baltic stocks were already fished hard in 1970 and harvest rates increased up to about 1995, and then declined. Stocks were above target levels in 1970, declined to about 2005 and then began to increase. Mediterranean stocks **(Figure 3.17C)** have seen increasing fishing pressure since 1970 and declining abundance. Fishing pressure is far above target levels and abundance well below. One species specific estimate is included here **(Figure 3.17 D** & **E)**. Global tuna fisheries were not fully developed in 1970 and saw generally increasing fishing pressure and declining abundance until recent years when abundance leveled off at or above target levels. Atlantic tuna fisheries were fished harder and earlier than Pacific **(Figure 3.17)** that would produce maximum sustainable yield.

In the FAO's state of the world fisheries and aquaculture annual reports there are many stocks that are evaluated using expert knowledge because there is no scientific stock assessment. Melnychuk *et al.*, (2017) used an expert opinion survey of the 28 countries landing the most fish to determine the status of stocks and found that generally temperate stocks were considered to be in good shape while tropical stocks where not **(Figure 3.18)**.

Costello *et al.* (2012) attempted to estimate the status of the half of the world's fisheries that are not scientifically assessed and combine this with the data from assessed stocks to provide a global estimate of status. They grouped stocks into four classes; (i) large assessed (large industrial fisheries of the world where a scientific assessment of status and trends is performed); (ii) large unassessed, (iii) small assessed and (iv) small unassessed stocks. The trends estimate showed that the large stocks, both assessed and unassessed, are on average about target levels, but small assessed stocks were declining and small unassessed stocks were well below target levels **(Figure 3.19)**.

Rosenberg *et al.* (2018) combined four different methods (one being the Costello *et al.* (2012)) to estimate the status of unassessed stocks using an approach called ensemble modelling **(Figure 3.20)**. However, when the stock status was compared to the status for stocks that were scientifically assessed, the performance was rather poor and the ensemble method provided roughly similar status estimates both in regions where scientific assessment show stocks are in very poor shape such as the Mediterranean Sea, and also in regions where stocks are in very good shape such as the Northeast Pacific. Thus, we know the status of fish stocks which provide half of the world's catch – largely from the temperate North, and do not know the status of the other half of the global catch – largely from Southeast Asia.

Christensen *et al.* (2014) examined 200 marine food web models covering the period 1880 to 2007 and compared the change in abundance of different trophic levels of fish. They estimated that high trophic level fish had declined by 2/3 (to roughly the level that would produce maximum sustainable yield) while the far more numerous low trophic level species would have more than doubled.

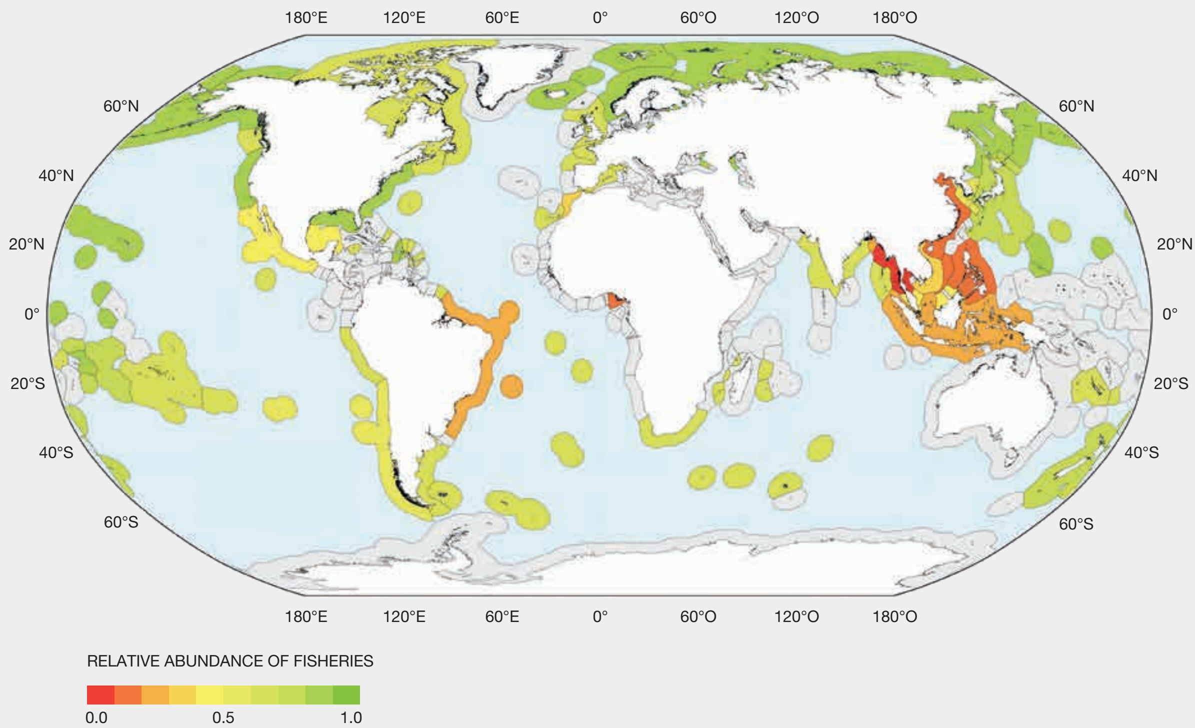


Figure 3 18 **Global abundance by coastline based on expert estimates.**

Green indicates experts believe that most stocks are at abundance consistent with long term maximum sustainable yield, red indicates few stocks are at that level.
Data from (Melnychuk *et al.*, 2017). See data management report for the figure at https://doi.org/10.5281/zenodo.6452953.

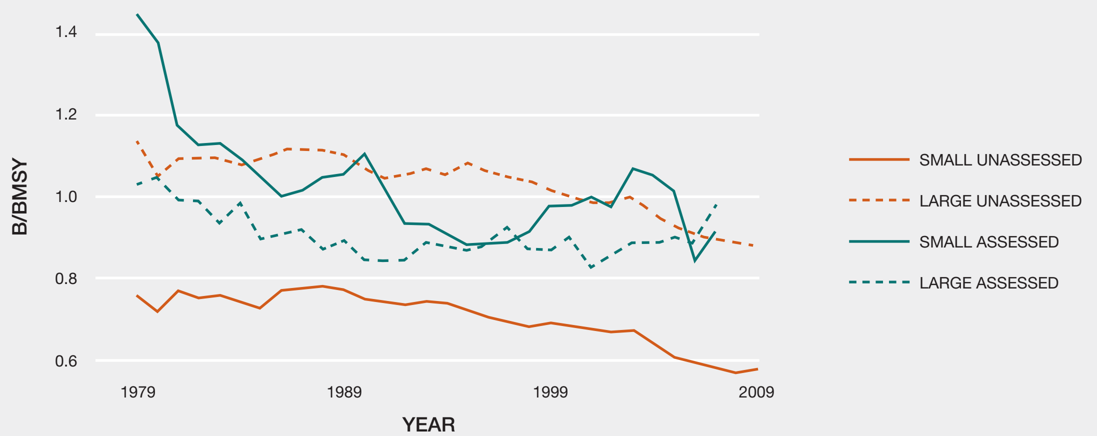


Figure 3 19 **Trend estimates for global large and small stocks.**

The black lines are for stocks scientifically assessed and are generally the same stocks as used in Hilborn *et al.*, 2020. The red lines are estimates of the trends for stocks not scientifically assessed. Source: (Costello *et al.*, 2012) © 2012, American Association for the Advancement of Science. CC-BY NC.

The performance of marine fisheries in terms of providing food security can be measured by comparing levels of sustainable yield at the current fishing pressure and if people fished at rates that would provide maximum sustainable yield. This is only available for the assessed fish stocks of the world. The status of assessed stocks is maintained on-line in the RAM Legacy Stock Assessment Database (Ricard, Minto, Jensen, & Baum, 2012). Using the data from assessed stocks and calculation of lost yield (Hilborn, 2018) the **Figure 3.21** shows the amount of potential yield that

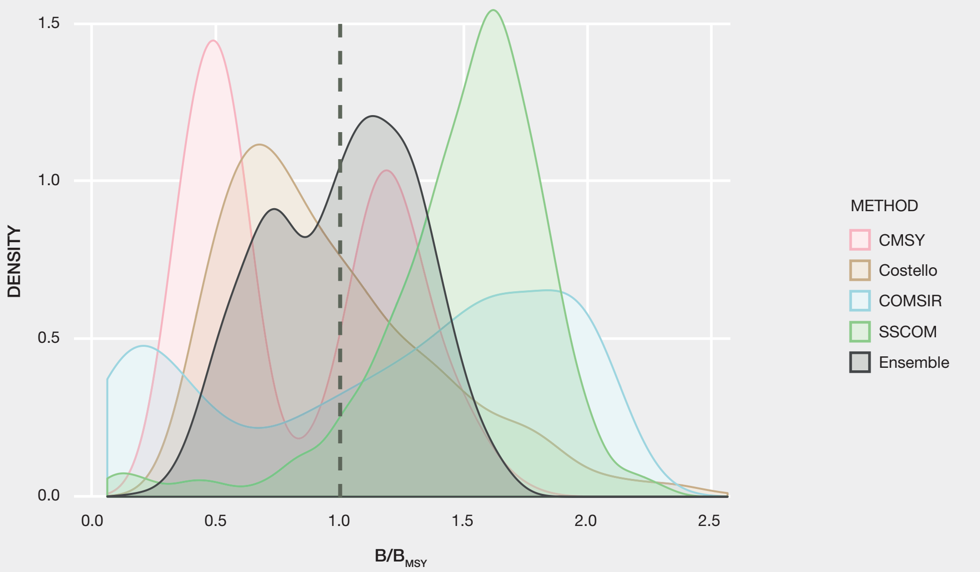


Figure 3 20 **Estimation of the status of unassessed stocks by several data poor methods.**

Abbreviations: B_{MSY}: Biomass that would support Maximum Sustainable Yield, C_{MSY}:Catch Biomass that would support Maximum Sustainable Yield, COMSIR: catch-only-model with sampling-importance resampling, SSCOM: state-space catch-only model. Source: (Rosenberg *et al.*, 2018) under license CC BY 4.0.

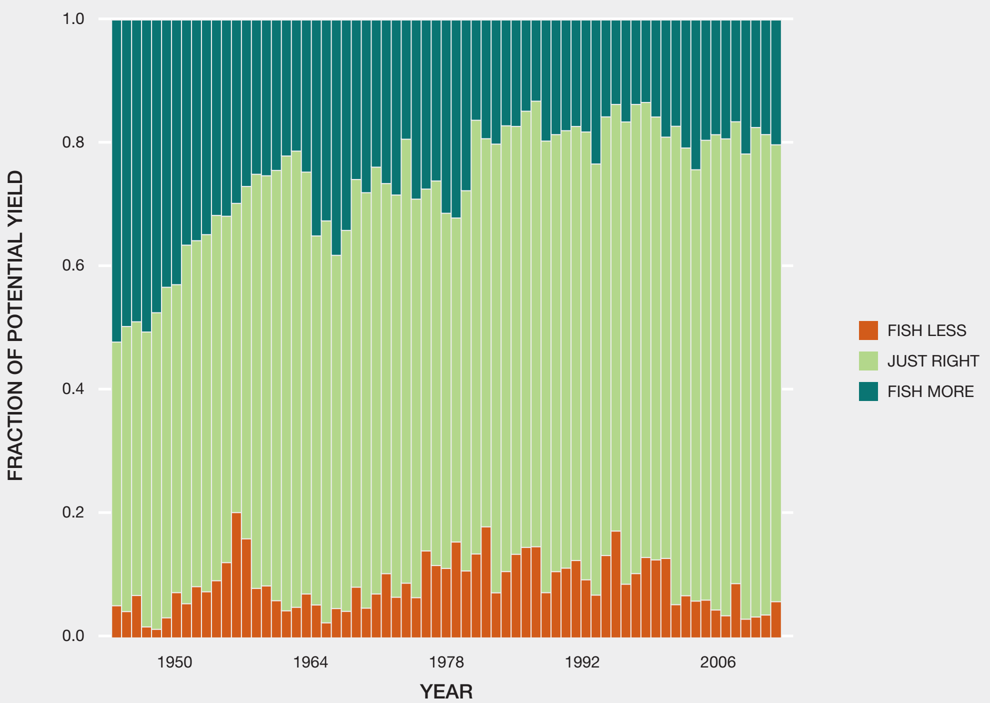


Figure 3 21 **The fraction of potential yield lost in each year by overfishing (red), and by fishing less than the Maximum Sustainable Yield (blue).**

Green shows the fraction of potential yield achieved at the fishing pressure for that year. See data management report for the figure at https://doi.org/10.5281/zenodo.6453019.

is lost by fishing too hard (red), or too little (blue) and how much of the potential yield is achieved at current fishing pressures (blue). It is estimated that in 1950 when the data began, roughly half of the potential yield was lost by low fishing pressure and there was little loss from fishing too hard (overfishing). The loss from overfishing rose to between 10% and 20% during the 1980s and 1990s and has now declined to about 5%. Potential increase in yield by fishing harder is now about 17%, and across these stocks the current fishing pattern is achieving about 73% of potential yield **(Figure 3.21)**. These calculations are based on the assumption that parameters that determine the productivity of fish stocks will remain unchanged at current estimated values. Note that fish production is not just a function of how hard people fish, but it depends on variable environmental conditions (temperature, food, ocean currents, etc.), including conditions affected by climate change.

3.3.1.3 Status and trends in selected fisheries

As no satisfactory global reviews were found in the literature, significant effort was invested in a systematic review of small-scale fisheries because of their importance for local communities. Due to high variability, the review of marine and inland small-scale fisheries was made by geographic region (see the data management report for Chapter 3 systematic literature review at https://doi.org/10.5281/zenodo.6452651). Three other sections dedicated to distinct fisheries were also developed: (i) small to medium pelagic or forage fish fisheries that are mainly coastal and provide about 25% of world capture fisheries; (ii) tuna and tuna-like fisheries, which are of high economic value and are widely spatially distributed from coastal regions to the high seas; (iii) industrial demersal fisheries in coastal areas, which are a complex set of heterogeneous fishing fleets using diverse fishing gears active within the exclusive economic zones of coastal countries. When necessary, for taxonomic groups of special concern, we added information on their status and trends in dedicated boxes (e.g., **Box 3.2**).

3.3.1.4 Small-scale fisheries

Small-scale fisheries are strongly anchored in local communities where fisheries represent a way of life (FAO, 2015). Despite their importance, small-scale fisheries around the world are facing major challenges from the effects of global change, e.g., climate change, urbanization, industrialization, aquaculture intensification, and large-scale fisheries (Berkes, 2015; Chuenpagdee, 2011). Ongoing threats to small-scale fisheries affect entire production systems (harvest, processing, retail and transport) and create vulnerabilities that have no easy solution (Chuenpagdee, 2011; Jentoft & Chuenpagdee, 2009). In many cases, these challenges have placed the livelihoods, economy, food security, values and identity, and the viability of small-scale fisheries communities at risk (Bavinck, Jentoft, & Scholtens, 2018; Bundy *et al.*, 2016; Jentoft & Chuenpagdee, 2015; Jentoft & Eide, 2011; Nayak & Armitage, 2018). An estimated 5.8 million fishers in the world who earn less than 1 United States dollars per day (FAO, 2014d). Ommer *et al.* (2007) characterize these large-scale, globalized processes as a crisis in social-ecological 'health', with dire consequences on small-scale fisheries communities.

The COVID-19 pandemic will affect many small-scale fisheries and coastal communities worldwide, especially those more vulnerable, mainly through reduced (or closure of) markets, decreases in revenues from tourism, increases in health risks to fishers and traders and increased occurrence of illegal fishing due to lack of enforcement. Mitigation of these factors would likely require institutions to provide short- and long-term responses (N. J. Bennett *et al.*, 2020). There can be some positive outcomes from the pandemic crisis, including enhanced local cooperation among fishing communities and other institutions, valorization of local markets, food sharing and some recovery of fishing resources (N. J. Bennett *et al.*, 2020).

The state of inland capture-fishery resources that includes small-scale inland fisheries is more difficult to monitor (Welcomme, 2011) for a number of reasons, including the diffused character of the practice due to: (i) large numbers of people being involved in the seasonal and subsistence nature of fisheries activities; (ii) much of the catch being consumed locally or traded informally; and (iii) fisher populations being greatly affected by activities other than fishing, including stocking from aquaculture and diversion of water for other uses such as agriculture and hydroelectric development (FAO, 2012c).

This section is based on a comprehensive review of 350 studies on small-scale fisheries from 107 countries worldwide **(Figure 3.22)**. With regard to ecological sustainability, 39 studies indicate sustainable fisheries but almost half the studies (#165) indicate unsustainability. Whereas fisheries reported by 129 studies were considered to be partially sustainable; a few studies (#16) do not assess ecological sustainability but include some accounts on economic or social sustainability. Most of the reviewed literature on small-scale fisheries addresses the use of fish as food and feed, and is presented in detail below by major world regions. Other uses for fish are also mentioned in some regions (see the data management report for Chapter 3 systematic literature review at https://doi.org/10.5281/zenodo.6452651). This review supports the text below, considering the available evidence from most of the revised studies (for details on the reviewed studies, see the data management report for Chapter 3 systematic literature review at https://doi.org/10.5281/zenodo.6452651).

Box 3 2 Status and trends of sharks, rays, and chimaeras: implications for species, the environment, and people.

There are approximately 1,250 species of sharks and rays found throughout the world's marine, and some freshwater, habitats. Sharks and rays are relatively large-bodied predators and, hence, are both highly susceptible to a wide range of fishing gears (predominantly trawls, longlines, gill and tangle nets) and highly sensitive to fishing mortality because of their long generation lengths and low fecundity resulting in very low maximum population growth rates and low density-dependent compensation (Forrest & Walters, 2009; Eric Gilman *et al.*, 2008; Pardo, Kindsvater, Reynolds, & Dulvy, 2016). Consequently, they are highly vulnerable to overfishing compared to the teleost fishes they are caught alongside and are particularly prone to disappearing prior to adequate monitoring (Myers & Worm, 2005; Yan *et al.*, 2021).

Global shark and ray catches reported to FAO rose to a peak in 2003 and declined at least 17% thereafter, likely due to overfishing (Davidson, Krawchuk, & Dulvy, 2016; Dent & Clarke, 2015). However, the global catch is underestimated and is likely to be two-to-four times greater (Clarke *et al.*, 2006). Based on these FAO data and accounting for discards and illegal, unreported, and unregulated fishing, it is possible that 63–273 million individuals were captured in the early 2000s (Boris Worm *et al.*, 2013). Only 4% of the global estimated catch is managed sustainably, based on 65 fisheries stock assessments from 47 species from Canada, United States of America, Australia, and New Zealand (Simpfendorfer & Dulvy, 2017). Catch estimates of unassessed data-poor fisheries show that large coastal sharks have been very unsustainably fished since 1975 (B/Bmsy – Biomass relative to the biomass that produces Maximum Sustainable Yield < 0.5) (Costello *et al.*, 2012). Consequently, steep regional declines of coastal sharks have been documented (Ferretti, Osio, Jenkins, Rosenberg, & Lotze, 2013; MacNeil *et al.*, 2020). Oceanic sharks and rays have limited spatial refuge from fisheries (Queiroz *et al.*, 2019) and declined by 71% since 1970 due to an 18-fold increase in relative fishing pressure (Pacoureau *et al.*, 2021). Sharks and rays from the tropical and subtropical coastal seas are currently at higher risk (Dulvy *et al.* 2021).

The International Union for Conservation of Nature Red List provides a framework for integrating disparate data sources ranging from historical ecology, to catch data and stock assessments (International Union for Conservation of Nature Standards and Petitions Committee, 2019; Mace *et al.*, 2008; Sherley *et al.*, 2020, p. 20). These comprehensive global assessments of sharks and rays offer a unique opportunity to calculate Living Plant and Red List indices to track progress toward the Convention on Biological Diversity and Sustainable Development Goals (Pacoureau *et al.*, 2021; Walls & Dulvy, 2021).

Shark and ray extinction risk has been rising over the past half century (Pacoureau *et al.*, 2021, Walls and Dulvy, 2021). Now, one one-third (391 of 1,199; 32.5%) of sharks and rays are classified as threatened (vulnerable, endangered, or critically endangered) (Dulvy *et al.*, 2021). Assuming the 155 data deficient species are threatened in the same proportion to the other species then an estimated 449 species are threatened (37.5%, range 32.6–45.5%). Three species are critically endangered (possibly extinct), because they have not been recorded for over 80 years but there have been insufficient surveys to confirm their extinction (Dulvy *et al.*, 2021). A further eight species are regionally extinct in one or more countries and there have been at least 28 local extinctions (Dulvy *et al.*, 2014, 2021). The shark and ray extinction rate of 25 E/MSY (extinction per million spesies-year) is 25–250 times greater than the background fossil record extinction rate and 2.5 times greater than the proposed target rate of 10 E/MSY (extinction per million spesies-year) over the next century (Rounsevell *et al.*, 2020). Nearly all (99.6%) species are taken incidentally, but are valuable and are retained for food: half (51.5%) for human consumption of the meat only, with remaining species used for food in combination with the production of animal feed, skins, and liver oil (Dulvy *et al.*, 2021). The International Union for Conservation of Nature classification scheme does not record shark and ray fins or devil ray gill plates (Mobulidae), but these significant trades are subject to increasing international regulation (Cardeñosa, Quinlan, Shea, & Chapman, 2018; Friedman *et al.*, 2018). The global value of the shark and ray trade is worth 4.1 billion United States dollars, with the meat trade (2.6 billion United States dollars) exceeding the value of the global fin trade (1.5 billion United States dollars) (Niedermüller *et al.*, 2021).

Widespread overfishing of sharks and rays will likely have profound consequences for the environment and people. The depletion and loss of sharks and rays, particularly in the tropics, does not bode well for the livelihoods of many coastal human populations, dependent on their meat and products for food and income (Booth, Squires, & Milner-Gulland, 2019; Seidu *et al.*, 2022). Indeed, the depletion of sharks and rays reflects increasing evidence that the target teleost fisheries are overfished in South America, Africa, and Southeast Asia (Dyhia Belhabib, Greer, & Pauly, 2018; Lam & Pauly, 2019).

A systematic literature review on small-scale fisheries was undertaken based on literature obtained through various combinations of a set of keywords: fisheries, sustainable, sustainability, small-scale, coastal, freshwater, catch, trend, success, local knowledge, use, fishers, co-management, increasing, review, catches, fish, and ecological. These keywords were selected to get a manageable number of hits (literature) to assess and to direct the search results to those articles analyzing sustainable fisheries, or at least to those showing trends of an increase in catches. The database SCOPUS was used for articles from the last 20 years (since 2000), which initially retrieved a total of 1635 articles. A complementary search was made on Google Scholar using a subset of these keywords. However, due to the large amount

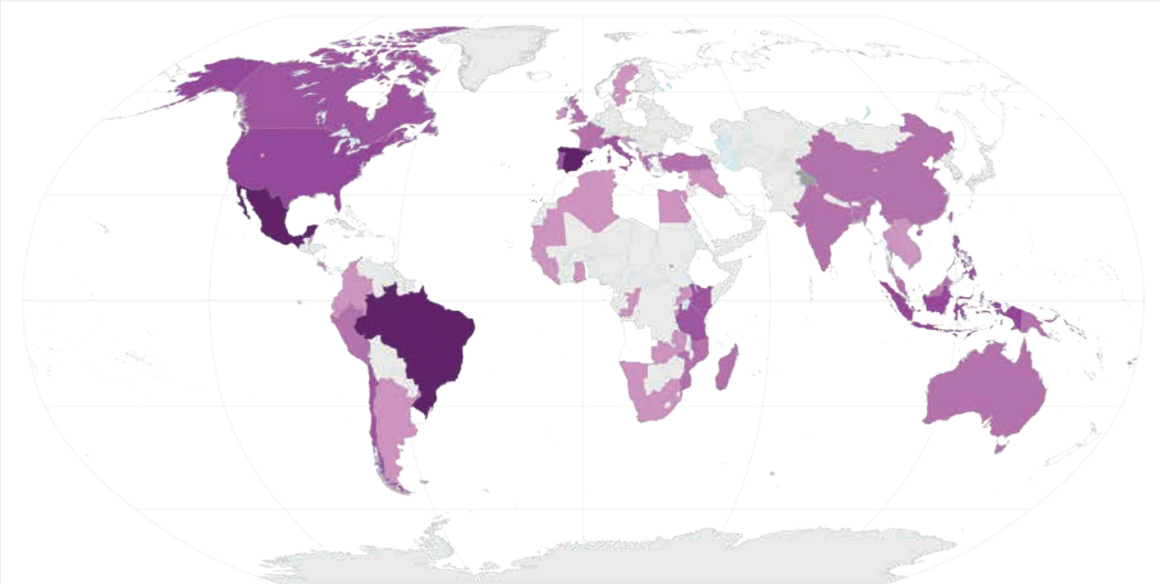


Figure 3 22 **Global distribution of the 350 reviewed studies on small-scale fisheries among 107 countries (countries in gray means that no study was included).**

See data management report for the figure at https://doi.org/10.5281/zenodo.6453056.

of literature retrieved (34300 hits), only the first 200 hits were reviewed on Google Scholar, including some of the more recent articles from the last 10 years. A total of 447 articles on small-scale fisheries were selected after an initial screening, including only articles that reported some data on fisheries, preferably trends and some kind of indicator, such as abundance, size or catch per unit of effort, or fishing effort among others. Articles addressing details of management or policy options which did not include data, or theoretical approaches and effects from drivers, such as climate change, pollution, or development projects, were not included.

These 447 articles were sorted by major regions and the case studies on small-scale fisheries were selected from these (see the data management report for Chapter 3 systematic literature review at https://doi.org/10.5281/zenodo.6452651). This literature review was complemented with relevant articles inserted by the authors from their personal libraries, by suggested articles from internal and external reviewers, and through cross-reference from the selected articles. Our review did not retrieve a large number of articles dealing with uses other than food (ornamental, medicinal, etc.) and those addressing social and economic dimensions of sustainability in small-scale fisheries.

The selected studies were sorted across a gradient of ecological sustainability, ranging from fully sustainable (exploited populations stable, no habitat damage, no ecological filters or shifts in the composition of exploited species) to unsustainable (exploited populations declining or overfished). Intermediate or partial sustainability included situations in which current exploited populations are stable, but some higher valued species were depleted or extinct, which are considered here as ecological filters (see also section 3.3.1.4.2), or the fishing practice has caused habitat damage or bycatch. Fisheries lacking data on temporal trends to clearly indicate sustainable catches were also allocated to these partially sustainable categories (for details on the reviewed studies, see the data management report for Chapter 3 systematic literature review at https://doi.org/10.5281/zenodo.6452651).

A major challenge in evaluating the sustainability of small-scale fisheries is the lack of data on catches and measures of exploited stocks (size, proportion of juveniles caught, etc.), especially over broader spatial or temporal scales. Nevertheless, participatory research in collaboration with fishers and analyses of the fishers' knowledge about fishing resources have contributed evidence to assess patterns of sustainability, catches and fishing effort.

Relatively few studies have evaluated the economic sustainability of small-scale fisheries. A review on global marine fisheries indicates that well-managed and locally supported small-scale fisheries could be a more sustainable option to provide employment and food than the current subsidy-driven industrial fisheries, which may increase effort in spite of declining fishing resources (Zeller & Pauly, 2019). However, conventional economic models that have been applied to assess fisheries economic viability may not be appropriate to small-scale fisheries, which need inclusion of social and environmental variables to conduct economic viability analyses that go beyond profit maximization (Schuhbauer & Sumaila, 2016).

The literature search retrieved 49 studies of global scope, which encompass multiple countries from more than one of the broad regions defined here, of which 18 studies were included in this review (see the data management report for Chapter 3 systematic literature review at https://doi.org/10.5281/zenodo.6452651). Among these, studies 15 address coastal fisheries, two address inland fisheries and two include both coastal and inland. These studies usually have a broad coverage in space or time, grouping data from many regions and communities and sometimes showing long time series of 50 up to 600 years (see the data management report for Chapter 3 systematic literature review at https://doi.org/10.5281/zenodo.6452651). One of these studies, which brings data for over 1,900 coastal indigenous communities around the world, representing 27 million people across 87 countries, claims that sustainability depends on increased recognition and directed research regarding the marine knowledge and resource needs of indigenous peoples, whose needs must be explicitly incorporated into management policies (Cisneros-Montemayor, Pauly, Weatherdon, & Ota, 2016).

Other studies point to the potential overfishing of marine invertebrates (including cephalopods, shellfish, lobsters, crabs, sea cucumbers) estimating that, in 2004, 34% of invertebrate fisheries were over-exploited, collapsed, or closed, as global invertebrate catches have increased 6-fold (Anderson, Flemming, Watson, & Lotze, 2011). This problem is especially severe for sea cucumber fisheries, 81% of which show population declines from overfishing, and 35% had declines in the average harvested body size. Harvesters moved from near- to off-shore regions in 51% of cases and from high- to low-value species in 76% of these fisheries (Anderson, Flemming, Watson, & Lotze, 2011). Similarly, a global survey indicates that sawfishes (family Pristidae) have been heavily affected by intense harvesting and habitat degradation and these sawfish are now extinct in 55 of the 90 nations where they originally occurred (Yan *et al.*, 2021).

A study comparing the fisheries in Florida (Atlantic) and Hawaii (Pacific) over a period of 600 years indicated that, although fishing had been sustainable in Hawaii for 400 years, landings have declined and some species are recorded as overexploited in both the study regions (Mcclenachan & Kittinger, 2013). A study reviewing context and attributes of co-management initiatives in small-scale fisheries concludes that more research is needed to discern when co-management initiatives can transform pre-existing conflicts, challenge power asymmetries and distribute benefits more equitably (d'Armengol, Prieto Castillo, Ruiz-Mallén, & Corbera, 2018). However, another study indicates that fishers perceived improved livelihoods and compliance in co-managed sites, thus evidencing contributions of co-management to improve social sustainability (Cinner *et al.*, 2012).

EUROPE AND CENTRAL ASIA

Out of the 56 papers reviewed for Europe (see the data management report for Chapter 3 systematic literature review at https://doi.org/10.5281/zenodo.6452651). The vast majority cover mainly the coastal and marine/oceanic fisheries in Europe or in European archipelagos in the Atlantic Ocean, the Mediterranean Sea (and its internal seas, like the Adriatic, the Aegean, the Marmara) or in the Black Sea. Ocean or marine small-scale fisheries is discussed in 48 papers, whereas only a small number of those (eight) investigated the European inland small-scale fisheries. A majority of these papers focused on Iberian freshwater fishing (Antunes, Cobo, & Araújo, 2015; Braga, Pereira, Morgado, Soares, & Azeiteiro, 2019; Marcos, Torres, López-Capel, & Pérez-Ruzafa, 2015; Maynou, Martínez-Baños, Demestre, & Franquesa, 2014), although there are other very important fishing practices, such as the trout fisheries, taking place in many different countries of the region (Shephard *et al.*, 2019).

The vast majority of the papers discuss the exploitation of fish species, but other organisms are also discussed, including a large diversity of targets in single fishing systems such as crustaceans and mollusks (Alonso-Fernández *et al.*, 2019; Antunes *et al.*, 2015; Azzurro *et al.*, 2019; Battaglia *et al.*, 2017; Carrà, Monaco, & Peri, 2017; Colloca, Scarcella, & Libralato, 2017; Corral & Manrique de Lara, 2017; Fabio, Silvia, Paolo, & Anelli Monti, 2016; Grati *et al.*, 2018; Guyader *et al.*, 2013; Palmer *et al.*, 2017; Quetglas *et al.*, 2017). A small number of papers also cover exploitation of crustaceans (Carvalho, Vasconcelos, Piló, Pereira, & Gaspar, 2017; Rivera *et al.*, 2016; Rivera *et al.*, 2017), mollusks (Baeta, Breton, Ubach, & Ariza, 2018; Duncan, Brand, Strand, & Foucher, 2016; Öndes, Kaiser, & Güçlüsoy, 2020; Pereira, Vasconcelos, Moreno, & Gaspar, 2019; Silva *et al.*, 2019; Szostek, Murray, Bell, & Kaiser, 2017), benthic invertebrates (Bastari, Beccacece, Ferretti, Micheli, & Cerrano, 2017; Fourt, Faget, Dailianis, Koutsoubas, & Pérez, 2020; Pita *et al.*, 2019) and even sea mammals (Maynou *et al.*, 2011). The diversity of topics is a sign of the high diversity of fishing practices, technologies and techniques present in the European small-scale fishing.

Contrary to the pattern observed in other regions, the literature on fishing rarely mentions lack of data on European small-scale fisheries. Still, lack of data does remain a concern in a number of cases including inaccuracy, large underestimation of parameters, undeclared information, and lack of stock assessment analysis for some fishing systems. Contrary to what is observed in the literature about the small-scale fisheries in other regions, no major cases of illegal, unreported and unregulated(Colloca *et al.*, 2017; Ulman *et al.*, 2013, 2015a) activities are focused upon in these studies (Colloca *et al.*, 2017; Dinesen *et al.*, 2019; Hornborg & Främberg, 2019; Marcos *et al.*, 2015).

Small scale fishing is an economically, socially, and culturally significant practice throughout Europe. It is well established that small-scale fishing plays an important role in many national economies (Guyader *et al.*, 2013; Lloret *et al.*, 2018), and almost 80% of the European fishing fleet belongs to small-scale fisheries (Quetglas *et al.*, 2016). Sometimes, in general terms, this fishing is more profitable than the large-scale fishing industry since costs are lower and catches are similar (Almeida, Vaz, Cabral, & Ziegler, 2014). In some parts, the increase in the tourism industry and, less conspicuously, the increase in recreational fishing, led to a slight expansion in local economies (Marengo, Culioli, Santoni, Marchand, & Durieux, 2015) and generated new incomes and additional revenues in the form of concessions and permits (Antunes *et al.*, 2015). On the other hand, it is also well established in the literature that small-scale European fishing is threatened by the competition among different uses of aquatic resources and by decreasing profitability, detected in almost all systems evaluated (Maynou *et al.*, 2014).

When European small-scale fishing systems are analyzed, the majority of the papers describe activities that are still profitable (Roditi & Vafidis, 2019; Ünal & Franquesa, 2010), but that these profits dropped consistently in recent decades (Maynou *et al.*, 2014; Pita *et al.*, 2019; Quetglas *et al.*, 2016). The reduction in market values and revenues is causing a marked change in local economies and in employment rates (Ünal & Franquesa, 2010), with serious impacts on traditional fishing communities. It is estimated that the European small-scale fisheries dropped from 30-50% in terms of income over this time period (Lloret *et al.*, 2018). But in most of these cases small-scale fisheries continues as an important source of employment (Baeta *et al.*, 2018) even if fishers have to work additional jobs to maintain their livelihoods (Braga *et al.*, 2019; Pereira, Vasconcelos, Moreno, & Gaspar, 2019b). The drop in profits, revenues and wages are not only due to overexploitation of stocks, the decrease in market values or to climate change. Competition is also increasing due to the introduction of industrial and recreational fishing, which have caused major reductions to commercial small-scale fisheries landings and profits (Marengo *et al.*, 2015; Maynou *et al.*, 2013).

European small-scale fishing the literature also highlights the exploitation of economically important and profitable high-valued stocks (Grati *et al.*, 2018), with particular emphasis on scallops (Duncan *et al.*, 2016; Szostek, Murray, Bell, & Kaiser, 2017), large demersal fish species (Quetglas *et al.*, 2017), octopuses (Silva *et al.*, 2019), carps (Hornborg & Främberg, 2019), cod (Dinesen *et al.*, 2019), barnacles (Carvalho *et al.*, 2017) and salmon (Antunes *et al.*, 2015). Some of the additional profits can also come with the opportunity or possibility to exploit "labels of topicality" (Dinesen *et al.*, 2019; Sartor *et al.*, 2019). There are increasing trends in the demand of international market for these items, and their market values may pose a threat to their stocks (Antunes *et al.*, 2015; Lloret *et al.*, 2018). Most of these high-valued stocks were severely overexploited for a long time, and some of them are only now recovering after the introduction of more careful management measures (Rivera *et al.*, 2016; Rivera *et al.*, 2017).

The strong economic and technological changes experienced in the last 60 or 80 years are accompanied by consistent social and cultural importance of these practices (Carvalho *et al.*, 2017). Most of the local populations show a marked dependence on small-scale fisheries, in terms of food security, for the maintenance of local employment and for the resilience of cultural heritage (Braga *et al.*, 2019; Colloca *et al.*, 2017; Grati *et al.*, 2018; Pereira *et al.*, 2019; Ünal & Franquesa, 2010). In some European countries, more than 50% of the fishers are linked to one of the small-scale fishing systems in place (Antunes *et al.*, 2015; Quetglas *et al.*, 2016; Sartor *et al.*, 2019; Silva *et al.*, 2019). Small-scale fisheries employ twenty-four times more fishers than large-scale fishing (Leleu *et al.*, 2014).

The history of more traditional fishing systems goes back thousands of years (Antunes *et al.*, 2015; Marcos *et al.*, 2015). This strengthens cultural and historical bonds, and provides ongoing social meaning for indigenous people and local communities (Guyader *et al.*, 2013). With the technological changes in the last 50 to 60 years, the efficiency of the fishing systems has (Alonso-Fernández *et al.*, 2019; Pita *et al.*, 2019; Quetglas *et al.*, 2017; Ünal & Franquesa, 2010). Besides unemployment, other problems such as mechanization (Lloret *et al.*, 2018).

While some unemployed fishers searched for new jobs, better wages or other sources of income (Maynou *et al.*, 2013), many families had to close down business and sell their fishing equipment and boats to larger companies (Dinesen *et al.*, 2019). The collapse of fishing systems and the overexploitation of stocks created new social contexts which demanded new and stricter management rules and improved governance, also seen as means to avoid social conflict (Marengo *et al.*, 2015). These needs were partially met with the official management measures adopted in many areas, with distinct levels of success. Apparently, the recovery of social recognition of those engaged in this practice and the relevance of the small-scale fisheries was also an outcome of successful management initiatives at some places (Carvalho *et al.*, 2017).

AFRICA

From the initial selection of 63 papers, this evaluation on African small-scale fisheries is based on 51 papers covering mainly the coastal and marine/oceanic small-scale fishing, which was the subject of approximately 40 papers (see the data management report for Chapter 3 systematic literature review at https://doi.org/10.5281/zenodo.6452651). Despite the importance of established fisheries in tropical and subtropical African rivers, the reviewed literature focused on inland small-scale fishing in the great African lakes and small rivers. The fishing practices in African great lakes was studied in eight papers (Bulengela, Onyango, Brehm, Staehr, & Sweke, 2019; Hara & Njaya, 2015; Jamu, Banda, Njaya, & Hecky, 2011; Kolding, Béné, & Bavinck, 2014; Mgana *et al.*, 2019; Mkuna & Baiyegunhi, 2019a, 2019b; van der Knaap & Ligtvoet, 2010). Similar analysis for fishing practices in some African smaller lakes was published in three studies (Kininmonth *et al.*, 2017; Obegi *et al.*, 2020; Tefera, Zerihun, & Wolde-Meskel, 2019), and there were a few examples of small river fishing in South Africa and Egypt (McCafferty, Ellender, Weyl, & Britz, 2012; Samy-Kamal, 2015). The majority of the papers describe fishing for fish species, but a small number also include fishing for crustaceans (Bush *et al.*, 2017; Cochrane, Eggers, & Sauer, 2020; Fulanda, Ohtomi, Mueni, & Kimani, 2011; Le Manach *et al.*, 2012; Le Manacha, Goughb, Humberb, Harperc, & Zellerc, 2011; Mirera, Ochiewo, Munyi, & Muriuki, 2013).

There is scarce published data about African small-scale fishing. However, it is well established that many peoples rely on small-scale fishing for their subsistence and livelihoods throughout Africa (Musembi, Fulanda, Kairo, & Githaiga, 2019). Absence or inadequacy of data, under-estimates, and lack of stock assessment analysis were consistently mentioned by almost all papers reviewed. Those data sets supported by the FAO in many countries are usually underestimates since they are based only on landings, not considering data from illegal, unreported and unregulated fishing. Some papers present a reconstruction of data series, which attempted to include illegal, unreported and unregulated catch (Barnes-Mauthe, Oleson, & Zafindrasilivonona, 2013; Jacquet, Fox, Motta, Ngusaru, & Zeller, 2010; Le Manach *et al.*, 2012; Seto *et al.*, 2017).

Only one third of the papers reviewed presented any socioeconomic evaluation of fishing sustainability across the continent, and only two papers were focused on this topic. All other social evaluations demonstrated the high level of dependence of local communities on fishing practices (Belhabib, Greer, & Pauly, 2018; Bush *et al.*, 2017).

Formal economic review shows that market prices either kept stable or increased in the last 60 years. This is an important factor to explain the increase in fishing effort and overexploitation of most stocks. Pressure from international markets for some high value species for exportation also added pressure on the stocks. This increased pressure led to increased competition between international fleets and local boats and sometimes conflict (Belhabib *et al.*, 2016; Seto *et al.*, 2017).

LATIN AMERICA

For the purpose of this assessment, Latin America includes the countries in South and Central America, Mexico and Caribbean Islands based on the similarities in their small-scale fisheries and social-ecological characteristics. This review is based on 107 articles (see the data management report for Chapter 3 systematic literature review at https://doi.org/10.5281/zenodo.6452651) from the review sources and those added by the assessment authors. These studies address coastal and inland small-scale fisheries in 15 countries with large numbers of studies from Brazil (55) and Mexico (20), which may reflect a larger number of fisheries scientists working in these countries rather than greater small-scale fisheries activity there. A selection of the studies provides international comparisons (Defeo *et al.*, 2016; Maldonado, Lopes, Fernández, Alcala, & Sumalia, 2017) or continental level comparisons (Brotz *et al.*, 2017). Most studies (78) addressed the use of finfish but also reported on sharks, shellfish, lobsters, octopus, crabs and jellyfish. More than two thirds of the studies (78) deal with coastal fisheries with fewer (29) studies addressing inland fisheries, and most of these (25) were in the Amazon region (for details on the reviewed studies, (see the data management report for Chapter 3 systematic literature review at https://doi.org/10.5281/zenodo.6452651). More than half of the studies (56) had short time series ranging from 1 to 15 years of data collection. A few studies (17) included long range data series of 50 years or more, some of which included indigenous and local knowledge through interviews with seniors.

As expected, those well managed and ecologically sustainable fisheries were also considered to be economically sustainable and showed improved economic indicators such as increased prices or profits from sales of managed resources. These offset eventual decreases in total catches due to management measures, as observed among coastal invertebrate fisheries under territorial rights in Chile and Mexico (Álvarez, Espejel, Bocco, Cariño, & Seingier, 2018; De la Cruz-González, Patiño-Valencia, Luna-Raya, & Cisneros-Montemayor, 2018; Defeo *et al.*, 2016; Gelcich *et al.*, 2010). Nevertheless, the territorial users' rights fisheries management in Chile also caused economic shortages through the collapse of a clam fishery and reduced economic opportunities to fishers not engaged in territorial users' rights fisheries management, who relied on depleted open access areas (Aburto & Stotz, 2013; Garmendia, Subida, Aguilar, & Fernández, 2021).

The positive economic effects observed in coastal shellfish fisheries were also observed in the pirarucu co-managed fishery in the Brazilian Amazonian rivers (Campos-Silva & Peres, 2016; Castello, Viana, Watkins, Pinedo-Vasquez, & Luzadis, 2009), where increased revenues from co-management led to further social benefits, through gender equality and improved income for women (Freitas, Espírito-Santo, Campos-Silva, Peres, & Lopes, 2020). Other studies on coastal small-scale fisheries employed economic modelling, which indicate that a fishery of octopus *(Octopus maya)* in Mexico would be more sustainable under current management, as economic performance does not improve under alternative management scenarios (Duarte, Hernández-Flores, Salas, & Seijo, 2018a). Similarly, the recovery of shellfish through co-management in a Mexican community was shown to be profitable under two of four estimated future economic scenarios (Palacios-Abrantes, Herrera-Correal, Rodríguez, Brunkow, & Molina, 2018).

One study on fisheries in French Guiana evaluated various sustainability indicators, which suggested average sustainability for ecological, economic and social dimensions. Smaller fishing fleets were considered to be more sustainable (Cissé, Blanchard, & Guyader, 2014). Several coastal small-scale fisheries considered to be less economically sustainable were the fishing of spawning aggregations of reef fish in Mexico (Erisman *et al.*, 2010) and the shark fishing in Mexico (Martínez-Candelas, Pérez-Jiménez, Espinoza-Tenorio, McClenachan, & Méndez-Loeza, 2020) and Brazil (Martins *et al.*, 2018). The decline in the economic sustainability of shark fishing is attributed to decreases in shark fishing activity, revenues and profits from shark fins.

Other economic problems refer to inequalities in the distribution of profits among crew members and boat owners (De Figueiredo Silva, Camargo, & Estupiñán, 2012), low prices paid to fishers by the middlemen, the concentration of profits in large private companies (Gamboa-Álvarez, López-Rocha, Poot-López, Aguilar-Perera, & Villegas-Hernández, 2020a; Jimenez, Barboza, Amaral, & Lucena Frédou, 2019) and increasing costs related to fishing operations such as fuel to reach more distant fishing grounds (Daw, 2008). A study with crab gatherers in the Brazilian Amazonian coast considers the fishery ecologically sustainable (catches and sizes of crabs did not change), but not economically and socially sustainable. The relative revenue for fishers also declined, which sometimes led to social conflicts (Glaser & Diele, 2004).

The social aspects of small-scale fisheries were addressed by only 19 of 78 studies on coastal small-scale fisheries and 5 of 29 studies on inland small-scale fisheries (see the data management report for Chapter 3 systematic literature review at https://doi.org/10.5281/zenodo.6452651). Some of the territorial co-management coastal fisheries of invertebrates, mainly in Chile and Mexico, show social benefits such as improved perceptions among fishers about the fishery, more time available to dedicate to other activities, decreased conflicts over resources, reinforced property rights over resources, improved institutional collaboration, community organization and capacity building (Álvarez *et al.*, 2018; Defeo *et al.*, 2016; Gelcich *et al.*, 2017, 2010; Palacios-Abrantes, Herrera-Correal, Rodríguez, Brunkow, & Molina, 2018). Similarly, the co-managed pirarucu fisheries in the Brazilian Amazon have improved social sustainability through more equalitarian distribution of income, sense of pride, stronger culture and indigenous and local knowledge (Campos-Silva & Peres, 2016; Freitas *et al.*, 2020).

In coastal small-scale fisheries some problems undermining social sustainability are ongoing conflicts between fishers and managers of protected areas (De Figueiredo Silva *et al.*, 2012; Jimenez *et al.*, 2019; Lopes, Rosa, Salyvonchyk, Nora, & Begossi, 2013; Lopes, Silvano, Nora, & Begossi, 2013). These include increased theft of fishing gear and potential competition for space with industrial vessels (Daw, 2008), high risk practices, such as diving, which can involve accidents (Gamboa-Álvarez *et al.*, 2020; Guebert-Bartholo, Barletta, Costa, Lucena, & Da Silva, 2011) and disruption of fishing cooperatives (Rubio-Cisneros, Aburto-Oropeza, Jackson, & Ezcurra, 2017). Even in the relatively successful co-managed Chilean shellfish. Other social problems at the Brazilian coast include increased commercialization and price of shark meat, which decreases the availability of shark meat for local people and threatens their food security (Barbosa-Filho *et al.*, 2019).

Scientific and indigenous and local knowledge informed assessments have at times differed about the sustainable use of certain fisheries. For example, in a Colombian lagoon community social conflict arose between fishers and researchers due to differences in how they conceptualize sustainability, (Torres-Guevara, Lopez, & Schlüter, 2016). A similar situation was observed in the Dominican Republic where fishers, based on their indigenous and local knowledge, considered the fisheries as more depleted through catches of juvenile fish of most species but scientists believed the fisheries targeted mostly adult fish and would thus be in a better state (Mclean & Forrester, 2018). Both cases draw attention to the need for better dialogue and cooperation between fishers and scientists.

The main social problems related to the inland ornamental fisheries in the Brazilian Amazon are the negative effects of a reduced trade in the Negro River and a potential collapse of exploited species in the Xingu River, which will drastically reduce income and negatively affect the livelihoods of many impoverished riverine people, most of whom lack employment alternatives (Evers, Pinnegar, & Taylor, 2019a). Another social problem of this fishery is the health issues related to the labor-intensive fishing performed mostly by aged fishers. Younger people are less and less involved in these activities. Not only does this have negative impacts on the labor distribution, but may also disrupt knowledge transmission of indigenous and local knowledge (Ladislau *et al.*, 2020).

NORTH AMERICA

From a total of 28 sources on coastal and inland small-scale fisheries in temperate North America retrieved, 22 are included in this review (see the data management report for Chapter 3 systematic literature review at https://doi.org/10.5281/zenodo.6452651), which are evenly distributed between the United States of America (12) and Canada (9). One study addressed both countries, which is also the only study on inland fisheries (Cooke & Murchie, 2015). The reviewed studies include a variety of fishing resources, such as coastal and reef fishes, crabs, lobster, shellfish and sea cucumbers (see the data management report for Chapter 3 systematic literature review at https://doi.org/10.5281/zenodo.6452651). We also include a case study on the sustainability of small-scale whaling activities in the north (see **Box 3.4**).

Six studies focus on economic and 12 studies highlight social considerations in small-scale fisheries of Canada and the United States of America (see the data management report for Chapter 3 systematic literature review at https://doi.org/10.5281/zenodo.6452651). Some of these studies underscore the high economic value of the recreational fisheries practice in Florida, which provides jobs and revenues (Ault, Bohnsack, Smith, & Luo, 2005). For example, the catch-and-release fishery in South Florida (and the Caribbean) has an estimated value of at least half a billion dollars per year (Kroloff *et al.*, 2019). Similarly, the lobster *(Homarus americanus)* fishery is very important to the region of the Gulf of Maine in Canada (Boudreau & Worm, 2010). Some studies indicate potential negative interactions among economic activities. For example, commercial fishing coupled with the expansion of sports (recreational) fishing in the last decades may had affected yelloweye rockfish *(Sebastes ruberrimus)* populations (Eckert *et al.*, 2018). Similarly, food security in Alaska has been negatively affected by the development of export-oriented commercial fisheries and tourism-oriented sport fisheries (Harrison & Loring, 2016). Another study reports changes in fishing area or practices in response to changing market infrastructure (e.g., switch to frozen from salt cod), besides changes in economic factors external to the fishery, such as loss of other income generating activities, which can affect the economic sustainability of cod *(Gadus morhua)* in Newfoundland, Canada (Murray, Neis, & Schneider, 2008).

Some of the studies that mention social characteristics of small-scale fisheries comment on the relevance of fishing resources to local peoples' livelihoods and food security (see the data management report for Chapter 3 systematic literature review at https://doi.org/10.5281/zenodo.6452651). For example, fishing of Arctic char *(Salvelinus alpinus)* has high value for food security, cultural identity and local economic development among Arctic communities (Roux *et al.*, 2019). Conversely, the observed decline in the catches of the Dungeness crab may compromise the ability of indigenous fishers to access traditional foods in Canada (Ban *et al.*, 2017). Other studies emphasize the relevance and benefits of integrating multiple knowledge sources in fisheries assessments, including fishers' indigenous and local knowledge, which may improve dialogue, cooperation and social relations between fishers and scientists (Ambrose *et al.*, 2014; Ban *et al.*, 2017; Murray, Neis, Palmer, *et al.*, 2008; Murray, Neis, & Schneider, 2008; Rehage *et al.*, 2019). The study on inland fisheries mentions that food security and the move towards eating locally may create new markets for freshwater fish, as long as they have low contaminant loads and are considered healthy (Cooke & Murchie, 2015).

ASIA-PACIFIC

The Asia-Pacific region includes countries from Asia, Oceania and the South Pacific Island countries. From a total of 119 sources originally retrieved for this region, 96 studies were included in the review, in conjunction with literature from assessment authors. These studies cover small-scale fisheries in more than 36 countries (see the data management report for Chapter 3 systematic literature review at https://doi.org/10.5281/zenodo.6452651) from Southeast Asia (Mattson, 2006), Western Asia (Al-Abdulrazzak, Zeller, Belhabib, Tesfamichael, & Pauly, 2015) and the Pacific (Cohen & Foale, 2013; Cruz-Trinidad, Aliño, Geronimo, & Cabral, 2014; Eriksson *et al.*, 2018; Kronen, Magron, McArdle, & Vunisea, 2010; D. Zeller *et al.*, 2015). Several countries appeared in only one or two studies; more studies addressed small-scale fisheries in Indonesia (18), the Philippines (10), Australia (7), India (9), Bangladesh (5) and the Solomon Islands (5). The overwhelming majority (82%) of studies addressed coastal or marine and only 10 studies focused on inland small-scale fisheries, whereas three recent studies in Southeast Asia included both coastal and inland fisheries (Jahan, Ahsan, & Farque, 2017; Liao *et al.*, 2019; Millar *et al.*, 2019). Most studies report the uses of finfish, while fewer studies focus on other organisms (sharks, invertebrates). Several studies included many species (finfish and other organisms), evidencing the multi-species characteristic of these small-scale fisheries (see the data management report for Chapter 3 systematic literature review at https://doi.org/10.5281/zenodo.6452651).

Although some studies had short time series of up to one year, several studies analyzed time series of 10 years or more (see the data management report for Chapter 3 systematic literature review at https://doi.org/10.5281/zenodo.6452651) and at least one study included indigenous and local knowledge and archeological data to analyze a time series of 3,000 years in American Samoa (P. Craig, Green, & Tuilagi, 2008). Among the studies analyzing long time series of 50 to 60 years, some include indigenous and local knowledge on temporal trends (Lavides *et al.*, 2016; Muallil, Mamauag, Cababaro, Arceo, & Aliño, 2014; Selgrath, Gergel, & Vincent, 2018a, 2018b; Thurstan, Buckley, Ortiz, & Pandolfi, 2016a), while others apply a methodology to reconstruct catches along time series with missing data (Al-Abdulrazzak *et al.*, 2015; Léopold *et al.*, 2017; D. Zeller *et al.*, 2015).

Considerations or analyses related to economic sustainability were included in 45 and 11 of the reviewed studies on coastal and inland (or coastal and inland) small-scale fisheries, respectively (see the data management report for Chapter 3 systematic literature review at https://doi.org/10.5281/zenodo.6452651). Some of the ecologically sustainable or partially sustainable coastal fisheries also show net economic benefits due to improved or maintained catches, as observed for the shrimp fisheries in Indonesia (Anna, 2017) and abalone fisheries in Australia (Mayfield, Mundy, Gorfine, Hart, & Worthington, 2012).

Fishing is an important economic activity for the Pacific Island countries located in the coral triangle area (Cruz-Trinidad *et al.*, 2014). Some of the co-managed reef fisheries in Pacific Island countries can deliver tangible economic benefits to local communities in the form of increased catches (Tilley, Hunnam, *et al.*, 2019; Webster *et al.*, 2017; Yang & Pomeroy, 2017), for example, through periodic harvesting in protected areas, which can provide a needed boost to local economies (Cohen, Cinner, & Foale, 2013). However, some highly valued economic resources, such as sea cucumbers or lobsters *(Panulirus ornatus)*, have been overfished, particularly in the Philippines and Indonesia, due to increased market demands (Hair, Foale, Kinch, Yaman, & Southgate, 2016; Macusi, Laya-og, & Abreo, 2019; Prescott, Riwu, Prasetyo, & Stacey, 2017). The sea cucumbers fishery has high export value and provides an economic insurance for island populations of Pacific Island countries, but some of these fisheries had to be closed to recover, which compromised the economic benefits (Eriksson *et al.*, 2018; Hair *et al.*, 2016).

The economic sustainability of coastal fisheries could also be negatively affected by long market chains with strong inequalities in the distribution of profits between fishers and final retailers (Ferse, Glaser, Neil, & Schwerdtner Máñez, 2014). The low price paid to fishers can interact with increased costs of fuel and other components of the fishing activity, prompting fishers to intensify their fishing effort to cover fishing trips to more distant fishing grounds (Sebastian Ferse, Knittweis, Krause, Maddusila, & Glaser, 2012; G. M. N. Islam, Noh, Sidique, & Noh, 2014; Muallil, Mamauag, Cababaro, *et al.*, 2014; Rhodes, Tupper, & Wichilmel, 2008).

Aspects related to the social and cultural sustainability were presented in 32 and 8 of the reviewed studies on coastal and inland small-scale fisheries, respectively (see the data management report for Chapter 3 systematic literature review at https://doi.org/10.5281/zenodo.6452651). Several studies highlighted the social benefits of these fisheries in the form of food provision and sustaining livelihoods of local communities (Al-Abdulrazzak *et al.*, 2015; Butler, Tawake, Skewes, Tawake, & McGrath, 2012; Cruz-Trinidad *et al.*, 2014; Friedlander *et al.*, 2014; Golden, Naisilsisili, Ligairi, & Drew, 2014; Rassweiler *et al.*, 2020). Fishing is also an important cultural and social activity among many of the coastal fishing communities, reinforcing cultural identity and social practices, such as sharing fish, in the Pacific Island countries (Golden *et al.*, 2014; Rassweiler *et al.*, 2020). Indeed, Maori coastal fishers in New Zealand have perceived declines in culturally important nearshore resources (fish and invertebrates), which has negative cultural effects on communal activities, social connections, traditions, connections to nature and loss of pride of being able to feed themselves and guests by using seafood (Mccarthy *et al.*, 2014).

Besides improving catches and increasing the abundance of fishing resources, the commons-based management systems implemented in Pacific Islands can promote social sustainability through empowerment of local communities, increased compliance with management rules and the development of a sense of ownership of fishing resources (Butler *et al.*, 2012; Cinner *et al.*, 2012; Friedlander *et al.*, 2014; Webster *et al.*, 2017; Yang & Pomeroy, 2017). These co-management systems often include community rules and beliefs, sometimes resulting in social benefits by participating communities even before perceived improvements on fisheries (Tilley, Hunnam, *et al.*, 2019).

Fishery closures imposed by co-management may exclude some social groups, such as women or immigrants, from access to fishing grounds, besides imposing social costs in the form of restricted harvestings (Ayunda, Sapota, & Pawelec, 2018; Cohen & Foale, 2013). The relationship between fishers and middlemen can either improve or undermine social sustainability. For example, in Indonesia, some of the middlemen (locally called patrons) may have social ties with fishers and contribute to social welfare by providing social security for impoverished fishers in need, whereas other, wealthier patrons (big patrons), may not have these social ties. This may result in provision of credit and loans to fishers to buy fishing gear (including illegal and high impact types) which may result in unsustainable fishing practices and further exploit fishers by making them sell catches at low prices (Ferse *et al.*, 2014; Ferse *et al.*, 2012).

3.3.1.4.1 Indicators of small-scale fisheries sustainability

Across the 350 small-scale fisheries studies the main indicators adopted were: (i) catch biomass or composition (landings' data) in 214 studies, (ii) measures of catch per unit of effort, in 78 studies), (iii) abundance estimates and trends (72 studies), (iv) based on either fishers' knowledge or biological sampling, fishing effort, such as number of boats and other measures (73 studies), (v) size of harvested species (57 studies) and varied measures of stock assessment (51 studies). The majority (214) of reviewed studies included indigenous or local knowledge from fishers to inform the indicators outlined here, so fishers' knowledge can be also considered an important indicator and information source for small-scale fisheries. Some studies have also included economic related indicators, such as market prices, costs, revenues (83 studies) or social indicators, such as culture, governance or management (46 studies), see the data management report for Chapter 3 systematic literature review at https://doi.org/10.5281/zenodo.6452651 for more detailed data.

EUROPE AND CENTRAL ASIA

A very diverse set of indicators using three perspectives (i.e., ecological, economic and social) was employed to assess sustainability in the papers reviewed. The use of parameters from stock assessments analysis as indicators for ecological sustainability assessments is not very common in the literature and only a few studies use them in conjunction with other indicators, such as maximum sustainable yield (Colloca *et al.*, 2017; Dinesen *et al.*, 2019; Hornborg & Främberg, 2019; Marcos *et al.*, 2015), or different measurements of stock abundance and distribution (Bastari *et al.*, 2017; Braga, Pardal, & Azeiteiro, 2018; Damalas *et al.*, 2015; Lloret *et al.*, 2015; Macdonald, Angus, Cleasby, & Marshall, 2014; Shephard *et al.*, 2019; Szostek, Murray, Bell, & Kaiser, 2017).

The use of fish biometry and size distributions in cohort analysis is not usual, but is present (Grati *et al.*, 2018; Shephard *et al.*, 2019; Vasconcelos *et al.*, 2020), also in association with other methods and indicators. However, as expected, most of the ecological assessments reviewed (41 out of 63 papers) support their conclusions with landing statistics (production/catch biomass, catch composition) and related parameters to measure fishing effort and catch-per-unit-of-effort.

Catch biomass or biomass landed (58.2% of reviewed studies), catch-per-unit-of-effort (40.3%) and catch composition or species landed (13.4%) were the indicators used more frequently in the ecological evaluations of small-scale fishing in Europe (see the data management report for Chapter 3 systematic literature review at https://doi.org/10.5281/zenodo.6452651). In addition, the use of indicators of local ecological knowledge from local fishers in association with other indicators, is notable (Azzurro *et al.*, 2019; Braga *et al.*, 2017, 2019; Coll *et al.*, 2014; Corral & Manrique de Lara, 2017; Damalas *et al.*, 2015; Dinesen *et al.*, 2019; Figus *et al.*, 2017; Lloret *et al.*, 2015; Maynou *et al.*, 2011; Öndes, Kaiser, & Güçlüsoy, 2020).

Socioeconomic assessment alone was a rare approach in the literature of fishing sustainability (Ünal & Franquesa, 2010). Nevertheless, the assessment of economic and social aspects of European small-scale fisheries as part of ecological assessments was not that unusual, and made use of a set of related indicators such as values of landings, market values, market prices, revenue and income generation, both by the fleets and by the individual fishers (Carvalho *et al.*, 2017; Grati *et al.*, 2018; Guyader *et al.*, 2013; Lloret *et al.*, 2018; Maynou *et al.*, 2014, 2013; Pita *et al.*, 2019; Quetglas *et al.*, 2017; Rivera *et al.*, 2016; Rivera *et al.*, 2017; Roditi & Vafidis, 2019; Sartor *et al.*, 2019; Silva *et al.*, 2019b; Tzanatos *et al.*, 2013; Ulman *et al.*, 2013).

Despite the fact that a very limited number of assessments based on the social perspective was found in the reviewed literature, these studies applied a diverse set of indicators. Those indicators were based on tradition (cultural, historic values) and on the level of dependence of the local communities on the fishing practices for their livelihoods (Guyader *et al.*, 2013; Ünal & Franquesa, 2010). The more frequent approach for social assessments was the use of the indicators of governance efficiency and effectiveness of fishers' organizations in charge of co-management, or participatory management systems of aquatic resources (Baeta *et al.*, 2018; Morales-Nin *et al.*, 2017; Silva *et al.*, 2019b). These may represent the main critical issues that are discussed by experts on the social perspectives of the European small-scale fishing and fishers.

AFRICA

Since proper stock assessments are not very common (due to high costs, lack of personnel, time and other means) the authors used a diverse set of indicators. Only a small number of studies used stock assessments to produce estimates of maximum sustainable yield, yield per recruit, or cohort analysis and species-specific life table parameters (Fulanda *et al.*, 2011; Hara & Njaya, 2015; Jamu *et al.*, 2011; Meissa, Gascuel, & Rivot, 2013; Rehren, Wolff, & Jiddawi, 2018). Most of the assessments support their conclusions based on series of catch/production, such as landing statistics. Catch biomass (biomass landed) and catch composition (species landed) are the more frequent parameters in the ecological evaluations. Nevertheless, catch-per-unit-of-effort and size distribution of fish landed are also frequently used indicators. More than 45 papers use fish landings and/or catch-per-unit-of-effort as indicators to support their analysis (see the data management report for Chapter 3 systematic literature review at https://doi.org/10.5281/zenodo.6452651).

Additional indicators were used for the assessment of economic and social aspects. Indicators for economic evaluation were revenue and market prices (Blythe, Murray, & Flaherty, 2013), relevance of foreign markets for exportations, and added costs and values (Baker-Médard & Faber, 2020). Indicators for social evaluation were level of dependence for livelihoods, employment, number of people involved (Belhabib *et al.*, 2015), influence of indigenous and local knowledge and the persistence/resilience of these last two (Bulengela *et al.*, 2019; Gaspare, Bryceson, & Kulindwa, 2015). In some cases, the persistence of cultural traits, like traditional knowledge, was seen as an indicator of social sustainability (Mirera *et al.*, 2013).

LATIN AMERICA

This review evidenced the limitations imposed by the lack of continuous monitoring to provide fisheries and biological data to evaluate sustainable use. Only a few studies included more detailed population analyses and measured conventional stock parameters, such as maximum sustainable yield, natural mortality, fishing mortality, among others (Aburto & Stotz, 2013; Baigún, Minotti, & Oldani, 2013; Catarino, Kahn, & Freitas, 2019; Cavieses Núñez, Ojeda Ruiz De La Penã, Flores Irigollen, Rodríguez Rodríguez, & Jardim, 2018; Duarte *et al.*, 2018a; Martínez-Candelas *et al.*, 2020; Mesquita, Cruz, Hallwass, & Isaac, 2019). The reviewed studies applied a varied set of indicators, often in combination, including total catches or landings (58), catch-per-unit-of-effort (22), and size of exploited fishing resources (37). Catch composition and its variation through time, measures of fishing effort, such as number of fishers, vessels and the distribution of effort in space and time, economic indicators (revenues and costs), and overall abundance trends estimated from indigenous and local knowledge were also used as indicators of sustainable use (see the data management report for Chapter 3 systematic literature review at https://doi.org/10.5281/zenodo.6452651).

A few studies calculated and compared sustainability indicators based on ecological, economic and social data (Cissé *et al.*, 2014; Robotham *et al.*, 2019; Torres-Guevara *et al.*, 2016). However, most of the reported trends are based on total catches only. The lack of effort or catch-per-unit-of-effort data makes it more difficult to properly assess the sustainability of these fisheries. Furthermore, while some species are preferred, most of these fisheries are multi-species and multi-gear (see the data management report for Chapter 3 systematic literature review at https://doi.org/10.5281/zenodo.6452651). This imposes further challenges to sustainability assessments, as exploited species may differ regarding their resilience to fishing pressure and stock status. These challenges were addressed by most of the reviewed studies through two main, non-mutually exclusive, approaches. First, to rely on a variety of the indicators described above and second, to include fishers' knowledge about catches, trends, details of fishing effort in combination with fisheries data, biological surveys or modelling. Indeed, indigenous and local knowledge was included in the majority (69) of studies reviewed (see the data management report for Chapter 3 systematic literature review at https://doi.org/10.5281/zenodo.6452651).

Many studies stressed the important economic role of both coastal and inland small-scale fisheries in the studied regions, but relatively few studies included economic indicators (profits, revenues), analyzed market chains, or evaluated the economic sustainability of the studied fisheries (see the data management report for Chapter 3 systematic literature review at https://doi.org/10.5281/zenodo.6452651). Economic considerations were mentioned by 42% of the 78 studies on coastal small-scale fisheries and by 45% of the 29 studies on inland small-scale fisheries, sometimes linked to the analysis of catch trends and ecological sustainability.

NORTH AMERICA

The indicators adopted in the reviewed studies include catch (landings data), population and stock parameters, environmental or ecological indicators, productivity susceptibility analysis and various modelling approaches (see the data management report for Chapter 3 systematic literature review at https://doi.org/10.5281/zenodo.6452651). Even considering that both countries have a well-developed fisheries science and management with strong financial and technical capacity, the majority of the reviewed studies (17) include fishers' knowledge or indigenous and local knowledge, usually in combination with the above-mentioned fisheries and ecological indicators (see the data management report for Chapter 3 systematic literature review at https://doi.org/10.5281/zenodo.6452651). Moreover, fishers' knowledge has been included in these studies on various forms or manifestations, from traditional knowledge of indigenous people, usually from the Artic (Ambrose *et al.*, 2014; Ban *et al.*, 2017; Eckert *et al.*, 2018; Roux *et al.*, 2019) to local knowledge held by recreational fishers or commercial harvesters (Frezza & Clem, 2015; Kroloff *et al.*, 2019; Murray, Neis, Palmer, *et al.*, 2008; O'Regan, 2015).

ASIA-PACIFIC

The reviewed studies employed a wide range of indicators, most commonly catches (landings data), catch-per-unit-of-effort, fishing effort, abundance (density) and size of exploited fishing resources, besides socioeconomic indicators (see the data management report for Chapter 3 systematic literature review at https://doi.org/10.5281/zenodo.6452651). Almost two thirds (66) of the reviewed studies included indigenous and local knowledge-based indicators to inform fish abundance trends, catches, catch-per-unit-of-effort, sizes, fishing effort, perceptions on management or socioeconomic status, thus indicating the relevance of indigenous and local knowledge and collaboration with fishers for research on these small-scale fisheries.

3.3.1.4.2 The role of indigenous and local knowledge in small-scale fisheries

Despite the review provided here, it is also widely acknowledged that most small-scale fisheries remain unreported and unmonitored, resulting in the lack of longer time series data to evaluate their sustainability. This is especially the case in tropical countries and the Arctic, where small-scale fisheries are widespread. These data gaps can be overcome through collaborative research to record and analyze fishers' local ecological knowledge, a form of indigenous and local knowledge based on an experiential understanding of one's environment coupled with communal and historical use. Fishers' local ecological knowledge contributes to estimates on temporal trends in abundance of fisheries resources, and can extend the time series available for the analysis to periods before scientific monitoring (Giglio, Luiz, & Gerhardinger, 2015; Hallwass *et al.*, 2020; Jahan, Ahsan, & Farque, 2017; Maia *et al.*, 2018; Stocks, Foster, Bat, Ha, & Vincent, 2019a) or data (Sáenz–Arroyo, Roberts, Torre, & Cariño-Olvera, 2005) were available. Indeed, in many cases worldwide fishers' knowledge is the only available knowledge source.

In the last 20 years, several studies have recorded fisher indigenous and local knowledge and local ecological knowledge through using qualitative methods, such as interviews with fishers, to reconstruct temporal trends in fisheries resources. This was the case in 56 of the studies reviewed here. Through these studies data were collected from an aggregated total of 13,565 fishers (through interviews), on approximately 454 fish species in 32 countries worldwide **(Table 3.3)**. All the studies further quantitatively analyzed fishers' local ecological knowledge to identify trends in abundance, size and composition of fisheries resources through a series of indicators such as estimated abundance categories (declined, same, increased), catch per unit of effort, amounts of regular, poor and best catches, and size (length or weight) of largest ever caught **(Table 3.3)**.

The time span covered by these studies varies from 5 to 10 years (Daw, Robinson, & Graham, 2011; Liao *et al.*, 2019; Lima, Begossi, Hallwass, & Silvano, 2016; O'Donnell, Molloy, & Vincent, 2012) to several decades, with some going back to the 1950s and 1960s (Ainsworth, 2011; Lavides *et al.*, 2016; Lozano-Montes, Pitcher, & Haggan, 2008). The influence of time on fishing parameters has been analyzed either as a continuous variable (for example, year of the best catch) or as an interval categorical variable (for example, discrete years or decades according to fishers' age groups, specific events, etc.) **(Table 3.3)**.

Most of the studies reported declining trends in abundance, catch-per-unit-of-effort or size of fishing resources **(Table 3.3)**. Reported declines were usually focused on threatened species, some of which had been intensely exploited, such as reef fishes from the genus *Epinephelus* and *Mycteroperca* (groupers) (Bender, Floeter, & Hanazaki, 2013; Bender *et al.*, 2014; Bunce, Rodwell, Gibb, & Mee, 2008; Castellanos-Galindo *et al.*, 2018; Giglio *et al.*, 2015; Ribeiro, Damasio, & Silvano, 2021a; Zapelini, Bender, Giglio, & Schiavetti, 2019), the large catfish *(Pangasius sanitwongsei)* in the Mekong River (Gray, Phommachak, Vannachomchan, & Guegan, 2017), seahorses (*Hippocampus* spp.) (Stocks, Foster, Bat, Ha, & Vincent, 2019b), the angel shark *(Squatina squatina)* in the Mediterranean (Fortibuoni, Borme, Franceschini, Giovanardi, & Raicevich, 2016), sawfish species (*Pristis* spp.) in coastal ecosystems (Jabado *et al.*, 2017; Leeney & Poncelet, 2015), and the paddlefish *(Psephurus gladius)* in Yangtze River (Turvey *et al.*, 2010), among others **(Table 3.3)**.

A phenomenon sometimes related to studies based on fishers' memories to reconstruct past events is known as shifting baseline syndrome, i.e., environmental changes may be recognized only by older fishers and underestimated or not recognized by younger ones (Papworth, Rist, Coad, & Milner-Gulland, 2009; Pauly, 1995). Shifting baseline has been observed by many studies worldwide, which reported an influence of age on fishers' perceptions about changes in the abundance of fisheries resources (Bender *et al.*, 2013, 2014; Katikiro, 2014; Lozano-Montes *et al.*, 2008; Maia *et al.*, 2018; Turvey *et al.*, 2010; Ulman & Pauly, 2016). However, further studies show that this is not always the case, and both older and younger fishers may hold similar perceptions (Barbosa-Filho *et al.*, 2020; Hallwass, Lopes, Juras, & Silvano, 2013; Ribeiro *et al.*, 2021; Thurstan, Buckley, Ortiz, & Pandolfi, 2016b). Furthermore, fishers also report stable catches or sizes of at least some fish species (Ribeiro *et al.*, 2021; Silvano & Hallwass, 2020).

Limitations related to the application of fishers' local ecological knowledge to estimate abundance trends include heavy reliance on fishers' memories, which at time may be inaccurate or biased due to memory illusion or shifting baseline syndrome (Daw *et al.*, 2011; O'Donnell, Molloy, & Vincent, 2012; Papworth *et al.*, 2009). However, it should be noted that the ways in which local ecological knowledge and indigenous and local knowledge data are collected include methods for minimizing bias such as data triangulation

Table 3 3 **Study cases applying fishers' local or indigenous ecological knowledge for quantitative analyses of temporal trends on small-scale fisheries.**

N: number of interviewed fishers; **Trend:** time trends (C: continuous; I: interval (categories)); **Abundance:** overall trends in abundance estimated by fishers (D: declined; S: stable, IC: increased); **CPUE:** catch per unit of effort; **Catches:** include estimates of either regular or best catches; **Size:** usually the largest individual ever caught, in length or weight; **Composition:** relative abundance and number of species in the catch; **Scientific data:** besides data from fishers' knowledge (AG: agreement between fishers' knowledge and scientific data; DA: disagreement; PA: partial agreement); (*) Taxonomic level: species groups.

Country	Ecosystem	N	Trend	Abundance	CPUE	Catches	Size	Composition	# Species	Taxon	Scientific data	Source
AFRICA												
Guinea	Coastal	178	C / I			D / S	D	Changed	06	Fish	No	[1]
Guinea-Bissau	Coastal	274	I	D					01	Fish (sawfish)	No	[2]
Mauritius Islands	Coastal	093	I			D	D	Changed	25	Fish	No	[3]
Red Sea (Eritrea, Sudan, Yemen)	Coastal	423	C		D	D			Several	Non specified	No	[4]
Seychelles	Coastal	040	I		D	D			Several	Fish	**Yes / DA**	[5]
Tanzania	Coastal	350	I			D	D	Changed	17	Fish, shrimp, squid, octopus(*)	No	[6]
ASIA												
Bangladesh	Freshwater	200	I			D			01	Fish	No	[7]
China	Coastal	400	I	D		D			02	Horseshoe, crabs	**Yes / AG**	[8]
China	Freshwater	599	C			D			03	Fish	**Yes / AG**	[9]
India	Freshwater	100	I	D					58	Fish, shrimp	No	[10]
Lao	Freshwater	120	I	D / S					08	Fish	No	[11]
UAE	Coastal	082	I	D					01	Fish (sawfish)	No	[12]
Vietnam	Coastal	077	C		D		S		06	Fish	No	[13]

Country	Ecosystem	N	Trend	Abundance	CPUE	Catches	Size	Composition	# Species	Taxon	Scientific data	Source
EUROPE AND CENTRAL ASIA												
Adriatic (Italy, Slovenia, Croatia)	Coastal	052	C / I	D			D		01	Fish (shark)	**Yes / AG**	[14]
Italy	Coastal	032	C / I	D / S / IC				Changed	59	Fish	No	[15]
Mediterranean (Spain, Italy, Greece)	Coastal	091	I	D / S / IC	D		D / S / IC		42	Fish, invertebrates (shrimp, lobster, mollusks)	No	[16]
Poland	Coastal	031	I	D			D		01	Fish	**Yes / AG**	[17]
Scotland	Coastal	062	I	IC	IC	IC			01	Fish	**Yes / AG**	[18]
Spain	Coastal	064	C			D	D	Changed	06	Fish	**Yes / AG**	[19]
Turkey	Coastal	176	C	D	D				Several	Non specified	**Yes / AG**	[20]
Turkey	Coastal	155	I	D					01	Shellfish	**Yes / AG**	[21]
NORTH AMERICA												
Canada	Coastal	020	I	D	D		D		1	Sea cucumber	No	[22]
Canada	Coastal	042	C	D / IC					16	Fish	**Yes / AG**	[23]
Canada	Coastal	038	C / I	D	D	D / S			1	Crab	**Yes / AG**	[24]
Canada	Coastal	042	I	D			D		1	Fish	**Yes / AG**	[25]
Mexico	Coastal	108	C			D	D		1	Fish	**Yes / DA**	[26]
Mexico	Coastal	127	C			D			2	Shellfish	**Yes / AG**	[27]
Mexico	Coastal	049	I			D			3	Fish	**Yes / AG**	[28]
Mexico	Coastal	081	I	D					22	Fish, shrimp, lobster, mollusks(*)	**Yes / AG**	[29]

Country	Ecosystem	N	Trend	Abundance	CPUE	Catches	Size	Composition	# Species	Taxon	Scientific data	Source
PACIFIC												
Australia	Coastal	0141	C			S			05	Fish, shrimp	Yes / PA	[33]
Indonesia	Coastal	0186	I	D				Changed	16	Fish (shark)	Yes / AG	[34]
Philippines	Coastal	2655	I		D	D		Changed	05	Fish	**No**	[35]
Philippines	Coastal	3446	I	D	D			S	Several	Fish	**No**	[36]
Philippines	Coastal	0025	C		S				01	Fish	Yes / PA	[37]
Tonga	Coastal	0029	I	IC					Several	Fish	Yes / DA	[38]
SOUTH AMERICA												
Brazil	Coastal	0081	C / I		D / IC	D / IC	D / S	Changed	08	Fish, crab, shrimp	No	[39]
Brazil	Freshwater	0203	C / I		D			Changed	15	Fish	No	[40]
Brazil	Coastal	0082	I		S			Changed	22	Fish	**Yes / DA**	[41]
Brazil	Freshwater	0041	I			D / IC	D / S	Changed	16	Fish	**Yes / PA**	[42]
Brazil	Coastal	0222	I	D			S		01	Fish	No	[43]
Brazil	Coastal	0240	C	D					01	Fish	No	[44]
Brazil	Coastal	0079	I				D		01	Fish (shark)	No	[45]
Brazil	Coastal	0034	C			D			04	Fish	No	[46]
Brazil	Coastal	0359	I	D					04	Fish	No	[47]
Brazil	Coastal	102	C / I			D	D		02	Fish	No	[48]
Brazil	Coastal	053	C				D / S		09	Fish	No	[49]
Brazil	Coastal	210	I	D / S		D	D		08	Fish	No	[50]
Brazil	Coastal	214	C			D	D		09	Fish	**Yes / AG**	[51]
Brazil	Coastal	022	C	D					01	Fish	**Yes / AG**	[52]
Brazil	Freshwater	182	I	D					01	Fish	**Yes / AG**	[53]

Country	Ecosystem	N	Trend	Abundance	CPUE	Catches	Size	Composition	# Species	Taxon	Scientific data	Source
SOUTH AMERICA												
Brazil	Freshwater	300	I	D / IC					14	Fish, shrimp	**Yes / AG**	[54]
Chile	Coastal	123	C / I			D	D	Changed	03	Fish	**Yes / AG**	[55]
Colombia	Coastal	046	C	D		S			01	Fish	**Yes / DA**	[56]

Sources: **[1]** (Maia *et al.*, 2018); **[2]** (Leeney & Poncelet, 2015); **[3]** (Bunce *et al.*, 2008); **[4]** (Tesfamichael, Pitcher, & Pauly, 2014); **[5]** (Tim M Daw *et al.*, 2011); **[6]** (Katikiro, 2014); **[7]** (Jahan *et al.*, 2017); **[8]** (Liao *et al.*, 2019); **[9]** (Turvey *et al.*, 2010); **[10]** (S. Dey *et al.*, 2019); **[11]** (T. N. E. Gray *et al.*, 2017); **[12]** (Jabado *et al.*, 2017); **[13]** (Stocks *et al.*, 2019); **[14]** (Fortibuoni *et al.*, 2016); **[15]** (Ernesto Azzurro *et al.*, 2011); **[16]**(Damalas *et al.*, 2015); **[17]** (Figus *et al.*, 2017); **[18]** (P. Macdonald *et al.*, 2014); **[19]** (Coll *et al.*, 2014); **[20]** (Ulman & Pauly, 2016); **[21]** (Öndes, Kaiser, & Güçlüsoy, 2020); **[22]** (O'Regan, 2015); **[23]** (Boudreau & Worm, 2010); **[24]** (Ban *et al.*, 2017); **[25]** (Eckert *et al.*, 2018); **[26]** (Sáenz-Arroyo *et al.*, 2005); **[27]** (Sáenz-Arroyo & Revollo-Fernández, 2016); **[28]** (Lozano-Montes *et al.*, 2008); **[29]** (Ainsworth, 2011); **[30]** (Rehage *et al.*, 2019); **[31]** (Frezza & Clem, 2015); **[32]** (Beaudreau & Levin, 2014); **[33]** (Thurstan *et al.*, 2016a); **[34]** (Jaiteh *et al.*, 2017); **[35]** (Lavides *et al.*, 2016); **[36]** (Muallil, Mamauag, Cababaro, *et al.*, 2014); **[37]** (O'Donnell *et al.*, 2012); **[38]** (Webster *et al.*, 2017); **[39]** (Santos Thykjaer, dos Santos Rodrigues, Haimovici, & Cardoso, 2019); **[40]** (G. Hallwass *et al.*, 2019); **[41]** (Damasio *et al.*, 2015); **[42]** (Strieder Philippsen *et al.*, 2017); **[43]** (Barbosa-Filho *et al.*, 2020); **[44]** (de Souza Junior, Nunes, & Silvano, 2020); **[45]** (Giglio & Bornatowski, 2016); **[46]** (I. M. Martins, Medeiros, Di Domenico, & Hanazaki, 2018); **[47]** (Jimenez *et al.*, 2019); **[48]** (Giglio *et al.*, 2015); **[49]** (Bender *et al.*, 2013); **[50]** (C. Zapelini *et al.*, 2019); **[51]** (Bender *et al.*, 2014); **[52]** (Cleverson Zapelini, Giglio, Carvalho, Bender, & Gerhardinger, 2017); **[53]** (Leandro Castello, Arantes, Mcgrath, Stewart, & Sousa, 2015); **[54]** (Gustavo Hallwass *et al.*, 2013); **[55]** (Godoy *et al.*, 2010); **[56]** (Castellanos-Galindo *et al.*, 2018).

amongst community members, data comparisons with archival and spatial data, and sampling techniques intended to identify the most robust knowledge holders. It is also quite common to search for points of comparison between indigenous and local knowledge/ local ecological knowledge and scientific knowledge. For example, more than half (29) of the reviewed studies included conventional scientific databases, such as biological sampling, fish catches, or governmental monitoring, which were compared with data gathered from fishers **(Table 3.3)**. Although disagreements or partial agreements were observed in eight studies, most studies (21) showed high levels of agreement between trends based on local ecological knowledge and those based on scientific data **(Table 3.3)**. This further reinforces the usefulness and reliability of fishers' local ecological knowledge to evaluate temporal trends in fisheries.

Other studies integrated fishers' local ecological knowledge and conventional scientific data in models to show fisheries trends (Ainsworth, 2011; Ban *et al.*, 2017). A few studies also analyzed and observed temporal changes in the composition of fishing resources **(Table 3.3)**, usually indicating a shift from the exploitation of more valuable large fish to less valuable smaller fish (Coll *et al.*, 2014; Godoy, Gelcich, Vasquez, & Castilla, 2010; G. Hallwass *et al.*, 2019; Jaiteh, Hordyk, Braccini, Warren, & Loneragan, 2017; Strieder Philippsen, Minte-Vera, Okada, Carvalho, & Angelini, 2017) or the disappearance of some species altogether (Damasio, Lopes, Guariento, & Carvalho, 2015; Katikiro, 2014; Lavides *et al.*, 2016). These temporal changes in catch composition **(Table 3.3)** suggest that fisheries may have experienced 'ecological filters' in some freshwater and marine ecosystems, indicating genetic selection through specific forms of harvesting activities. The extent to which species diversity or specific species characteristics are affected in this way is uncertain.

Some of the reviewed studies have also provided useful information based on fishers' local ecological knowledge related to drivers or consequences of observed trends, including protected areas (Hallwass *et al.*, 2020), environmental impacts including dams or pollution (S. Dey, Choudhary, Dey, Deshpande, & Kelkar, 2019; Frezza & Clem, 2015; Gustavo Hallwass, Lopes, Juras, & Silvano, 2013; Jahan, Ahsan, & Farque, 2017; Strieder Philippsen *et al.*, 2017), climate change (Ernesto Azzurro, Moschella, & Maynou, 2011; Eckert *et al.*, 2018), distribution and ecology of invasive species (Araujo Catelani, Petry, Mayer Pelicice, & Azevedo Matias Silvano, 2021; Ernesto Azzurro & Cerri, 2021; Boughedir *et al.*, 2015; van Putten *et al.*, 2016), or trophic cascades associated with fishing (Boudreau & Worm, 2010; Ulman & Pauly, 2016).

Literature based on fishers' local ecological knowledge provides relevant and new data about many ecological parameters of fisheries including reproduction (season, sizes, sites), migratory behavior, spatial distribution, conditions, and trophic relationships (Aswani & Hamilton, 2004; Begossi, Salivonchyk, Lopes, & Silvano, 2016; Begossi *et al.*, 2011, 2019; Figus *et al.*, 2017; Gaspare *et al.*, 2015; Gerhardinger, Marenzi, Bertoncini, Medeiros, & Hostim-Silva, 2006; Hamilton, Giningele, Aswani, & Ecochard, 2012; Johannes, Freeman, & Hamilton, 2000; Le Fur, Guilavogui, & Teitelbaum, 2011; Leite & Gasalla, 2013; Lopes, Verba, Begossi, & Pennino, 2019; Mclean & Forrester, 2018; Nunes, Cardoso, Soeth, Silvano, & Fávaro, 2021; Nunes, Hallwass, & Silvano, 2019; Silva *et al.*, 2019b; Silvano & Begossi, 2012; Silvano, MacCord, Lima, & Begossi, 2006). Fishers' knowledge has also contributed to participatory spatial planning to map bycatch potential of endangered species, such as sea turtles by artisanal fisheries in the coast of Mexico (Cuevas, Guzmán-Hernández, Uribe-Martínez, Raymundo-Sánchez, & Herrera-Pavon, 2018) or to assess bycatch rates and mortality of the Ganges River dolphins *(Platanista gangetica gangetica)* (Dewhurst-Richman *et al.*, 2020). These ecological data provided by fishers could also be useful to assess sustainability of small-scale fisheries and improve their management.

A promising way forward to better integrate fishers' local ecological knowledge and provide needed data about poorly known small-scale fisheries includes collaborations with fishers. This could include participatory monitoring that facilitates fisher involvement in abundance surveys and recording catch, size and information on reproduction of fisheries resources, and occurrence of bycatch (Begossi, Salivonchyk, & Silvano, 2016; Cuevas *et al.*, 2018; Dias, Cinti, Parma, & Seixas, 2020; Keppeler, Hallwass, Santos, da Silva, & Silvano, 2020; Keppeler, Hallwass, & Silvano, 2017; Obura, Wells, Church, & Horrill, 2002; O'Donnell *et al.*, 2012; Schemmel *et al.*, 2016; Silvano, 2020; Silvano & Hallwass, 2020; Webster *et al.*, 2017).

3.3.1.4.3 Pelagic fisheries for forage fish

Small pelagic fish populations, also called forage fish, such as sardine, capelin, anchovy, herring and mackerel, provide about 25% of the total annual production of capture fisheries worldwide (FAO, 2020d). These resources contribute significantly to the well-being of coastal communities around the world, particularly in developing countries. Small pelagic fish are plankton feeders and represent the main prey items for several predators (piscivorous fish including sharks, mammals and birds), and play a key role in marine ecosystems by sustaining numerous higher trophic level species, many of which are commercially targeted (Alder, Campbell, Karpouzi, Kaschner, & Pauly, 2008; Bakun, Babcock, Lluch-Cota, Santora, & Salvadeo, 2010; Essington *et al.*, 2015; Smith *et al.*, 2011). Fisheries for small pelagic fish have a high economic value because of their use for human consumption and for the production of fish meal and fish oil. These fisheries are not only critically important in terms of future global food security but are also pivotal to the economies of small-scale fisheries communities (Pikitch *et al.*, 2014). It has been estimated that fisheries supported by forage fish are actually more than twice as valuable as forage fisheries themselves, providing a strong economic argument for their conservation (Pikitch, 2015).

Populations of small pelagic fish exhibit extreme fluctuations in abundance and geographic distribution due to the impact of environmental factors, which are often amplified by anthropogenic influences (Essington *et al.*, 2015; Izquierdo-Peña, Lluch-Cota, Hernandez-Rivas, & Martínez-Rincón, 2019; Stephenson & Smedbol, 2019). The exploitation of many stocks of pelagic fishes has exhibited a pattern of sharply increasing catches followed by an even more rapid decline **(Figure 3.23)**, leading in several cases to closure of the fishery (Stephenson & Smedbol, 2019). Nonetheless, Froehlich *et al.* (2018) calculated the maximum sustainable yield for 401 stocks that comprise 99% of global forage fish catch, and estimated that the average small pelagic fish catch could increase by 30% from 2012 levels, which would correspond to raising the average (post-1980) small pelagic fish limit by 1.8 million tons per year.

3.3.1.4.4 Pelagic fisheries for billfishes, tuna and tuna-like species

Fisheries targeting tuna and tuna-like species and billfishes are of great socioeconomic importance due to high economic value and extensive international trade and are therefore highlighted in the sustainable use assessment. Tuna accounts for over 9% of total marine fisheries catch, is the fourth most valuable globally traded fishery product, and is about 8% of the 129 billion United States dollars value of internationally traded fishery products (FAO, 2014d, 2018d). Fisheries targeting these species provide substantial economic revenue, employment and food security to fishing and coastal states (Bell & Secretariat of the Pacific Community, 2011; FAO, 2018d; Gillett, 2009).

Tunas and billfishes have been an important food source since ancient times, and are target species of fisheries worldwide (Majkowski, 2007; Miyake, Guillotreau, & Sun, 2010). In the 19th century, most tuna fisheries were coastal, conducted by locally-based fleets (Majkowski, 2005, 2007). Industrial tuna fisheries began in the 1940s. Over the next few decades, fishing grounds quickly expanded as did the number of countries with large-scale coastal and distant-water tuna fleets. About 82% of world tuna is consumed as canned product, and 18% as fresh product (including as sashimi) (Miyake *et al.*, 2010). Japan consumes an estimated 78% of the fresh tuna (Miyake *et al.*, 2010). Demand for both canned and fresh tuna has increased rapidly, with reported landings of principal market tunas increasing from about 700 thousand tons in 1960 to almost 4.8 million tons in 2014 (SPC, 2015) **(Figure 3.24)**.

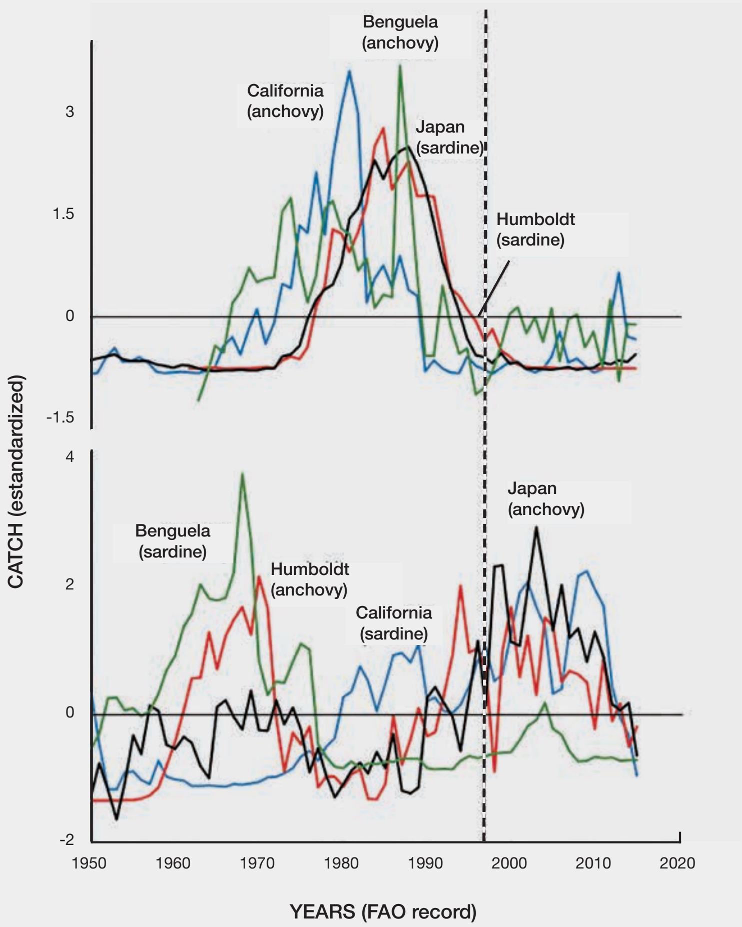


Figure 3 23 **Standardized catch time series for sardines and anchovies from the four largest small pelagics fisheries: Japan, Humboldt, Benguela, and California ecosystems.**

Data from the Food and Agriculture Organization of the United Nations' fishstat data base. Source: (Izquierdo-Peña *et al.*, 2019)

Since 2006, over half of principal market tunas have come from the western and central Pacific Ocean (SPC, 2015). Several Pacific Island countries and territories obtain a large proportion of their gross domestic product through revenue from tuna fisheries, as high as 63% of total government revenue in some cases (Aqorau, 2009; Bell *et al.*, 2015; FFA, 2015; Gillett, 2009). This includes licensing, fees, and granting access to foreign purse seine and longline tuna fisheries to fish in their exclusive economic zones. Capture and processing practices generate additional revenue and substantial employment in the Pacific Islands (Bell *et al.*, 2015; FFA, 2015; Gillett, 2009). In 2014, the Pacific sslands forum fisheries agency (15 Pacific small islands developing states, Australia and New Zealand) obtained an estimated 556 million United States dollars of their combined gross domestic product from the tuna fisheries, and employed over 22,000 people in processing and various other tuna-practice related positions (FFA, 2015). Some locally-based tuna fisheries supply largely low-value fishes (smaller tunas, incidental tuna-like species) to local markets in Pacific Island countries and territories, contributing to local food security and tourism industries (Bell *et al.*, 2015; Gillett, 2009).

Single-stock assessment models are the most common approach used by fisheries management authorities to assess the sustainability of stocks of principal market species of tuna, tuna-like species and billfishes. The four regional fisheries management organizations for tropical tunas have recently adopted and implemented single-stock harvest strategies. The main elements of harvest strategies are outlined in the following literature: (Sainsbury, 2000; WCPFC, 2014).

The status of most but not all stocks of principal market tunas and billfishes is relatively certain (ISSF, 2020; Juan-Jordá, Mosqueira, Freire, & Dulvy, 2013; Pons *et al.*,

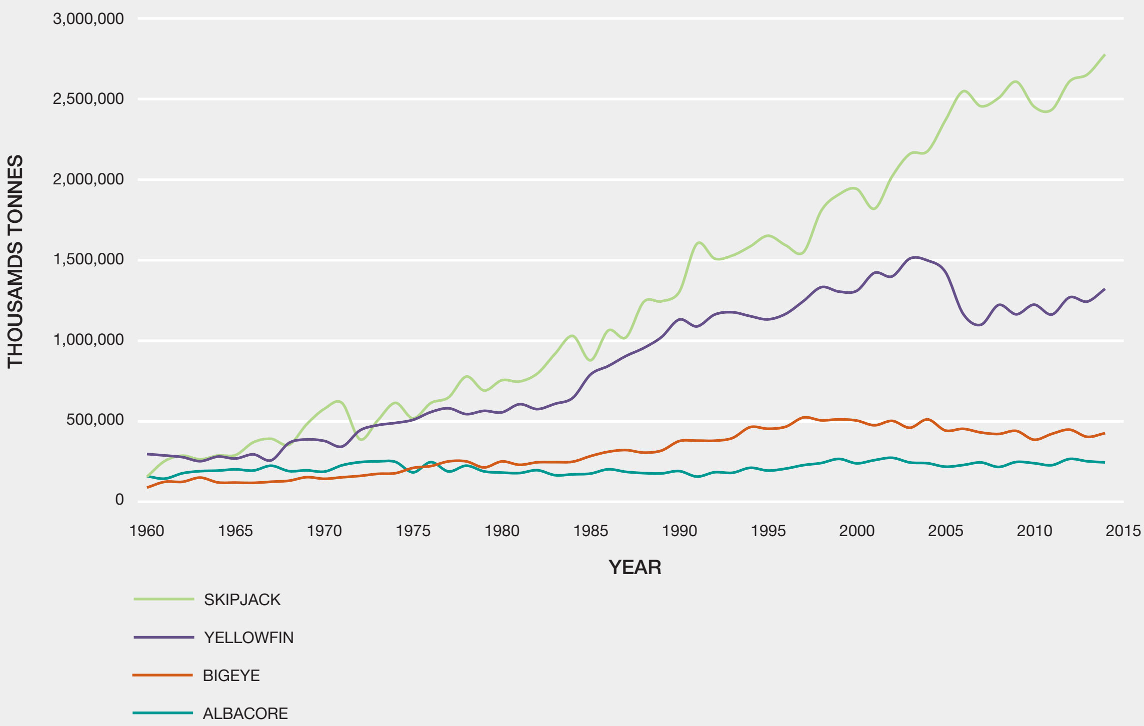


Figure 3 24 **Global reported landings of principal market species of tunas by region, 1960–2014.**

Source: (Gilman, Allain, Collette, Hampton, & Lehodey, 2016; SPC, 2015)

2017). Direct mortality caused by pelagic marine fisheries is the main driver of reductions in the size and abundance of pelagic apex predators, including target stocks and incidentally caught species. Many target species are considered to be above limit thresholds and near targets. However, as discussed earlier in this chapter, the fisheries that catch these principal market species also intentionally or accidentally capture species that are highly vulnerable to anthropogenic mortality sources. There is extremely high uncertainty of the status of stocks and populations of these other species.

Fisheries that target tuna and tuna-like species, billfishes and other relatively fecund species can have large impacts on incidentally caught species that, due to their lower reproduction rates and other life history traits, are relatively vulnerable to increased mortality. This includes seabirds, sea turtles, marine mammals, elasmobranchs and some teleosts (Branch, Lobo, & Purcell, 2013; E. L. Gilman, 2011; M. A. Hall, Alverson, & Metuzals, 2000). Pelagic fisheries selectively remove individuals based on certain traits (e.g., behavioral traits for boldness; life-history traits for size-at-age; physiological traits for visual acuity; morphological traits for mouth dimensions), reducing intraspecific genetic diversity and altering fitness and evolutionary processes (Heino, Díaz Pauli, & Dieckmann, 2015; Hollins *et al.*, 2018). Fishing gear can alter and damage habitat (Dagorn, Holland, Restrepo, & Moreno, 2013; Escalle, Brouwer, Phillips, Pilling, & PNA, 2017)). Thus, fisheries targeting large, highly migratory pelagic predators of high trophic levels (total lenght > 4.0) indirectly modify trophic food web structure and processes and functionally-linked systems (J. A. Estes *et al.*, 2011; Pace, Cole, Carpenter, & Kitchell, 1999; Polovina, Abecassis, Howell, & Woodworth, 2009; J. Stevens, 2000; Ward & Myers, 2005). At this latter broad level, there is limited understanding of what magnitudes of interacting natural (e.g., large scale climate variability) and anthropogenic pressures (including from fishing) cause pelagic ecosystems to reach a tipping point where they undergo a protracted or permanent regime shift, and how altered components of the state of pelagic ecosystems affect functionally-linked systems (**Box 3.3**; (Ortuño Crespo & Dunn, 2017; Pace *et al.*, 1999).

Of the 23 stocks of the seven principal market tuna species, 9 have biomass levels that are below a level estimated to produce maximum sustainable yields or similar thresholds. The fishing mortality rate exceeds a maximum sustainable yield-based or similar reference point, indicating that the stock is not rebuilding its biomass, or both (ISSF, 2016).

Box 3 3 Ecosystem effects resulting from combined natural and anthropogenic impacts and their influence on the fisheries.

Although populated and exploited since the Neolithic, the Black Sea has undergone dramatic ecosystem changes in the last half century, mainly related to anthropogenic impacts such as uncontrolled fishing, cultural eutrophication and invasions by alien species. Fisheries collapses, harmful algal and jellyfish blooms, benthic community loss, and upper shelf hypoxia have had dire consequences for ecosystems and human livelihood depending on them. Recent research studies (G. Daskalov, 2003; Daskalov *et al.*, 2017; Oguz & Gilbert, 2007) have demonstrated that these major changes resulted from synergistic effects of climate forcing, trophic interactions and anthropogenic pressures (overfishing, eutrophication and introduction of invasive species).

Historical trends in fishing and environmental change in the Black Sea

Following the development of the fisheries, the pelagic top-predators have declined by the early 1970s in the Black Sea. For instance, the large population of dolphins diminished about tenfold through overexploitation (Özturk, 1996; Sirotenko, Danilevskiy, & Shlyakhov, 1979). Before 1970, the fishery targeted mainly large, valuable migratory species, such as bonito, mackerel, bluefin tuna and swordfish. All of these important fisheries collapsed mainly due to heavy and unregulated fishing (Daskalov, Demirel, Ulman, Georgieva, & Zengin, 2020; Daskalov, Prodanov, & Zengin, 2008). In the early 1970s, the stocks of planktivorous fishes (sprat, anchovy and horse mackerel) increased considerably and became a target for the industrial fishery (Barange *et al.*, 2009). Their increase in biomass and catch promoted the expansion of powerful trawl and purse seine fishing fleets and a steady increase in fishing effort (Gucu, 1997). The highest catch and fishing mortality were recorded in the late 1980s, but biomasses of exploited populations were declining due to recruitment failures in the previous years. Sharp reductions in biomass and catch in the early 1990s were described as stock collapses (Daskalov *et al.*, 2008). After 1990, the fishing effort decreased and a slow recovery of small pelagic fishes occurred during the 2000s (Daskalov *et al.*, 2017). Starting in the 1970s several human activities further induced a deterioration of the environmental conditions. Intensive bottom trawling on the shelf provoked dispersal of sediment, which severely decreased water transparency, and its re-sedimentation buried demersal life under thick silt layer (Samyshev & Rubinstein, 1988). Increased nutrient loading from rivers and coastal sources (Zaitsev & Mamaev, 1998) favoured frequent plankton blooms, equally contributing decreasing transparency and ventilation leading to benthic life kills. The degradation of massive phytoplankton blooms by aerobic bacteria that pump oxygen from the water further promoted hypoxia, especially near the bottom. By the 1990s, biological invasion of the ctenophore *Mnemiopsis leidyi* (brought in ship ballast water) has contributed to depletion of zooplankton and collapses of small pelagic fisheries (Knowler, 2005).

Driving factors and mechanisms of trophic cascades and regime shifts

Ecosystem shifts cascading down from top-predators to primary producers and affecting water quality were registered along the 1970s and 1990s (Daskalov *et al.*, 2008). The first shift followed the depletion of top predators from the 1950–1970, after which the ecosystem stabilized at low abundance of top predators, high abundance of planktivores, low zooplankton biomass and high phytoplankton biomasses during the 1970s and 1980s. The second shift was associated with the collapse of planktivorous fish and outburst of *M. leidyi* in the early 1990s, which resulted in a second system-wide trophic cascade, with similar alternating effects on zoo- and phytoplankton, and on water chemistry. Overfishing was recognised as the structuring factor affecting not only fish stocks, but the whole ecosystem and held responsible for the system shifts to unhealthy states (Daskalov *et al.*, 2008). Overfishing also contributed to hypoxia by cascading increase of phytoplankton and subsequently bacteria activity. Regional and global climate change, eutrophication, and invasive species were also reported to synergistically contribute to ecosystem shifts (Daskalov *et al.*, 2017; Oguz & Gilbert, 2007).

Effects of trophic cascades and regime shifts on fisheries

The cascading shifts have affected the whole food web from top-predators to primary producers, with repercussions on water chemistry (Daskalov *et al.*, 2008). The environmental degradation has naturally affected fish stocks and fisheries relying on them (Daskalov *et al.*, 2008; Zaitsev & Mamaev, 1998). The effect of 1970s trophic cascade on fisheries catches has been positive as small pelagic stocks boomed after being released from predation. The 1990s shift however entrained small pelagic stock and fisheries collapses and substantial socio-economic losses (Knowler, 2005). Although recovery of previous states is unlikely, some components of the ecosystem have been subject to partial recoveries (Daskalov *et al.*, 2017). The overall state on the marine environment has improved with the reduction of the nutrient load, partial control over *M. leidyi*, and more intense turnover rates related to warmer sea water. Following reduction in the fishing pressure, stocks and catches of small pelagic species recovered to intermediate levels, but large valuable species such as turbot, bonito and bluefish remain scarce according to historical abundances. Current single-species based management practices seem insufficient to deal with consequences of ecosystem regime shifts. At present the existing management bodies at national and international levels fail to implement ecosystem-based management. Recovery of resilient ecosystems should mean restoring all important components (including top-predators) into a desirable state with reduced anthropogenic impacts, normalized species interactions, buffered trophic cascades, increased biodiversity and improved environmental quality. This ecosystem state would provide strategic benefits, such as a clean marine environment, abundant and diverse fish stocks and sustainable economic activities (e.g., fishing, tourism), to a range of stakeholders and society as a whole.

Most tuna stocks are either under-exploited or fully-exploited, dominated by skipjack, albacore and yellowfin tunas. As discussed above, while the use of some of these principal market species is considered sustainable when assessed against certain metrics such as the FAO's definition of overexploited (3.3.1), vulnerable species bycatch accompanies the fishing activity. As political attention to problematic bycatch in marine capture fisheries has increased over recent decades, more resources have been allocated to assess the status of incidentally captured stocks and populations that are of relatively high risk, including, for example, silky and oceanic whitetip sharks, false killer whales, leatherback and loggerhead sea turtles, and several pelagic seabirds including albatrosses and large petrels. These assessments have included semi-quantitative ecological risk assessments using productivity-susceptibility analysis that informs the relative risk of affected stocks and populations, and quantitative, model-based and data-intensive stock assessments and population models that provide information on the absolute risks to affected stocks and populations.

The International Union for Conservation of Nature Red List global species-level categorizations do not provide information on the status of individual populations/stocks. Of the 61 species belonging to Suborder Scombroidei, species assessed against the International Union for Conservation of Nature Red List criteria, 13% were listed as Threatened and 7% as Near Threatened (Collette *et al.*, 2011; IUCN, 2014). Of the Scombroidei, Pacific bluefin, Southern bluefin, Atlantic bluefin and bigeye tuna were categorized as Threatened. The characteristics that these four species of threatened tunas have in common are long generational lengths, longer-lived and later maturity. When combined these traits results in longer time to recover from population declines (Collette *et al.*, 2011). These threatened tuna species also have higher economic values per unit of weight relative to the other market tunas (Miyake *et al.*, 2010).

While there were some early concerns over their application to exploited fishes this largely reflects a misunderstanding of how the criteria work (Mace & Hudson, 1999; Reynolds & Mace, 1999). The International Union for Conservation of Nature Red List assessments are global in scale whereas fisheries assessments are regional in scale and hence these different assessment processes are used for different purposes. However residual concerns about the applicability of the International Union for Conservation of Nature criteria have been refuted by extensive empirical evidence that consistently show strong alignment and harmony with fisheries management reference points, based on simulations using data from the global population dynamics database (Connors, Cooper, Peterman, & Dulvy, 2014) and multiple global meta-analyses of all fisheries stock assessments (e.g. Davies & Baum, 2012; d'Eon-Eggertson, Dulvy, & Peterman, 2015; P. G. Fernandes *et al.*, 2017; Porszt, Peterman, Dulvy, Cooper, & Irvine, 2012). The greatest concerns were raised for the highly fecund broad cast-spawning fishes, yet the International Union for Conservation of Nature Red List categories and criteria have been shown to highly aligned with fisheries assessment. As a result, marine fishes assessed by the International Union for Conservation of Nature as being Endangered or Critically Endangered are consistently fished beyond target and limit reference points (Dulvy, Jennings, Goodwin, Grant, & Reynolds, 2005; Simpfendorfer & Dulvy, 2017). For species that are not subject to fisheries assessments, the International Union for Conservation of Nature assessments offer valuable information on the need for fisheries management (ICES, 2018).

3.3.1.4.5 Whaling

Aquatic mammals are an important hunting target species for subsistence, culture and identity of some indigenous and local communities (IWC, 2021; S. L. Newell & Doubleday, 2020) **(Box 3.4)**. Across South America and West Africa hunted aquatic mammals includes 33 small cetaceans and all three manatee species (Cosentino & Fisher, 2016; Porter & Lai, 2017). The vast majority of whales hunted for aboriginal subsistence in the United Kingdom of Denmark, Norway and Iceland (**Figure 3.25A**, (International Whaling Commission, 2021)) are common minkes. Besides, Greenland (United Kingdom of Denmark) has been conducting aboriginal subsistence whaling targeting fin, bowhead and humpback whales as well as commercial whaling targeting narwhal and other small cetaceans. Faroe Islands (United Kingdom of Denmark) has been conducting the drive fishery targeting pilot whales. Norway and Iceland have been conducting commercial whaling on fin whales. In these countries local hunters often sell whale meat to foreign tourists or in European Union markets (Eklund T., 2017). Indigenous communities in the Russian Federation mostly hunt the gray whale (IWC, 2019a) and in the United States of America, the bowhead whale (IWC, 2019b). In India, Pakistan and Sri Lanka cetaceans are also hunted (often illegally) for use as bait in other fisheries (Porter & Lai, 2017).

Aquatic wild species can also be utilized on a commercial basis **(Figure 3.25)**. Since 1982, the International Whaling Commission which regulates commercial whaling has maintained a "zero quota" on commercial whaling (with the exception of catches set by countries under objection or reservation) because of historical overexploitation and the challenge of managing whaling sustainably. The organization currently has 88 members. Japan suspended commercial whaling in 1988 and began whaling for scientific research in 1987 to gather population data in accordance with the paragraph 10e of the Schedule of the International Convention for the Regulation of Whaling (Cosentino &

Box 3 4 Small-scale indigenous whaling in the North.

Many northern Indigenous peoples continue traditional whale hunting, a practice dating back centuries or more (Stoker & Krupnik, 1993). Whaling provides substantial quantities of food, is a central part of community activities and culture, and a source of fulfillment and identity (Sakakibara, 2020). Collaborative hunts and sharing of the products promote social cohesion, an essential component of thriving in a challenging environment (Huntington *et al.*, 2021). In some places whale products are sold in local markets, which occasionally creates conflicts among users (Sejersen, 2001), but rarely leading to excessive exploitation. The legacy of large-scale commercial whaling continues to affect some whale populations in the Arctic, but most stocks appear to have recovered and concerns about unsustainable takes at present are limited (Givens & Heide-Jørgensen, 2021; NAMMCO, 2018).

The bowhead whale (*Balaena mysticetus*); (Huntington *et al.*, 2021; Suydam & George, 2021) is hunted primarily by Iñupiaq and Yupik whalers in Alaska under a quota of 67 whales per year established by the International Whaling Commission based on population status and also cultural need. An annual hunt is conducted by Inuit in Nunavut of about one whale per year, with the hunt rotating among communities. The bowhead is also occasionally hunted in Chukotka, Russia, and in Greenland. The beluga whale *(Delphinapterus leucas)* is hunted in Greenland, Canada, Alaska, and Chukotka by Inuit, Inuvialuit, Gwichin, Iñupiat, Yupik, Yup'ik, and Chukchi. Worldwide, over 1000 beluga are taken per year on average, and the hunt is regarded as sustainable in nearly all locations (Hobbs *et al.*, 2019; NAMMCO, 2018). The narwhal *(Monodon monoceros)* is hunted in Canada and Greenland by Inuit (Lee, 2017). The worldwide annual harvest is similar to that for beluga whales and is considered sustainable for most populations currently hunted (Hobbs *et al.*, 2019; NAMMCO, 2018). Chukchi and Yupik whalers in Chukotka hunt about 125 gray whales per year (*Eschrichtius robustus*; (IWC, 2019a)) under an Illegal Whaling Commission quota. The harvest is considered sustainable. In 1999, Makah whalers in the American state of Washington resumed a cultural tradition of gray whale hunting that had been interrupted by colonization and its disruptions, but since 2002 domestic regulations have prevented the hunt from taking place (IWC, 2019b). In Greenland (IWC, n.d.), hunters take approximately 150 minke whales *(Balaenoptera acutorostrata)*, 11 fin whales *(Balaena physalus)*, and 7 humpback whales (*Megaptera novaeangliae*) per year. All of these Greenland large whale harvests are under an Illegal Whaling Commission quota and are considered sustainable. In addition to larger cetaceans, some dolphins and porpoises are taken in Arctic communities. Although not indigenous, Faroe Islanders in the North Atlantic hunt long-finned pilot whales *(Globicephala melas)* each year (Statbank, 2020), a small-scale traditional harvest dating back centuries, which has averaged around 650 whales per year over the last decade.

Fisher, 2016). In accordance with the provisions of the International Convention for the Regulation of Whaling (Article 8), all meat taken from whales caught for scientific whaling was processed and sold in stores and restaurants, and the proceeds obtained from the sales were used for the research activities in the following years in accordance with the direction by the Government of Japan. The International Court of Justice, using various criteria, ruled that Japan's whaling was not "for purposes of scientific research" as required by Article VIII of the International Convention for the Regulation of Whaling, and ordered Japan to immediately cease its JARPA II whaling program (JARPA II: second phase of Japan's whale research program under special permit in the Antarctic) (Clapham, 2015).

In 2019, Japan withdrew from the International Convention for the Regulation of Whaling, in line with Japan's basic policy of promoting sustainable use of aquatic living resources based on scientific evidence, and resumed commercial whaling after 31 years of suspension (Holm, 2019). Norway and Iceland are members of the International Whaling Commission, but have continued to commercially hunt whales either under objection to the moratorium decision or under reservation to it (IWC, 2021). The Russian Federation has also objected to the moratorium, but has not resumed whaling. Countries members of the Illegal Whaling Commission that take whales are obliged to provide statistical, scientific and other pertinent information to the International Whaling Commission. While the Western North Pacific stock of common minke and Bryde's whales are confirmed by the Illegal Whaling Commission Scientific Committee to be relatively abundant, abundance estimate of North Pacific stock of sei whale is still under examination by the Illegal Whaling Commission Scientific Committee although it has been substantially recovered. Thus, sei whales as a whole are still classified as endangered by the International Union for Conservation of Nature.

In 2018, despite a substantial number of opposition, the International Whaling Commission adopted a resolution which reaffirms "that the moratorium on commercial whaling, which has been in effect since 1986, has contributed to the recovery of some cetacean populations, and aware of the cumulative effects of multiple, existing and emerging threats to cetacean populations such as entanglement, bycatch, underwater noise, ship strikes, marine debris and climate change" and "agrees that the role of the International Whaling Commission in the 21st century includes inter-alia its responsibility to ensure the recovery of cetacean populations to their pre-industrial levels, and in this context reaffirms the importance in maintaining the moratorium on commercial whaling" (**Figure 3.25 A, B** and **C**).

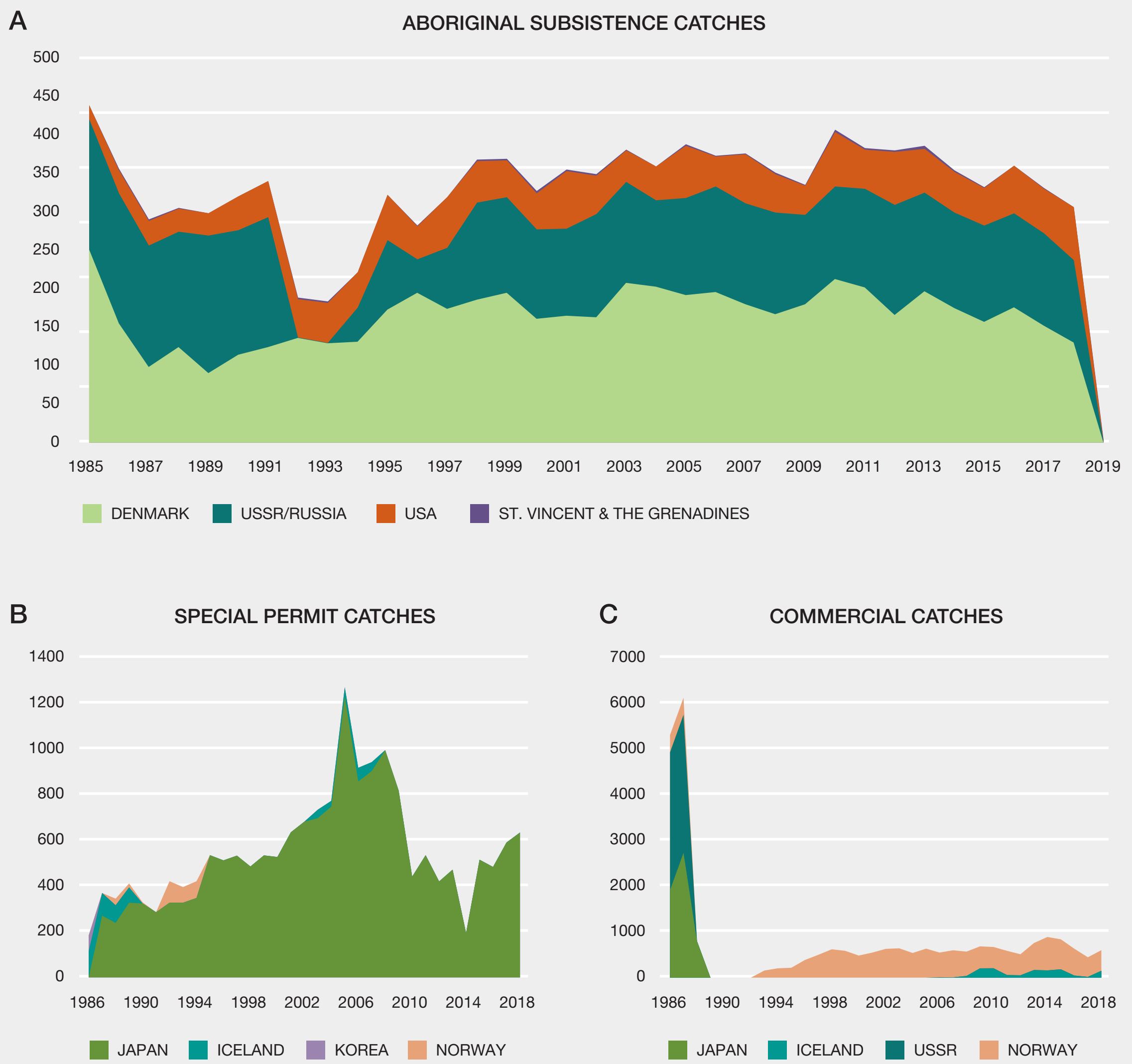


Figure 3 25 **Global catches of whales according to the International Whaling Commission, 1985–2018.**

(A) Aboriginal subsistence catches **(B)** special permit catches **(C)** commercial catches taken under objection or reservation. CC-BY-NC. See data management report for the figure at https://doi.org/10.5281/zenodo.6453071.

3.3.1.4.6 Industrial demersal fisheries in coastal areas

The status of demersal fisheries in temperate countries is well documented in the RAM Legacy Stock Assessment Database (Christopher Costello *et al.*, 2016; RAM Legacy Stock Assessment Database, 2018; Ricard *et al.*, 2012), where approximately 53% of global reported catch is counted. Those consist of three dominant taxonomic groups, gadids (cod, haddock, pollock and hake), pleuronectids (flatfish), and sebastids (rockfish). While many of these fisheries underwent a historical phase of overexploitation, recent evidence suggests that many of these fisheries have been managed since the 1990s and 2000s in ways that reduced fishing mortality rates (Christopher Costello & Ovando, 2019). In many cases these measures improved stock status (Hilborn & Ovando, 2014; B. Worm *et al.*, 2006) and increased biomass to the point that some authors now focus on underfishing of some key stocks (Hilborn, 2019). The **Figure 3.26** shows the trend in abundance and fishing mortality for these species in temperate areas (Europe, North America, Japan, Chile, New Zealand and Australia). Stock abundance tends to be above the level that would produce long-term maximum sustainable yield and fishing pressure is lower. This has resulted in increasing general stock abundance **(Figure 3.26)**.

Figure 3 26 **Stock abundance trends.**

These figures describe the trend in ratios of B/Bmsy (Biomass relative to the biomass that produces Maximum Sustainable Yield), U/Umsy (Fishing pressure or mortality relative to the fraction of the population harvested), and catch/(mean catch) across assessed stocks in the taxonomic group. Source: (Hilborn *et al.*, 2021) under license CC-BY_NC-ND 4.0.

The status of demersal fisheries in the rest of the world is much less documented. A quarter of the remaining global reported catch has undergone some form of data-limited stock assessment (FAO, 2016b) while 22% remains unassessed, with little information about population status or risk of over-fishing (Christopher Costello & Ovando, 2019). These data limited stocks make up an increasing proportion of globally reported catch over time, from 20% to 47% in the last 60 years (Vasconcellos & Cochrane, 2005). From two areas for which information is available, the Mediterranean and Western Africa, the evidence is that these stocks are very heavily exploited and almost certainly over-fished and subject to over-fishing (Hilborn *et al.*, 2020).

The demersal species from the regions not well covered in the RAM Legacy Stock Assessment database, along with the small pelagic fisheries of the same regions, constitute the dominant component of the unassessed fish stocks of the world.

Most of those demersal stocks belong to coastal fisheries which contribute much, if not most, of global catches, but quantitative estimates of the extent of their contribution depend on how coastal fisheries are defined, especially in relation to small scale fisheries. Palomares and Pauly (Palomares & Pauly, 2019) used the “Sea Around Us” reconstructed catch database (Zeller *et al.*, 2016) to

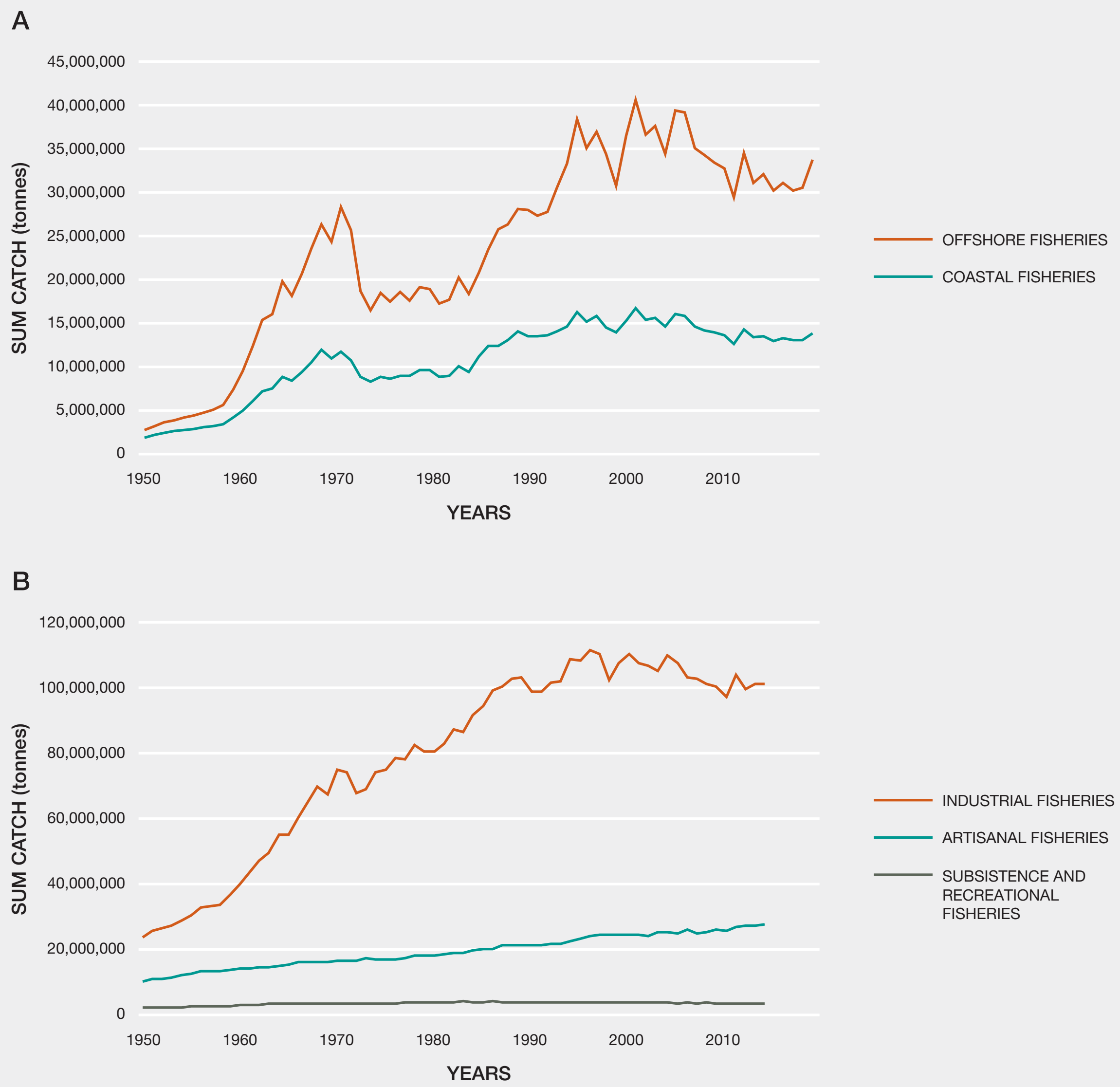


Figure 3 27 **Catches from 1950–2014 of (A) the world's marine fisheries (B) coastal fisheries.**

Areas <=50 km from inhabited coastlines or to a depth of <=200 m. Based on (Palomares & Pauly, 2019) under license number 5278770226780. CC-BY NC.

estimate the catch in an area at most 50 km from inhabited coastlines or down to a depth of 200 m **(Figure 3.27)**, considered to be the area in which small scale fisheries (artisanal, subsistence, and recreational) are located. Coastal fisheries made up an average of 55% of global marine fisheries in the 5-year period from 2010 to 2014, while small-scale fisheries in the same period contributed 36% of the marine catches consumed directly by people **(Figure 3.28)**.

Lower-income countries lack the capacity to industrially harvest fish populations off their shores, and thus frequently host foreign fishing fleets through fishing access agreements or joint venture operations (Belhabib *et al.*, 2015; Kaczynski & Fluharty, 2002). The higher capacity and improved technology of higher-income nations has enabled these countries to build and operate distant water fishing fleets, and often to subsidize those fleets heavily (Sala, Aburto-Oropeza, Reza, Paredes, & López-Lemus, 2004; Dirk Zeller & Pauly, 2019). Describing fishing patterns of those industrial fleets in comprehensive and quantitative terms is challenging due to the lack of open access to detailed records on the behavior of fishing vessels. However, McCauley *et al.*, (McCauley *et al.*, 2018) produced fishing patterns of industrial fishing vessels (>24m) based on high-resolution fishing vessel activity information derived from automatic

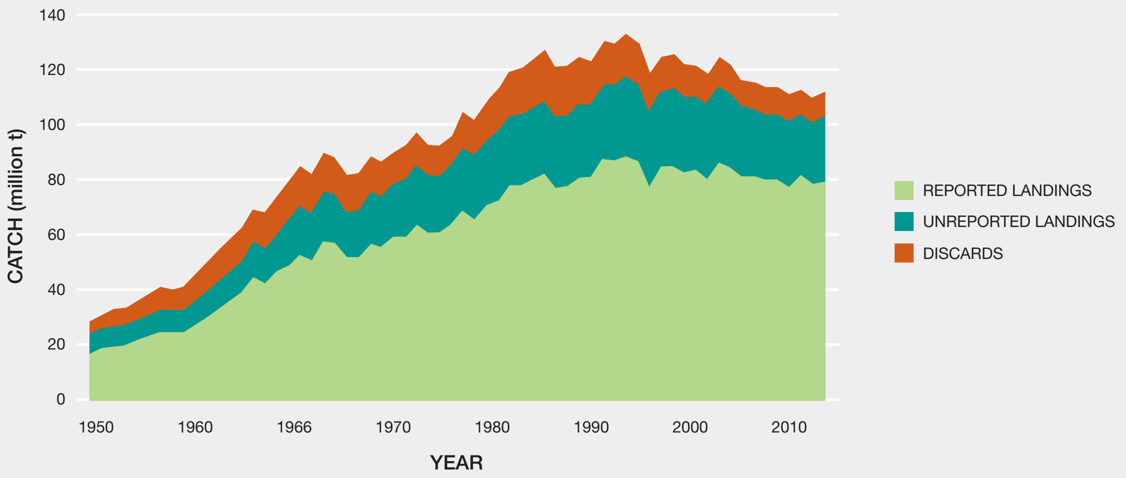


Figure 3 28 **Global catch data as reported to the Food and Agriculture Organization of the United Nations by fishing countries.**

Reported catch: black line) (1950–2016). Source:(Dirk Zeller, Cashion, Palomares, & Pauly, 2018) under license CC-BY NC.

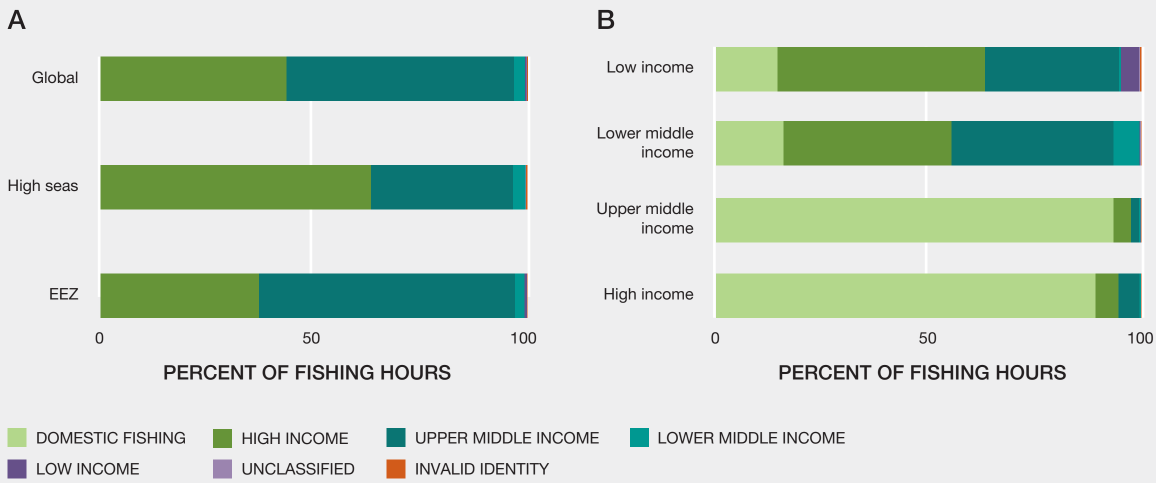


Figure 3 29 **Distribution of industrial fishing effort by vessels flagged to nations from different income classes as measured using automatic identification systems data and convolutional neural network models.**

(A) The percent of fishing effort (measured in fishing hours) detected globally on the high seas and in all exclusive economic zones for vessels flagged to nations from four different World Bank income groups. **(B)** The percent of automatic identification systems-detected industrial fishing effort in all exclusive economic zones, grouped by the World Bank income groups of the exclusive economic zones. Here, the category Domestic fishing is included, which refers to instances when a fishing country was fishing in its own exclusive economic zone. Other categories represent foreign fishing effort conducted within an exclusive economic zone by a nation flagged to one of the four World Bank income classes. “Invalid identity” refers to vessels with a Maritime Mobile Service Identity (MMSI) number that did not accurately refer to an individual country. “Unclassified” refers to fishing entities that were fishing in an exclusive economic zone but did not have a World Bank income group. All data presented here are summarized from the year 2016. Source: (McCauley *et al.*, 2018) under license CC BY 4.0.

identification systems data (**Figures 3.29** and **3.30**). Such patterns address one of the fundamental issues of fisheries sustainability, namely direct and collateral impacts by fishing gear on habitats, target and non-target species (Amoroso *et al.*, 2018, 2018; Lewison *et al.*, 2004a; Palomares & Pauly, 2019), directly related to the amount of gear deployed rather than to the amount of target yield extracted coming from catch data (Stewart *et al.*, 2010).

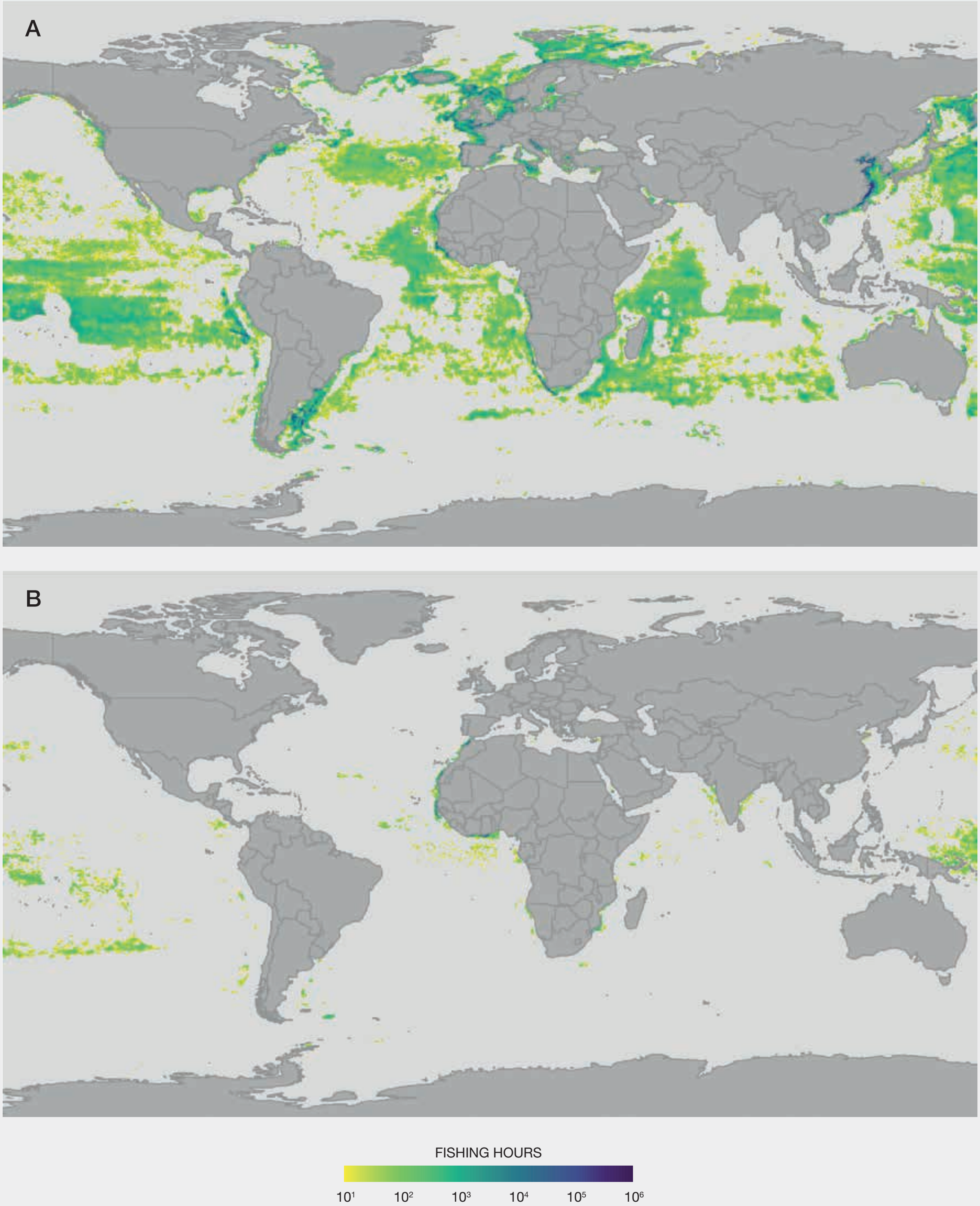


Figure 3 30 **Density distribution of global industrial fishing effort, derived using automatic identification systems data.**

(A) Vessels flagged to higher-income countries and **(B)** vessels flagged to lower-income countries. Industrial fishing effort is estimated using convolutional neural network models and plotted as the log10 number of fishing hours. *This map is directly copied from its original source (McCauley et al., 2018) and was not modified by the assessment authors. The map is copyrighted under license CC BY 4.0. The designations employed and the presentation of material on the maps used in the assessment do not imply the expression of any opinion whatsoever on the part of IPBES concerning the legal status of any country, territory, city or area or of its authorities, or concerning the delimitation of its frontiers or boundaries. These maps have been prepared or used for the sole purpose of facilitating the assessment of the broad biogeographical areas represented therein and for purposes of representing scientific data spatially.*

The density distribution of global industrial fishing effort reveals global dominance of industrial fishing by wealthy nations (www.worldbank.org; using 2016 classifications). Vessels flagged to higher-income nations are responsible for 97% of the trackable industrial fishing on the high seas and 78% of such effort within the national waters of lower-income countries (McCauley *et al.*, 2018).

While legal, these arrangements raise many challenges regarding their sustainability and equity. For instance, the expected benefits of these partnerships, such as revenues and investments in local infrastructure and technologies, have not always materialized (Antonova, 2016; Crona *et al.*, 2016). Distant water fleets are also involved in illegal, unreported and unregulated fishing (Pauly *et al.*, 2014), which are considered as a serious threat to fisheries and fisheries-dependent communities, marine ecosystems and societies at large (Hutniczak, Delpeuch, & Leroy, 2019). Agnew *et al.* (2009) estimated that 11–26 million tons (from exclusive economic zones and high seas), or roughly one-quarter of the world catch of fish goes to illegal, unreported and unregulated fishing every year. The same authors found a correspondence between their regional estimates of illegal and unreported fishing and the number of depleted stocks in those regions. As exemplified in the case study in **Box 3.5**, the relationship between industrial fisheries, small scale fisheries, population status, food security, and livelihoods is a complex one indeed.

Box 3 5 **Bottom trawling: assessing seabed habitat and biota impacts.**

The recognition that sustainability of fisheries not only involves maintaining target stocks at productive levels, but also minimizing wider ecosystem impacts of fishing has turned increasing attention to the evaluation of the environmental footprint of different fishing methods. In particular, the use of bottom-contact mobile gears as a means of catching fish has sparked heated debates in fishery and conservation sciences. On the one hand, bottom trawling contributes close to 20 million tons of fish and invertebrates per year to the global food supply and provides food and livelihoods for millions of people as well as significant export revenues to many countries (Amoroso *et al.*, 2018). On the other hand, bottom trawling impacts seabed habitats, damaging biogenic structures and altering sediment composition and its biogeochemical dynamics, kills benthic organisms and alters ecosystem functions (Clark *et al.*, 2016; De Borger, Tiano, Braeckman, Rijnsdorp, & Soetaert, 2021; Hiddink *et al.*, 2017; O'Neill & Ivanović, 2016; Pusceddu *et al.*, 2014) (Pusceddu *et al.* 2014, Clark *et al.* 2016, O'Neill and Ivanović 2016; Hiddink *et al.* 2017, De Borger *et al.* 2021). Concerns about environmental impacts of bottom trawling have fueled strong public campaigns and resulted to its ban in some countries and regions. Less extreme approaches for reducing the negative impacts of trawling have been pursued, including changes in gear design and fishing operations, prevention of further expansion of trawled area, ocean zoning, bycatch and habitat quotas and the closure of large areas to protect sensitive habitats (McConnaughey *et al.*, 2020; Williams *et al.*, 2020). United Nations General Assembly Resolutions 61/105 (2007) and 64/72 (2010) required Regional Fisheries Management Organizations to identify vulnerable marine ecosystems on the seabed within their jurisdictions and ensure that fisheries did not cause serious adverse impacts to them. Of particular concern has been the expansion of trawling into deeper areas, leading for example to the ban on bottom trawling in deep waters (below 800 m) and in areas with vulnerable marine ecosystems (below 400 m) adopted by the European Union in 2016.

Assessments of the global and regional seabed impacts of bottom trawling require information on the distribution and intensity of trawling, the direct impact of the gear on the swept habitats and communities, and their capacity to recover from trawling disturbances (Mazor *et al.*, 2021; McConnaughey *et al.*, 2020; Pitcher *et al.*, 2017). A study using high-resolution satellite vessel monitoring system and logbook data on 24 continental shelves and slopes to 1,000-m depth (covering 7.8 million-km^2 in total) showed that 14% of the overall study area was trawled and 86% was not trawled (Amoroso *et al.*, 2018). However, the seabed proportion impacted by trawling varied markedly among and within regions, from less than 1% in southern Chile to a maximum of 80% in the Adriatic Sea and from areas (within region) trawled several times per year and others only disturbed sporadically. Trawling activity was aggregated; the most intensively trawled areas accounting for 90% of activity comprised 77% of footprint on average trawled (R. Amoroso *et al.*, 2018). In most heavily trawled areas of Europe a large fraction of the area (e.g., North Sea, West Iberia and Skagerrak and Kattegat) was trawled at least once per year, while more than half of the seabed was not trawled during the 2-6-year study period in 20 of 24 regions examined. Trawling footprints were also smaller in regions where fishing rates met sustainability benchmarks trawled (Amoroso *et al.*, 2018).

To evaluate biotic impacts, the frequency of trawling events further needs to be compared to the rate of recovery of the different types of organisms inhabiting seabeds. Recent meta-analyses of more than three decades of published results for sedimentary habitats have shown that the immediate mortality of animals in the path of the trawl is correlated with the penetration depth of the gear in the sediment, which vary with the type of gear (Hiddink *et al.*, 2017; Sciberras *et al.*, 2018). The most commonly used trawl gear (otter trawls) kills 6% of the biomass per pass, whereas the most destructive gear (hydraulic dredges) kills 41% of the seabed biota present. Estimated recovery rates after trawling ranged from 1.9 to 6.4 years on average, depending on the type of sediment, trawl gear and benthic species longevity (with longer-lived animals showing larger depletion effects in comparative studies (Hiddink *et al.*, 2019, 2017)). Repeated trawling would thus induce a

Box 3 5

shift toward species with faster life histories in communities exposed to frequent trawl events (Hiddink *et al.*, 2017; Jennings & Cotter, 1999). A reduction in median longevity of the community of close to 20% on average was estimated for the relatively heavily trawled North Sea (McConnaughey *et al.*, 2020). Selective effects linked to chronic trawling are likely to be much stronger for long-lived sessile epifauna, such as sponges and corals (Hiddink *et al.*, 2017).

By combining known distribution of trawl intensity from Amoroso *et al.* (Amoroso *et al.*, 2018) with predicted abundance distributions of different benthos groups for 13 diverse regions of the globe, Mazor *et al.* (2021) found that expected benthic community status ranged between 86% and 100% of untrawled status (mean 99%), with more than three-quarters of benthic groups predicted to be at 95% or more of their benchmarks. Mean benthos status was lowest in regions of Europe and Africa and for taxonomic classes Bivalvia and Gastropoda. Communities prevalent in sedimentary habitats of the continental shelves could thus sustain moderate levels of trawling, provided that target fishing mortalities are maintained within accepted sustainability benchmarks. Biogenic habitats, such as coral reefs, maerl beds and sea mounts habitats (not covered by (Hiddink *et al.*, 2017)) are nonetheless expected to be the much more sensitive to trawling impacts due to their long recovery times. The limited data available for long-lived habitat-forming species indicate that post-trawling recovery may take decades (Kaiser, Hormbrey, Booth, Hinz, & Hiddink, 2018; Williams *et al.*, 2010) and be unachievable within acceptable timeframes; spatial closures are therefore essential (Clark *et al.*, 2016).

The studies discussed above highlight the importance for policy analysis and implementation of collecting local data on the intensity and distribution of trawling, and the distribution of sediment types and vulnerable marine habitats. These data are needed to identify local best practices and most effective approaches to reduce habitat impacts of fishing, and to allow quantification of trade-offs between fish production for food and the environmental costs associated with different fishing methods and marine policies.

3.3.1.5. Uses of wild caught aquatic organisms

Regarding fishing practices, the following uses are well-documented in the literature and available data sources: food and feed (3.3.1.5.1), medicine and hygiene (3.3.1.5.2), recreational fishing (3.3.1.5.3), decorative and aesthetic (3.3.1.5.4), and ceremony and cultural uses (3.3.1.5.5). The following uses are not relevant to this practice or were not documented: energy, education and learning, and materials and shelter. With regards to non-lethal uses of wild aquatic organisms, a review of catch and release recreational fishing (3.3.1.6.1) and ornamental and aquarium fisheries (3.3.1.6.2) are included.

3.3.1.5.1 Food and Feed

Fish and seafood products are important for human diet, providing about 3.1 billion people with almost 20 percent of their average daily animal protein intake (Sunderland *et al.*, 2019). Human consumption of fish in 2018 totaled 96.4 M tons (FAO, 2020d). Of the landed catch of industrial fisheries, about 80% is used for direct human consumption, and close to 100% of the retained catch of small-scale fisheries is eaten by people (FAO, 2018d). It is important to note that some of these estimates include wild fish and farmed fish from aquaculture. For different indicators available at a global scale, especially fish consumption, much of the available literature does not clearly distinguish between farmed and wild caught fish. Further, in several data sets the information on both wild and farmed fish is so intricately mixed up that it is impossible to distinguish between the two. Indeed, this lack of clarity makes proper assessment of the sustainable use of wild fish species extremely challenging and presents a serious issue for accurate reporting and tracking. This issue is discussed in more detail in the knowledge gaps section (3.5). Thus, despite the focus of this assessment on wild species, assessment experts consistently evaluated the various material they reviewed in relation to two options: (i) select not to use available data and literature on farmed fish and exclude the state of the knowledge on this topic or (ic) include available data on farmed fish in combination with wild fish.

Since 1973, the global consumption of fish has doubled, due to increased demand in developed and developing countries (Delgado, International Food Policy Research Institute, & WorldFish Center, 2003; FAO, 2018d). Consumption grew from approximately 9.0 kg per capita in 1961 to 20.2 kg per capita in 2015, at an average rate of about 1.5 percent per year (FAO, 2018d). A higher rate of 2.4 percent is observed in developing countries for the same period. The growth of world per capita fish consumption from 18 kg in 2008 to 20 kg in 2013 was due to an increase in per capita consumption of freshwater & diadromous fish (migrate between freshwater and saltwater, example: salmon, eel, etc.), crustaceans and shell molluscs, whereas that of marine fish and cephalopods declined (Cai & Leung, 2017).

The importance of global fish production for nutrition and food security varies geographically across regions, countries, and communities dependent on fish at rates far above the global average **(Box 3.6)**. Some of the most fish-dependent populations are located in countries in which the contribution of fish is relatively low at the national level

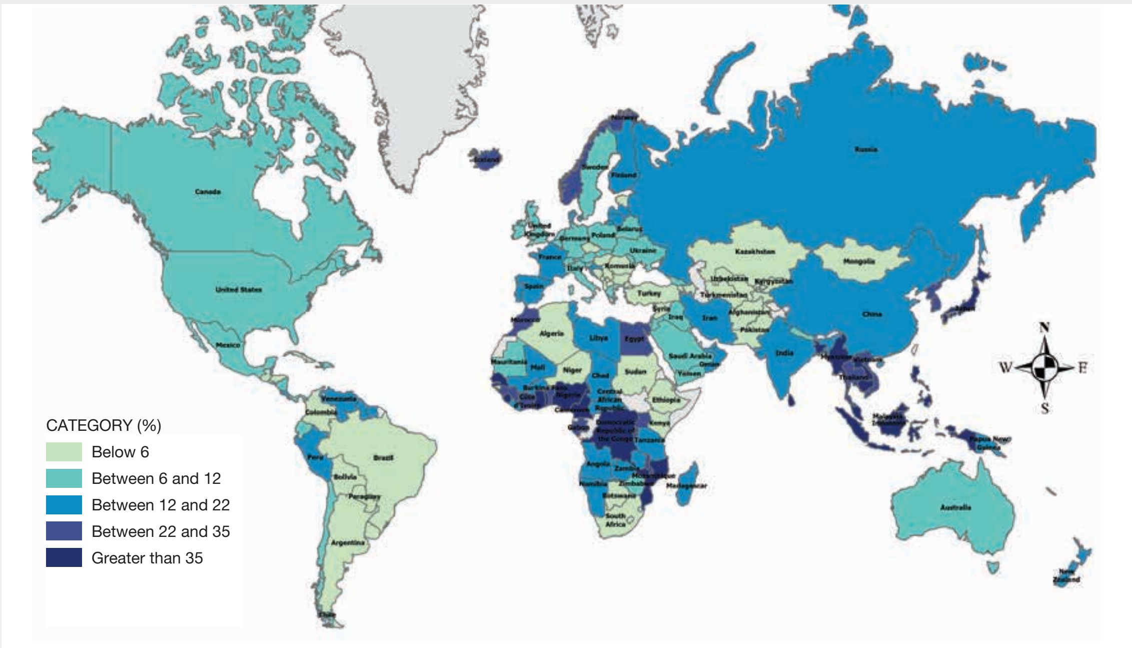


Figure 3 31 **Fish dependency around the world.**

This map is directly copied from its original source (Bennett et al., 2018) and was not modified by the assessment authors. The map is copyrighted under license CC BY 4.0. The designations employed and the presentation of material on the maps used in the assessment do not imply the expression of any opinion whatsoever on the part of IPBES concerning the legal status of any country, territory, city or area or of its authorities, or concerning the delimitation of its frontiers or boundaries. These maps have been prepared or used for the sole purpose of facilitating the assessment of the broad biogeographical areas represented therein and for purposes of representing scientific data spatially.

Box 3 6 **Dried fish in Asian countries.**

Dried fish is an important part of small-scale fisheries (FAO, 2018c; Kawarazuka & Béné, 2010) and includes fish that has been cured, dried, salted, brined, fermented, or smoked fish (see Supplementary material Table S3.1). These are often small and low market value fish from capture fisheries. Approximately 12% of fisheries are prepared and preserved, and 12% are cured. In some countries dried fish consumption is significantly higher (FAO, 2018c).

The voluntary guidelines for securing sustainable small-scale fisheries considers fishing and fish processing as important drivers of food security and poverty eradication (FAO, 2015). In Asia and Africa wide varieties of species are dried (Ruddle & Ishige, 2010), including many small pelagic species (Doe, 2017, see Supplementary material Table S3.1). In Bangladesh, dried fish are eaten more frequently than any other type of fish. The contribution of dried products to total fish consumption is disproportionately important for low-income consumers (Belton & Thilsted, 2014). Although dry fish is not cheap, the quantity needed for a meal is less and therefore economical and may explain popularity in rural areas (Samaranayaka, Perera, & Warnasuriya, 2013).

Dried fish contribute to food and nutrition security in both coastal and arid mountainous regions of low-income countries as they are a concentrated source of animal protein, rich in calcium and other micronutrients and fats, easily transportable and have a long self-life (Belton, Hossain, & Thilsted, 2018; Thilsted, James, Toppe, Subasinghe, & Karunasagar, 2014). For example, in Malawi, a serving of 24 g of small dried fish twice a day provides an intake of calcium, zinc and iron which is 327%, 152% and 22% higher, respectively than a daily diet without fish (R. S. Gibson & Hotz, 2001; Kawarazuka & Béné, 2010). Since low end processing activities are mostly done by women (Samanta, Bhaumik, & Patra, 2016), their control over family income directly affects household food security and nutritional outcomes (Kawarazuka & Béné, 2010). Women have been involved in the dried fish sector in developed countries and regions as well. Historically, and for centuries (until the 1960s) dried fish processing was a major activity in places such as Newfoundland and other Eastern North American locales and was undertaken significantly by women (Doe, 2017; Neis, 1999). Men were engaged in the pursuit and capture of fish; women in the spreading, turning and drying of fish. This produced food and income security for workers in this profession.

Box 3 6

Subsistence and artisanal fisherfolk communities in Asia mostly belong to socially and economically marginalized groups (Hapke, 2001), with those engaged in dried fish activity even more marginalized among fisher communities. Hence, the importance of wild marine species in life and livelihoods of the poorest of the poor is immense. At the same time the fisher communities draw life satisfaction by engaging in fishing activities they find challenging and skillfully providing them with a different identity (Nayak, Dias, & Pradhan, 2021).

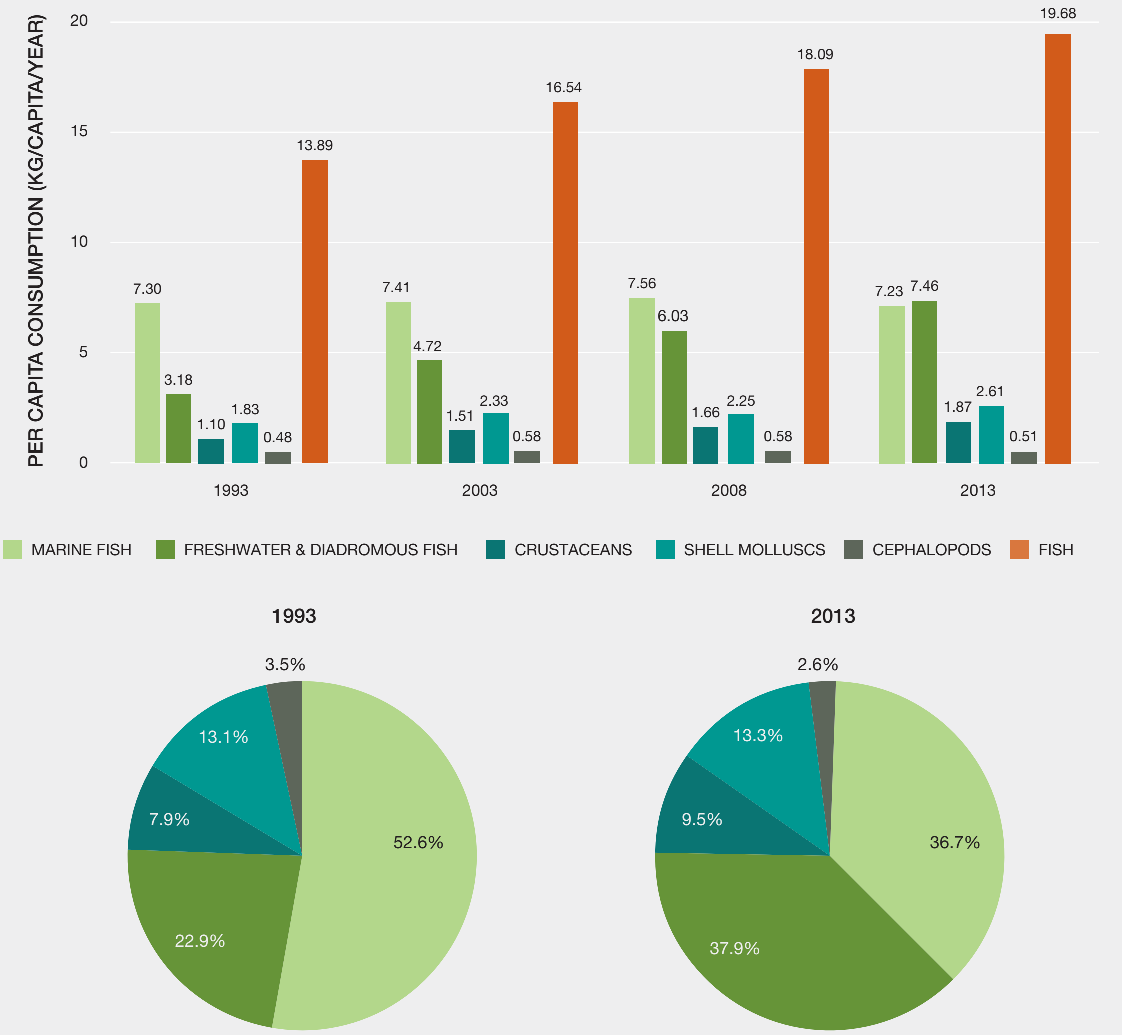


Figure 3 32 **Species composition of world per capita fish consumption.**

Source: (Cai & Leung, 2017) © FAO, 2017. CC-BY NC.

(Bennett *et al.*, 2018). At sub-national scales, individual communities can be almost entirely dependent on seafood for protein. Fish is crucial for coastal indigenous groups, who on average consume fish at a rate that is 15 times higher than the global average **(Figure 3.31)**.

Marine fish used to be the largest species group in world fish consumption, but its share declined from 53% in 1993 to 37% in 2013. Marine fish are still the dominant species consumed in many countries. Indeed, in 2013 marine fish accounted for more than half of fish consumption in more than 170 countries. Over the same period freshwater & diadromous fish consumption grew rapidly, increasing from 3.2 kg in 1993 to 7.5 kg in 2013 (A. Bennett *et al.*, 2018). Crustaceans accounted for nearly 10% of world fish consumption in 2013, increasing from 8% in 1993. Shell molluscs accounted for 13% of world fish consumption in 2013; nearly the same as in 1993. Cephalopods accounted for 2.6% in 2013; down from 3.5% in 1993 (Cai & Leung, 2017) **(Figure 3.32)**.

Hatchery-based aquaculture relies on the use of wild fish as feed. The share of fed species in total aquaculture production accounts for the majority (69.5%) of "food fish" production from aquaculture (Clavelle, Lester, Gentry, & Froehlich, 2019; FAO, 2018d). Capture-based mariculture depends on wild-caught juveniles for "seed," which are then raised and fattened in captivity (Boyd *et al.*, 2020; Ottolenghi *et al.*, 2004). This practice, sometimes referred to as "ranching," is widespread and an important source of production for many species, including tuna, shrimp, lobster, grouper and eels (Lorenzen, Leber, & Blankenship, 2010). However, no current estimates of the extent of capture-based mariculture exist.

Production of fed species depends on feeds containing high concentrations of proteins and lipids traditionally sourced from fishmeal and oil rendered from wild-caught forage fish, such as herring, sardines and menhaden (Tacon, Hasan, & Metian, 2011; Tacon & Metian, 2008b, 2008a, 2015). The total annual production of fish meal was 4.5 million tons, and the total annual production of fish oils was 0.9 million tons in 2016, of which 69% and 75%, respectively, were used in aquafeeds (Hua *et al.*, 2019). An additional 23% and 5% of this fish meal is used in pig and chicken feeds. The aquaculture industry is making important gains in improving feed conversion ratios, reducing the inclusion of fishmeal in feed and developing substitutes (FAO, 2016b; Klinger & Naylor, 2012; R. L. Naylor *et al.*, 2009). Nonetheless, the use of wild fish for feed by the aquaculture sector is increasing as a result of overall growth, intensification of farming practices, and from the rising share of higher trophic level species in total production menhaden (Tacon, Hasan, & Metian, 2011; Tacon & Metian, 2008b, 2008a, 2015) **(Figure 3.33A)**.

Forage fish have been captured and reduced into fishmeal and oil for decades (reduction fisheries), supporting production of terrestrially farmed species, such as pigs and poultry. Aquaculture did not become the dominant user of rendered forage fish until the 2000s, well after global catches of forage fish had plateaued (Shepherd & Jackson, 2013). These pelagic species now help support over 70% of aquaculture production (FAO, 2016b; Tacon & Metian, 2015) acting as feed for carnivorous species (for example, salmon, tuna) and increasingly non-obligate carnivores (for example, carps, shrimp) alike (Tacon & Metian, 2008b, 2015). The added demand from the rapid growth of aquaculture resulted in terrestrial husbandry substituting forage fish with alternative feed sources, reducing fishmeal and oil use by pigs and poultry to roughly 25% of total forage fish use **(Figure 3.33B)**.

To date, two factors have helped avoid resource limitations of forage fish affecting aquaculture growth. First, forage fish have become an increasingly smaller fraction of fish feed inputs over the decades, driven in part by price **(Figure 3.33C)**. Most aquaculture (and agricultural) feed is now largely crop-based (for example, soy), and this trend continues to increase **(Figure 3.33D)**. Additionally, some countries use trimmings (fish by-products) from aquaculture and fisheries, as well as other aquatic species, as forage fish alternatives **(Figure 3.33D)**. Second, aquaculture of selected species is continuously becoming more efficient, as measured by feed conversion ratios. Together, these factors contribute to lower fish-in-fish-out ratios (weight of forage fish used relative to fed cultured species produced).

The issue of fishmeal and oil use from aquaculture is continuing to raise diverging views and the sustainability of such practices remains dispersed in the literature (Natale, Hofherr, Fiore, & Virtanen, 2013). Cashion *et al.* (2017) underscore the concerns around directing ~20 million tons of wild fish every year towards feeding farmed fish, pigs and chickens instead of humans (Belton & Thilsted, 2014). Importantly, 90% of fish destined for uses other than direct human consumption are food-grade or prime food-grade fish (Cashion *et al.*, 2017). Tacon & Metian (2013) indicate that feed use of small pelagic fish competes with its use for food especially in developing countries. Much of the literature warns against aquaculture's reliance on forage fish, citing the fully exploited, over-exploited or recovering status of many forage fisheries (Rosamond L. Naylor *et al.*, 2000), though the current global amount being extracted appears below maximum sustainable levels (Froehlich *et al.*, 2018; Hilborn & Costello, 2018).

Switching from feed fish to direct human food would depend upon affordability and development of low-cost conserved products. A regional approach is needed to assess the consequences of using more feed fish for human consumption. While there are possible benefits of switching at least part of the catches of forage fish to food in South American countries, in Asia this is a less clearly understood, since cheap fish and trash fish contribute to the development of small-scale aquaculture, which reportedly has positive effects on livelihood and human consumption. In sub-Saharan Africa the effects would be limited since feed fisheries are an exception and aquaculture is not yet widespread or dependent on compound feed (Hasan & Halwart, 2009).

Nonetheless, the use of wild fish for feed by the aquaculture sector is increasing as a result of overall growth, intensification of farming practices, and the rising share of fed, higher trophic level species in total production (Tacon *et al.*, 2011; Tacon & Metian, 2008b, 2008a, 2009, 2015). Furthermore, fishmeal and oil are also used in terrestrial livestock feed, and their demand is increasing for pet food, and human food and medicine. Given current trends in aquaculture and demand for seafood and terrestrial meat, estimates suggest that ecological limits of forage fish could be reached as soon as 2037, or even sooner if precautionary measures do not further limit access to the wild resource (e.g., Atlantic herring, Clupea harengus, Clupeidae) (Froehlich *et al.*, 2018).

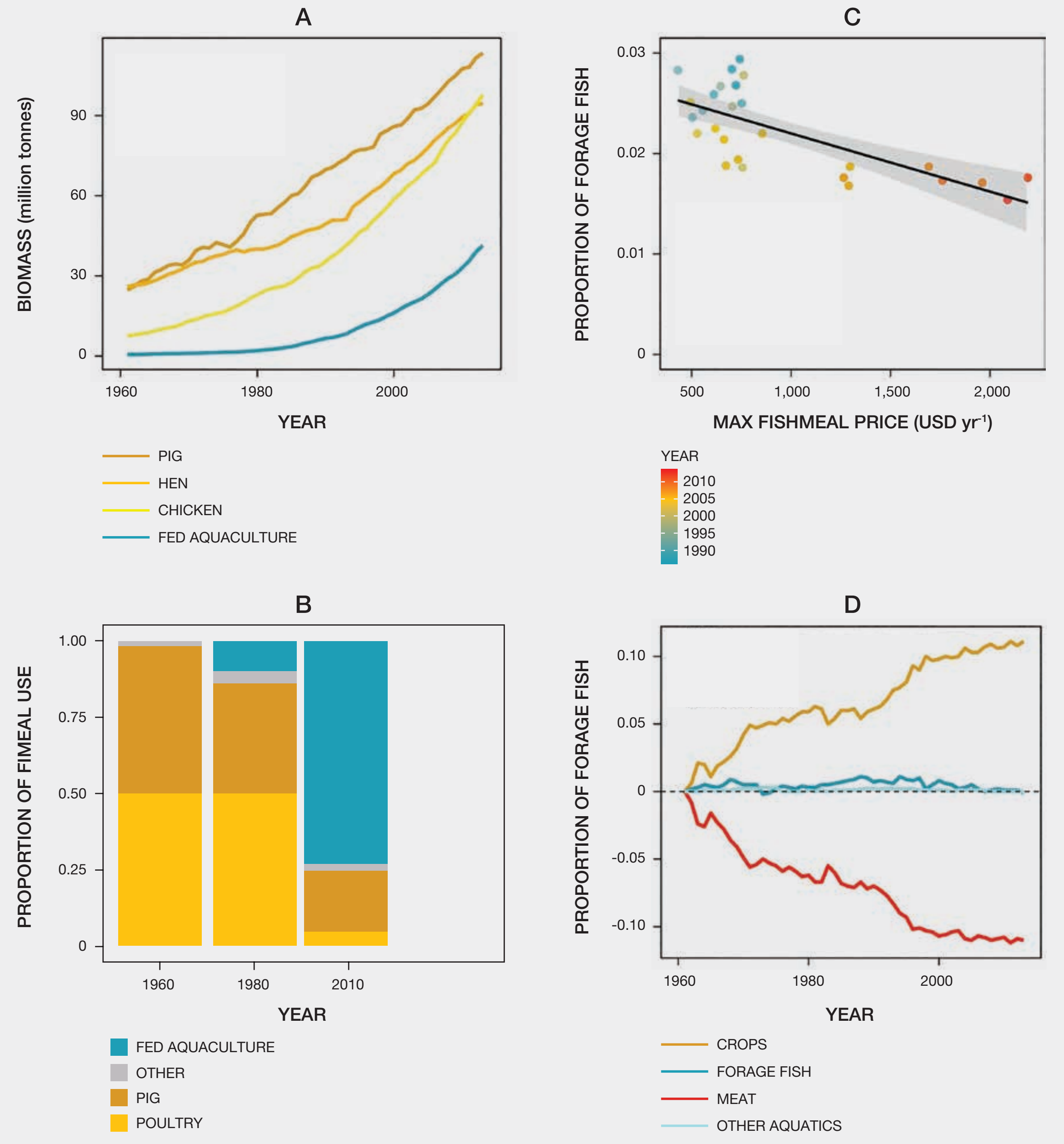


Figure 3 33 **Animal production (livestock, poultry and fed aquaculture species) and forage fish use trends.**

(A) Time-series of biomass of primary farmed animal groups (poultry and pig, fed aquaculture) that consume forage fish. **(B)** Proportion of fishmeal use by the primary consumer groups over time. **(C)** Proportion (mean ± 95% confidence interval) of forage fish in global feed relative to maximum fishmeal price per year (see color scale). **(D)** Difference in proportion of inclusion in total animal feed over time compared to 1961 for crops, forage fish (also depicted in panel C), other aquatic inputs and terrestrial meat-sourced contributions (by-products) in livestock and aquaculture feed, combined. Source: (Froehlich *et al.*, 2018) © 2018, Springer Nature under license CC BY-NC-SA 4.0.

Small Scale Fisheries contributing to Food and feed uses

This section was written following the methods used for the systematic review described in 3.3.1.3. (see the data management report for Chapter 3 systematic literature review at https://doi.org/10.5281/zenodo.6452651).

EUROPE AND CENTRAL ASIA

Small-scale fishing is still the most important component of commercial fishing in the European Union with special relevance in Southern Europe (Lloret *et al.*, 2018). It is a highly diversified fishery, involving fishing systems of many forms and sizes, and targeting a wide range of taxonomic groups. Small-scale fisheries in Europe are responsible for a catch equivalent to one large-scale fishery when it comes to human consumption (Leleu *et al.*, 2014). Traditional European small-scale fisheries are more than 2,000 years old (C. Antunes *et al.*, 2015), and data on some fisheries can be found as far back as 400 years (Marcos *et al.*, 2015). Stocks of small pelagic fish species and of larger demersal fish species have been exploited since the Middle Ages and are still important today (Almeida *et al.*, 2014; Bastari *et al.*, 2017; Battaglia *et al.*, 2017; Braga, Azeiteiro, Oliveira, & Pardal, 2017b). A number of local fish species exploited by European small-scale fisheries are famous worldwide, such as trout (Shephard *et al.*, 2019), cod (Dinesen *et al.*, 2019), anchovies and sardines (Sartor *et al.*, 2019).

As indicated by 47 out of the 63 papers reviewed, small-scale fisheries systems are often unsustainable, although varying levels of sustainability were observed in 30% of the papers (see the data management report for Chapter 3 systematic literature review at https://doi.org/10.5281/zenodo.6452651). This applies equally to inland, coastal and marine/oceanic fishing. A number of studies described multi-faceted fishing systems, where some stocks were sustainably exploited while others were not. In 74% of the papers, focusing on broader analysis, and using long and consistent series of data (a set comprising 19 papers), unsustainability elements were still observed.

Approximately 10% of the reviewed papers report cases of partial sustainability and 16% report sustainable cases of small-scale fishing (see the data management report for Chapter 3 systematic literature review at https://doi.org/10.5281/zenodo.6452651). The cases of sustainable fishing reported are mainly supported by enabling factors that reflect the adoption of sound management decisions and measures. These are systems that successfully controlled the fishing effort (Fourt *et al.*, 2020), and enforced the regulation of zones of use and no-use (no-take), determined by officially protected marine areas or estuaries (Antunes *et al.*, 2015; Guidetti & Claudet, 2010; Marengo *et al.*, 2015; Morales-Nin *et al.*, 2017).

The reported cases of unsustainability of small-scale fishing for food and feed are due to a larger and more diverse array of inter-related causes. Fishing pressure above the capacity of the stocks was mentioned by 41% of the papers. This leads to overfishing (either of the targeted species or of their prey species), catches above the maximum sustainable yield, or to the reduction in catch-per-unit-of-effort (Azzurro *et al.*, 2019; Corral & Manrique de Lara, 2017; Duncan *et al.*, 2016; Figus *et al.*, 2017; Lloret *et al.*, 2018; Quetglas *et al.*, 2016).

Overfishing may be a consequence of bad management practices, which was the second most frequently reported cause in the literature, mentioned in 22% of the reviewed papers. This includes ineffective control of fishing effort, incongruence between different management measures adopted simultaneously (Baeta *et al.*, 2018), adoption of dubious measures (Corral & Manrique de Lara, 2017), slow implementation of management measures (Colloca *et al.*, 2017), bad communication with the local fishers (Morales-Nin *et al.*, 2017), adoption of weak governance systems (Pita *et al.*, 2019) and competition between fishing modalities (Battaglia *et al.*, 2017; Carvalho *et al.*, 2017; Das & Afonso, 2017; Lloret *et al.*, 2018; Öndes, Kaiser, & Güçlüsoy, 2020). Environmental disturbances (14% of the papers) leading to the reduction of stocks, either by pollution, inadequate use of fishing gears or climate change were mentioned (Azzurro *et al.*, 2019; Braga *et al.*, 2018; Dinesen *et al.*, 2019; Pita *et al.*, 2019). Excessive discards or bycatch were also mentioned by a smaller number of papers (Öndes, Kaiser, & Güçlüsoy, 2020).

AFRICA

It is well established that most of the non-artisanal small-scale fishing in Africa is unsustainable (see the data management report for Chapter 3 systematic literature review at https://doi.org/10.5281/zenodo.6452651). This general statement applies equally to inland, coastal and marine/oceanic fishing.

Small species dominate the African fishing practice for both food and feed that involves the inland and coastal fisheries driven by tradition, species displacement, and substitution (Jamu *et al.*, 2011). A transition from large piscivorous species to small omnivorous species took place during the last half century, when larger fish were almost extinguished from the catch as a result of increasing numbers of fishers, fishing fleets and gear efficiency. A limited number of high value species continue to be targeted, often for export and for higher prices in international markets. There are many cases in the literature reporting on local problems with foreign industrial fleets competing with small-scale fleets, and both affect the artisanal fishing systems. Consequently, these systems show many indicators of unsustainability, such as declining stocks, catches, average size of fishes, catch-per-unit-of-effort, and so on (see the data management report for Chapter 3 systematic literature review at https://doi.org/10.5281/zenodo.6452651). Many official fish landing data series available are underestimated, and important attempts for reconstruction are taking place. Researchers have tried to uncover records of unregulated artisanal catches to produce realistic series of data. These reconstructed series also show the same declining trends.

In comparison with commercial fleet fisheries, artisanal fisheries present more diverse scenarios and different trends (see the data management report for Chapter 3 systematic literature review at https://doi.org/10.5281/zenodo.6452651). Some artisanal fishery systems show signs of reduction of catch volume and size over the decades due to excessive fishing pressures (Dyhia Belhabib *et al.*, 2018; Tuda & Wolff, 2015), mainly because fishing pressure has stayed high. Control of fishing pressure is, sometimes, an inherent trait of a system. Systems based on indigenous and local knowledge use the available habitats and fishing grounds (Mirera *et al.*, 2013) to distribute fishing pressure among a large number of species, or to focus the pressure on specific cohorts or in specific times (Musembi *et al.*, 2019). All of these are effective measures of controlling effort and off-take.

LATIN AMERICA

In Latin America, almost all studies (99) analyze the use of fisheries resources as food, either for subsistence, commerce or both. Overall, 15% of these studies report sustainable use, 48% report unsustainable use (exploited populations declining and other sustainability problems), and 37% indicated partially sustainable use (see the data management report for Chapter 3 systematic literature review at https://doi.org/10.5281/zenodo.6452651). Considering 76 studies of coastal small-scale fisheries (including oceanic islands, bays and estuaries), about 13% (10 studies) mention sustainable use, while 53% indicate unsustainable fishing (see the data management report for Chapter 3 systematic literature review at https://doi.org/10.5281/zenodo.6452651). Among 23 studies on inland or freshwater small-scale fisheries, 22% indicate sustainable and 30% unsustainable uses, whereas the remaining majority of studies point to partially sustainable use, suggesting less data availability for inland fisheries compared to coastal cases (see the data management report for Chapter 3 systematic literature review at https://doi.org/10.5281/zenodo.6452651). Sustainable coastal small-scale fisheries examples include co-management systems through territorial rights granted to fishing communities and well-established rules to exploit mainly shellfish, oysters and lobsters in Chile and Mexico (Álvarez *et al.*, 2018; Castilla, Espinosa, Yamashiro, Melo, & Gelcich, 2016; De la Cruz-González *et al.*, 2018; Defeo *et al.*, 2016; Gelcich *et al.*, 2017, 2010). Some of these co-management systems were effective in supporting recovery of resources after a fishery collapse, such as the shellfish *Concholepas concholepas* in Chile (Castilla *et al.*, 2016; Defeo *et al.*, 2016; Gelcich *et al.*, 2010), *Atrina maura, A. tuberculosa, Pinna rugosa*, oyster *(Crassostrea iridescens)*, lobster and fish in Mexico (Álvarez *et al.*, 2018; De la Cruz-González *et al.*, 2018; (Palacios-Abrantes, Herrera-Correal, Rodríguez, Brunkow, & Molina, 2018).

In association with small-scale fisheries, a management strategy called "Territorial Users' Rights Fisheries management" (TURF) has been implemented with varying success in Chile (Defeo *et al.*, 2016; Gelcich *et al.*, 2010). For example, population and catches of the clam *(Mesodesma donacium)* declined over time after the establishment of a territorial users' rights fisheries system, causing the collapse of the clam fishery. However, this was at least in part due to management restrictions preventing fishers from moving fishing grounds to cope with natural variability of clam abundance (Aburto & Stotz, 2013). A study across 500 km of the Chilean coast indicates effects of displacement caused by Territorial Users' Rights Fisheries, which intensify fishing efforts and thereby reduce shellfish abundance in open access areas which have been reduced in size compared to surrounding areas which have entered into government management. This creates conflict and resource shortages for fishers not engaged Territorial Users' Rights Fisheries management (Garmendia *et al.*, 2021).

Other cases of sustainable coastal small-scale fisheries include two fish species *(Paralabrax nebulifer, Caulolatilus princeps)* that have been fished sustainably mostly because they are only occasionally fished, when preferable resources are unavailable in Baja California Sur, Mexico (Cavieses Núñez *et al.*, 2018). The sustainability of the important fishery of octopus *(Octopus maya)* in Yucatan (Mexico) is unresolved (Duarte, Hernández-Flores, Salas, & Seijo, 2018b; Raya & Berdugo, 2019). From one hand, the regulations, including fishing season, applied over 30 years and the interaction of fishing gear (baits) with the reproductive behavior of parental care without feeding performed by females, of may have contributed to maintain stocks of this octopus, even in face of intense fishing pressure (Duarte *et al.*, 2018a). However, an increased market demand and search for profit maximization may have pushed fishers to adopt a combination of legal and illegal, furtive, and undeclared fishing tactics (including diving), which may undermine sustainability and threatens the long-term viability of the octopus' fishery in the Yucatan Peninsula (Raya & Berdugo, 2019). Most of the studies identifying partially sustainable coastal small-scale fisheries include those lacking temporal series of data to estimate species declines, showing distinct trends among exploited species (i.e., some declining, others stable or increasing) or other indicators (e.g., catch volume and size), besides some studies indicating shifts in composition of fished species (see the data management report for Chapter 3 systematic literature review at https://doi.org/10.5281/zenodo.6452651). Several studies on coastal finfish fisheries fall in this partially sustainable category in Brazil (Barbosa-Filho *et al.*, 2020; Damasio *et al.*, 2015; Lima *et al.*, 2016; Silvano, Nora, Andreoli, Lopes, & Begossi, 2017), Mexico (Erisman *et al.*, 2010; Rife *et al.*, 2013) and Colombia (López-Angarita *et al.*, 2018), besides the fishery for king crab *(Lithodes santolla)* in Chile (Bozzeda, Marín, & Nahuelhual, 2019). Some of these partially sustainable cases involve a temporal shift in the exploited fishing resources, in the form of a decline (or even disappearance) in catches of large, slow growing and high valued fish, such as reef predators, coupled with an increase in catches of smaller, fast growing and usually less valued fishery resources, such as shrimp, reef herbivores, or pelagic fish, as indicated in Brazil (Damasio *et al.*, 2015; Ribeiro, Damasio, & Silvano, 2021; Zapelini, Bender, Giglio, & Schiavetti, 2019), Ecuador (Schiller, Alava, Grove, Reck, & Pauly, 2015), Costa Rica (Sánchez-Jiménez, Fujitani, MacMillan, Schlüter, & Wolff, 2019) and Mexico (Erisman *et al.*, 2010; Rubio-Cisneros, Aburto-Oropeza, & Ezcurra, 2016; Rubio-Cisneros *et al.*, 2017). This pattern was also observed in fisheries of elasmobranchs (sharks and rays) in Mexico, where catch of large and threatened species has declined, whereas smaller and more resilient species have increased and tend to have sustained an intense fishing pressure (Ramírez-Amaro & Galván-Magaña, 2019; Saldaña-Ruiz, Sosa-Nishizaki, & Cartamil, 2017).

Although benthic invertebrates have been usually among the more sustainable fisheries (see the data management report for Chapter 3 systematic literature review at https://doi.org/10.5281/zenodo.6452651), there are some cases of overexploitation and even fisheries collapses of high valued and easy to catch invertebrates, such as the abalones (*Haliotis* spp.) or sea cucumbers (*Isostichopus badionotus*, among other species), in Chile (Sáenz-Arroyo & Revollo-Fernández, 2016), Ecuador (Schiller, Alava, Grove, Reck, & Pauly, 2015) and Mexico (Gamboa-Álvarez *et al.*, 2020). The size and density of the shellfish Queen conch *(Lobatus gigas)* had declined over a 15-year period in Belize, raising concerns of recruitment or overfishing, but deep water and protected areas may provide a refuge from fishing pressure (Tewfik, Babcock, Appeldoorn, & Gibson, 2019).

Only a few studies were considered to be partially sustainable or unsustainable because of side effects from some fishing practices that would cause habitat damage or by-catch, for example, trawling to catch shrimp (Martins *et al.*, 2018; Rosa *et al.*, 2011; Sánchez-Jiménez *et al.*, 2019), pufferfish (Eduardo *et al.*, 2020) or jellyfish (Brotz *et al.*, 2017). This may be partially due to two factors that may have reduced the number of articles on trawling in this review: first, some or most of these trawling fisheries may be considered to be large-scale to have made it into the small-scale fisheries review. Second, some studies were perhaps not retrieved in a review using search words as 'sustainable, sustainability, success and increasing'.

The sustainable cases of inland small-scale fisheries (see the data management report for Chapter 3 systematic literature review at https://doi.org/10.5281/zenodo.6452651) are mostly related to the successful co-management of the large and valuable commercial fish pirarucu *(Arapaima gigas)* in the Brazilian Amazon (Campos-Silva & Peres, 2016; Castello *et al.*, 2009; Petersen, Brum, Rossoni, Silveira, & Castello, 2016), (see also **Box 6.5** on community-based fishery of pirarucu in the Amazon in Chapter 6). Other mechanisms that could lead to sustainable fisheries are the exploitation of fish resilient to either fishing pressure or environmental change (e.g., dams). For example, the fishing of *Plagioscion squamosissimus*, especially in communities within extractive reserves or other kinds of protected areas, as observed in several rivers in the Brazilian Amazon (Gustavo Hallwass, Luís Henrique Tomazoni da Silva, Paula Nagl, Mariana Clauzet, & Alpina Begossi, 2020; Gustavo Hallwass *et al.*, 2011;Hallwass *et al.*, 2020; Gustavo Hallwass & Silvano, 2016; Keppeler *et al.*, 2017; Mesquita *et al.*, 2019; Silvano *et al.*, 2014).

Small-scale fisheries have cultural and socioeconomic relevance to indigenous Tacana people in the Beni River, Bolivian Amazon, where a participatory survey indicated that this fishery, which exploits 43 species for food and income, has been ecologically and economically sustainable as catches and sizes of exploited fish remained unchanged for a period of seven years, providing a regular source of revenues to local communities (Salinas *et al.*, 2017). Similarly, a study conducted 20 years ago indicates that the large frugivorous fish *Colossoma macropomum* in the Bolivian Amazon supports a sustainable fishery partly due to its linkages with a small population and a well-preserved floodplain forest habitat (Reinert & Winter, 2002). Unsustainable cases include migratory fish, such as *Prochilodus* species among others, which have suffered intense fishing pressure, sometimes aggravated by environmental impacts (e.g., dams in rivers in Brazil) (Catarino *et al.*, 2019; Santos, Pinto-Coelho, Fonseca, Simões, & Zanchi, 2018; Philippsen *et al.*, 2017) and Argentina (Baigún *et al.*, 2013). Another unsustainable pattern in the Brazilian Amazon refers to the decrease in catches and size of some of the larger and most valuable commercial fishes, such as *Colossoma macropomum* and large catfish (Pimelodidae), among others (Castello, McGrath, & Beck, 2011; Garcez Costa Sousa & de Carvalho Freitas, 2011; Hallwass *et al.*, 2019; Tregidgo, Barlow, Pompeu, de Almeida Rocha, & Parry, 2017). Even the pirarucu *(Arapaima gigas)* that increased in co-managed small-scale fisheries is considered to be unsustainably exploited in non-managed Amazonian rivers, where the abundance of this fish has reportedly reduced (Leandro Castello *et al.*, 2015; G. Hallwass *et al.*, 2019), mainly due to widespread illegal fishing (Cavole, Arantes, & Castello, 2015).

An interesting exception of this pattern is the fishing of some large and migratory catfish (*Brachyplatystoma* spp.) in the Brazilian Amazon, as catches of some of these species have increased in some rivers either due to successful regulations, improved fishing technologies (larger nets, motorized boats) or to market opportunities (Cruz *et al.*, 2020; Gustavo Hallwass *et al.*, 2020; G. Hallwass *et al.*, 2019). However, these catfish fisheries are difficult to manage due to the long migrations (more than 1,000 km) that these fish perform along the main Amazon River and its tributaries (Barthem *et al.*, 2017; Nunes *et al.*, 2019; Petrere, Barthem, Córdoba, & Gómez, 2004), which make these fishes especially susceptible to impacts caused by dams (Santos *et al.*, 2018) and may thus require basin wide or even transboundary international management approaches(Doria *et al.*, 2020; Goulding *et al.*, 2019).

NORTH AMERICA

In North America, most of the reviewed studies focus fish and invertebrates as food (15) were from the Artic and Alaska (see the data management report for Chapter 3 systematic literature review at https://doi.org/10.5281/zenodo.6452651), whereas fewer studies address recreational (3), or both uses (4). Among the studies on use of fishing resources as food, only one reported a sustainable lobster fishery in California (United States of America), where a collaborative approach improved the ecological assessment and feedback with human dimensions of the system (Kay *et al.*, 2012). Similarly, a study including fishers' knowledge indicated that the depletion of Atlantic cod caused an increase in the population of lobsters, thus improving the sustainability of lobster fisheries (Boudreau & Worm, 2010). Another study indicated that Pacific salmon species have been increasing in recent years (1990s and early 2000s) in the Beaufort Sea (Carothers, Sformo, Cotton, George, & Westley, 2019). Some problems affecting the sustainability of coastal small-scale fisheries are the potential serial depletion and regional overfishing of the rock crab fishery in California (Fitzgerald, Wilson, & Lenihan, 2018), severe declines in the abundance and catches of the Dungeness crab *(Cancer magister)* in Canada (Ban *et al.*, 2017), and overfishing and declines of stocks of salmon in Alaska (H. L. Harrison & Loring, 2016; Loring, Harrison, & Gerlach, 2014) and sea cucumber in Canada (O'Regan, 2015). One study that combines traditional and scientific ecological knowledge showed that two exploited shellfish species were also impacted by local and regional environmental factors (Ambrose *et al.*, 2014). However, other studies have shown that the involvement of indigenous peoples and local communities were critical to the reversion of a declining trend in local populations of lake sturgeons *(Acipenser fulvescens)* in Wisconsin and the Great Lakes region (United States of America), a very relevant social and economic traditional small-scale fishery (Kline, Bruch, & Binkowski, 2012; Runstrom, Bruch, Reiter, & Cox, 2002). The previous population decline of this species was due to unsustainable practices, such as overfishing and habitat loss or transformation, trends also observed in other parts of the world, including the large-scale fishing of sturgeons (Tavakoli *et al.*, 2021). In the Great Lakes region, the local community-built co-management rules across the last six decades for the sturgeon fishery, including fishing festivals and competitions (Kline *et al.*, 2012), and this fishery is also important for the Menominee Nation (an indigenous tribe in upper Midwestern in the United States of America). Together with local authorities and researchers, the local community build a successful restoration program to reintroduce lake sturgeon larvae to areas where they could no longer be found (Runstrom *et al.*, 2002).

ASIA-PACIFIC

In Asia-Pacific, the majority of studies (77) address the use of fishing resources as food, either as subsistence, commercial or to support livelihoods. Only 6 studies report sustainable use of fishing resources for food in coastal small-scale fisheries, including reef fish and invertebrates in the American Samoa (Craig *et al.*, 2008), Solomon Islands (Cohen *et al.*, 2013), the Torres Strait Islands in Australia (Busilacchi, Russ, Williams, Begg, & Sutton, 2013), besides fisheries of shrimp in Indonesia (Anna, 2017), abalone (*Haliotis* spp.) in Australia (Mayfield *et al.*, 2012) and co-managed finfish fisheries in Bangladesh (Mazumder *et al.*, 2016). The analysis of a long time series of 3,000 years involving both indigenous and local knowledge from fishers and archaeological data indicates no major changes in catch composition of fish and invertebrates exploited in the American Samoa, where catches are at lower levels than the estimated stock sizes of reef fish and fishing yields (kg/ha) correspond to those of less fished Pacific Islands (Craig *et al.*, 2008). This sustainable pattern may be related to a relatively small population of fishers who fish primarily for subsistence and, even considering that sales increased over time, other economic opportunities may have reduced reliance on fishing and hence fishing pressure (Craig *et al.*, 2008).

The observed fisheries' sustainability in the Torres Strait Islands could also be partially related to more subsistence-oriented fisheries (Busilacchi *et al.*, 2013). Similarly, in French Polynesia catches of reef fish have been stable for nine years, even after major natural disturbances including a cyclone. This could be partially due to government subsidy that reduced poverty among fishers, who are mostly part time (Rassweiler *et al.*, 2020). Nevertheless, these are exceptions among the Pacific Island countries, where most cases of sustainable or potentially sustainable fisheries are usually linked to some form of co-management or customary management system, such as periodic harvest closures (Cohen & Alexander, 2013;Cohen *et al.*, 2013; Cohen & Foale, 2013). Although promising, these co-management systems have shown variable results depending on the life history of exploited species, the size of managed area and the regime of opening and closing the area to fishing, which regulates the fishing pressure (Cohen & Foale, 2013b). Therefore, some of these co-management systems improved fisheries yields for fast-growing exploited species in a context of moderate or low fishing intensity. Others have shown a decline of larger and slow growing species (reef fish), usually associated with smaller closed areas or more intense fishing promoted by shorter closed intervals (less than one year) and longer opening periods (Cohen & Foale, 2013; Goetze, Langlois, Claudet, Januchowski-Hartley, & Jupiter, 2016; Hamilton, Hughes, Brown, Leve, & Kama, 2019; Rhodes *et al.*, 2008; Yang & Pomeroy, 2017).

Even the more sustainable reef fisheries observed in American Samoa and French Polynesia show a lack of larger piscivorous reef fish, suggesting these larger predators may have been intensively fished in the past (Craig *et al.*, 2008; Rassweiler *et al.*, 2020). In some Pacific Island countries, fisheries for small pelagic fish could be a sustainable alternative for food production, as these fish seem to be more resilient to fishing pressure compared to larger reef fish, as observed in the Solomon Islands (Roeger, Foale, & Sheaves, 2016) and Timor Leste, where fishing aggregation devices and co-management has helped to improve sustainability of coastal and reef fisheries (Tilley, Hunnam, *et al.*, 2019; Tilley, Wilkinson, *et al.*, 2019).

There are examples of co-management measures that helped to recover the abundance and hence to improve sustainability of fisheries resources, such as shellfish in Fiji (Thaman, Thaman, Balawa, & Veitayaki, 2017) and reef fish in Hawaii (Friedlander, Shackeroff, & Kittinger, 2013; Friedlander *et al.*, 2014). However, some studies also indicated declines in catches of Hawaiian fisheries for octopus and reef fish (Delaney *et al.*, 2017; Kittinger *et al.*, 2015). An analysis of bioeconomic modelling, which included stock parameters in addition to data on catch, effort, revenues and costs, indicated that shrimp fisheries could be sustainable in Indonesia by showing increased catches over a period of 27 years and surplus stocks (Anna, 2017). However, potential side effects or impacts from shrimp fisheries, such as by-catch or habitat damage, were not included in this study (Anna, 2017), which could compromise the overall sustainability of this fishery. A moratorium of one year imposed on large scale fisheries for tuna in Indonesia had mixed effects on catches of the small-scale pole and line tuna fisheries, but fishers considered that the moratorium was positive and increased their catches, indicating the potential conflicts and competition between large- and small-scale fisheries (Khan, Gray, Mill, & Polunin, 2018).

Some cases of unsustainable coastal small-scale fisheries include sharks in Indonesia (Ainsworth, Pitcher, & Rotinsulu, 2008; Ferse *et al.*, 2014; Jaiteh *et al.*, 2017) and China (Lam & Sadovy De Mitcheson, 2011a), and sawfish in Bangladesh (Hossain *et al.*, 2015). In China, a comprehensive study including market surveys and interviews with fishers in Hong Kong and mainland southern China indicates an overall depletion of sharks in the South China sea, where shark fisheries collapsed between 1970s and 1990s, (Lam & Sadovy De Mitcheson, 2011b). Notwithstanding management measures implemented by the Chinese government, a recent study analyzing landings' data and fishing effort from China for the period of 1955–2019, shows variable decadal trends with an overall adverse effect from increased fishing intensity on piscivorous fishes, including sharks and rays (Liu *et al.*, 2021). Effects from overfishing can interact synergistically with effects from climate change (Liu *et al.*, 2021). Moreover, China, especially Hong Kong, is the world largest market for shark fins, thus driving exploitation and trade of sharks worldwide, usually at unsustainable levels (Eriksson & Clarke, 2015; Fields *et al.*, 2018). However, shark fisheries can be at least partially sustainable in the Great Barrier Reef (Australia), where susceptibility to fisheries vary among species and their life histories, as smaller species are caught mainly as adults (less vulnerable), whereas larger ones are regularly caught as juveniles, and thus more vulnerable to fishing (Harry *et al.*, 2011). Furthermore, in Eastern Indonesia the whale shark *(Rhincodon typus)* is not commercially exploited due to customary beliefs among Bajao people that prohibit harvesting this fish, low market values of shark meat and skin, and a lack of technology to harvest such a large fish. No catch data is provided for other regions of Indonesia where this shark could be commercially fished (Stacey, Karam, Meekan, Pickering, & Ninef, 2012).

Unsustainable patterns have been also observed in several countries for fisheries of sea cucumbers (Holoturidae), which usually shows a typical cycle of boom and bust typically ending in sharp declines (Eriksson *et al.*, 2018; Hair *et al.*, 2016; Prescott *et al.*, 2017). The large reef fish that form predictable spawning or feeding aggregations, such as groupers or large herbivores, may be negatively affected by unsustainable practices, such as night spearfishing and catches of juveniles, as these slow growing fish are vulnerable to intense fishing during aggregation periods, even in regions under co-management systems (Hamilton *et al.*, 2019; Hamilton *et al.*, 2012; Rhodes *et al.*, 2008; Robinson, Cinner, & Graham, 2014).

Unsustainable nearshore coastal and reef fisheries have been observed in Southeast Asian countries located in biodiversity hotspots, such as Indonesia and Philippines, among others (see the data management report for Chapter 3 systematic literature review at https://doi.org/10.5281/zenodo.6452651). Multiple factors, including increased population, poverty, lack of economic alternatives other than fishing, pressure from domestic and international markets, open access and illegal fishing by using destructive practices (bombs, cyanide) lead to unsustainable levels of fishing effort and overall declines in catches of many fishing resources, such as reef and coastal fish, sharks, rays, sea cucumbers and lobsters in these biodiversity rich countries (Acebes, Barr, Pereda, & Santos, 2016; Ainsworth *et al.*, 2008; Ferse *et al.*, 2014; Jaiteh *et al.*, 2017; Khasanah, Nurdin, Sadovy de Mitcheson, & Jompa, 2020; Macusi *et al.*, 2019; Muallil, Mamauag, Cababaro, *et al.*, 2014; Muallil, Mamauag, Cabral, Celeste-Dizon, & Aliño, 2014; Prescott *et al.*, 2017; Selgrath *et al.*, 2018a, 2018b).

None of the 10 studies addressing use of fish or invertebrates for food in coastal and inland small-scale fisheries in Asia Pacific indicate sustainable fisheries (see the data management report for Chapter 3 systematic literature review at https://doi.org/10.5281/zenodo.6452651). These fisheries are usually considered unsustainable due to multiple and interacting effects of overfishing, lack of proper management, illegal or destructive fishing practices, coupled with habitat alteration by river dams, deforestation, pollution and increased water temperature, as observed in Bangladesh (Ahmed, Rahman, Bunting, & Brugere, 2013; Jahan, Ahsan, & Farque, 2017), Laos (Gray *et al.*, 2017; Millar *et al.*, 2019) and India (Dey *et al.*, 2019; Keskar, Raghavan, Kumkar, Padhye, & Dahanukar, 2017). Nevertheless, an increase in low value fish has been observed in Cambodia (Enomoto *et al.*, 2011) and co-management initiatives including fishers' indigenous and local knowledge could be strategic and promising for recovery of fish stocks in the Mekong River Basin (Baird & Flaherty, 2005) and through community-based freshwater reserves in Thailand (Koning, Perales, Fluet-Chouinard, & McIntyre, 2020). The widespread small-scale coastal fisheries in Japan have a long history of a strong bottom-up, co-management system of governance, actively including local fishers through the fishery cooperative associations, which cooperate with scientists and government to regulate fishing activity and allocate fishing grounds among coastal fishers, among other management activities (Ganseforth, 2021; Makino, Matsuda, & Sakurai, 2009; Matsuda, Makino, & Sakurai, 2009; Teh, Teh, Abe, Ishimura, & Roman, 2020). This co-management system can contribute to the sustainability of small-scale fisheries and to marine conservation in Japan, to the extent that local communities can implement fishery regulations to cope with declining fishing resources. This may include protected areas, gear modifications and restrictions on fishing effort (number of boats), as observed in the Shiretoko World Natural Heritage Site where the management plan considers fishers as part of the ecosystem (Makino *et al.*, 2009; Matsuda *et al.*, 2009). Nevertheless, the social and economic sustainability of the Japanese small-scale fisheries face some challenges, such as limited workforce due to an ageing population and lower incomes from fishing compared to other activities (Teh *et al.*, 2020), besides institutional changes that may reduce participation of local fishing association in fisheries management (Ganseforth, 2021).

3.3.1.5.2 Medicine and hygiene

Aquatic organisms provide diverse sources of bioactive compounds of interest for nutraceutical, pharmaceutical, and cosmeceutical industries **(Table 3.4)**. Fish, crustaceans and molluscs produce a variety of biologically active compounds that have been characterized by their antimicrobial, antiviral, anti-inflammatory, antioxidant, anti-cancer/antitumor, antihypertensive, anti-atherosclerotic, anticoagulant, and immunomodulatory properties and other medicinal functions (Chbel, Asmaa, Delgado, Aurelio Serrano, Soukri, Abdelaziz, & El Khalfi, Bouchra, 2021; Nisticò, 2017; Šimat *et al.*, 2020). Fish oil, chitin, peptides, polysaccharides, gelatin, pigments, polyphenols, vitamins and minerals are examples of the compounds that have been used as functional food ingredients (Venugopal, 2018) with health benefits. For a number of countries, especially in the tropics, nutrients such as zinc, calcium and iron available from marine fish are essential to the health of local populations, especially for children under five years old (Hicks *et al.*, 2019). Biological properties of fish have also been used to treat or prevent different kinds of health disorders.

The food industry introduced several components to improve the properties of foods (i.e., emulsifier, stabilizer, texture modifier, coating or thickening agent) or to enrich foods with functional components and allow their application in health-promoting foods for direct consumption (Šimat *et al.*, 2020). There are many papers promoting the benefits of biologically active components from wild caught animals, but little data were found on the number of wild animals caught and used in pharmaceuticals, nutraceuticals and hygiene products. Thus, the below review focuses on selected uses for which there is enough information to provide an overall assessment.

Fish oil as a source of omega-3 long chain polyunsaturated fatty acids

Fish oils contain high levels of omega-3 long chain polyunsaturated fatty acids (n-3 LC-PUFA), including those known as EPA and DHA (eicosapentaenoic acid [20:5n-3] and docosahexaenoic acid [22:6n-3]). Those components are well accepted as being essential for a healthy and balanced diet, and a large number of studies demonstrate the positive effects of food supplementation with fish oil on human health and the prevention of certain diseases (see (Ghasemi Fard, Wang, Sinclair, Elliott, & Turchini, 2019) for a review).

Table 3 4 **Major nutraceuticals and bioactive components from seafood.**

Source: (Venugopal, 2018) © 2018, Springer International Publishing A, license number 5153531358540. CC-BY-NC.

Finfish
Bioactive peptides
Biological calcium Cartenoids Ensymes includinf cold-adapted enzymes
Glycosaminoglycans inclusing chondroitin sulfate, dermatan sulfate and hyaluronic acid
Long-chain omega-3 polyunsaturated fatty acids (PUFAs)
Phosphopeptide from fish bone
Protein hormones such as calcitonin
Protein isolates including collagen and gelatin
Suqlene and squalamine
Shellfish (crustaceans and mollusks)
Bioactive peptides
Carotenoids
Chitn, chitosan and chitosan derivatives
Enzymes
Glucosamine
Long-chain omega-3 polyunsaturated fatty acids (PUFAs)
Mussel polysaccharides, lipids and other prodcuts
Protein isolates including collagen and gelatin

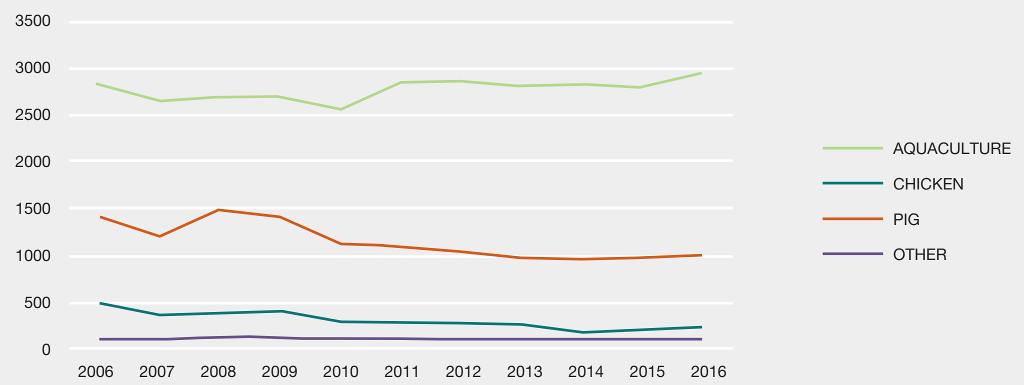


Figure 3 34 **World fish oil market use by sector 2006–2016 (000Mt).**

Source: (Seafish, 2018) under license CC-BY.

The vast majority of n-3 LC-PUFA is produced by marine micro-organisms, predominantly microalgae (Harwood & Guschina, 2009), whereas terrestrial wild plants do not produce EPA or DHA (eicosapentaenoic acid [20:5n-3] and docosahexaenoic acid [22:6n-3]). (Harwood, 1996). Therefore, the supply of these components for humans come from the ocean, and predominantly from capture fisheries (almost 90%), whether as food fish or via fish oil and fishmeal, with relatively small additional amounts estimated from seafood by-products and recycling, unfed aquaculture and traditional macroalgal sources (Tocher *et al.*, 2006).

The global supply of fish oil remains relatively stable (FAO, 2020d; J. Shepherd & Bachis, 2014), constrained largely by natural supply constraints in the fisheries (Misund, Oglend, & Pincinato, 2017) **(Figure 3.34)**. Supplements in the food industry use 20 to 25 percent of globally available fish oil

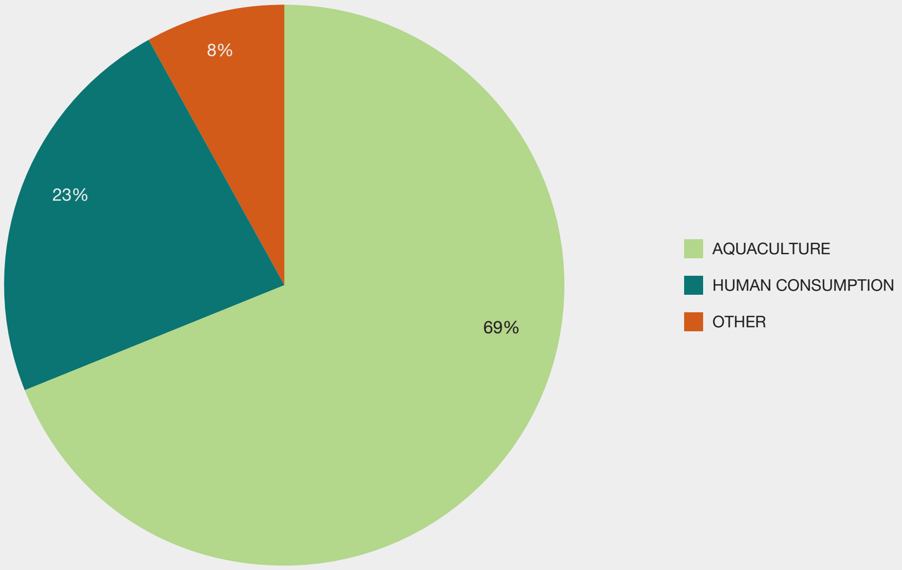


Figure 3 35 **Global fish oil use per destination in 2017 (volume in tonnes).**

Source: (EUMOFA, 2019) under license CC-BY.

(2017), up from only 5% in 1990 **(Figure 3.35)**. While Fish oil is currently the only economically viable source of n-3 LC-PUFA for feed purposes (Misund *et al.*, 2017), the growing demand from the human nutritional supplement industry has tightened the competition noticeably (J. Shepherd & Bachis, 2014). Based on the recommended dose for cardiac health, the total demand for n–3 LC-PUFA is over 1.25 million metric tonnes (mt) whereas total supply is optimistically estimated at just over 0.8 million mt indicating a shortfall of over 0.4 million mt (Tocher, 2015).

Squalene, squalane, and related compounds from shark's liver

Livers of deep-sea shark species contain high contents of squalene and other hydrocarbons like pristine, which are of interest for cosmetics and medical uses (Macdonald & Soll, 2020). Many shark species, particularly from the deep-sea >200 m, have relatively large livers (up to 20% of animal weight) (Abel & Grubbs, 2020; Vannuccini, 1999). The proportion of liver oil varies between species from 10 to 70% of liver weight (Nichols, Rayner, & Stevens, 2001), and 15 to 82% of liver oil is squalene (Bakes & Nichols, 1995; Deprez, Volkman, & Davenport, 1990). The preferred commercial source of squalene remains shark liver oil, although produced by different animals and plants, presumably due to availability and high yields relative to most plant-derived sources.

Squalene is a skin rejuvenating agent and together with its hydrogenated product squalane (produced from squalene), there is huge potential in nutraceutical, pharmaceutical, and cosmeceutical industries (Venugopal, 2018). Squalene is also used as an adjuvant in vaccines (Brito & O'Hagan, 2014) especially in influenza vaccines (Panatto *et al.*, 2020; Schultze *et al.*, 2008). Shark liver oil also contains Pristane, a natural saturated terpenoid alkane, and squalamine, an amino sterol antibiotic with antiviral, antitubercular, anti-angiogenic properties (Venugopal, 2018).

Recent data shows an increase in reported import and processed production of shark liver oil, with trade volumes reaching 752 tons as the largest reported volume in decades **(Figure 38)** (FAO, 2020d). A review of scientific and management literature by Macdonald and Soll (C. Macdonald & Soll, 2020) identified 133 shark species which are known to be involved in the liver oil trade. One-third of identified species are classified as threatened (vulnerable, endangered, or critically endangered) according to the International Union for Conservation of Nature Red List criteria **(Figure 3.36)**. Population trends for 56% of these species are unknown, and 34% are assessed as showing a decreasing trend **(Figure 3.37)**.

Deep-sea sharks offer larger volumes of liver oil compared to other shark species and are therefore of greater interest to the shark liver oil trade **(Figure 3.38)**. The knowledge on these species remains relatively poor due to low research priority added to the difficulties to conduct research in the deep sea (Kyne & Simpfendorfer, 2007; Neiva, Coelho, & Erzini, 2006; Verissimo, MacMillan, & Smith, 2011). Therefore, little is known about population structure, habitat use and reproduction of many of these species. Nevertheless, shark reproductive rates and recovery potential are known to decline when depth increases, and

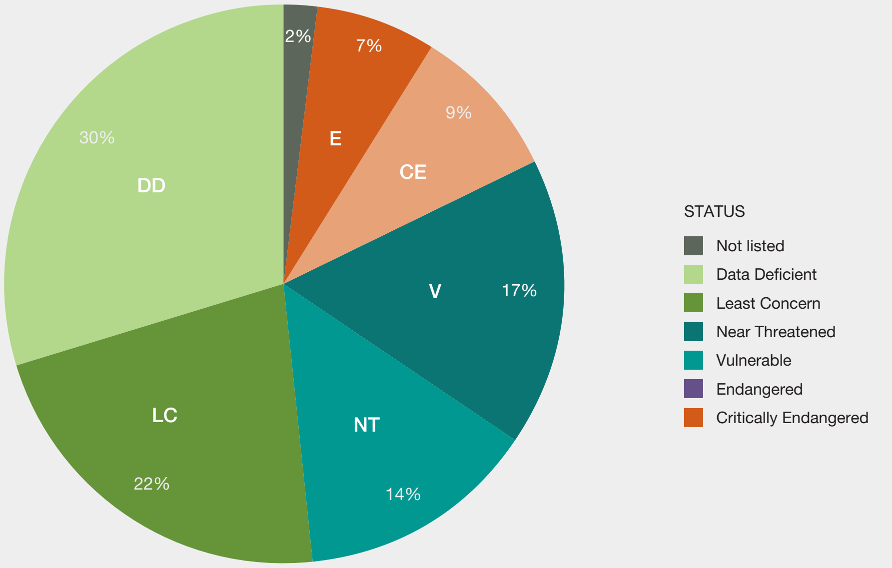


Figure 3 36 **The International Union for Conservation of Nature Red List conservation status of elasmobranch species reported in the liver oil trade.**

Source: (Macdonald & Soll, 2020) under license CC BY-NC-ND 4.0.

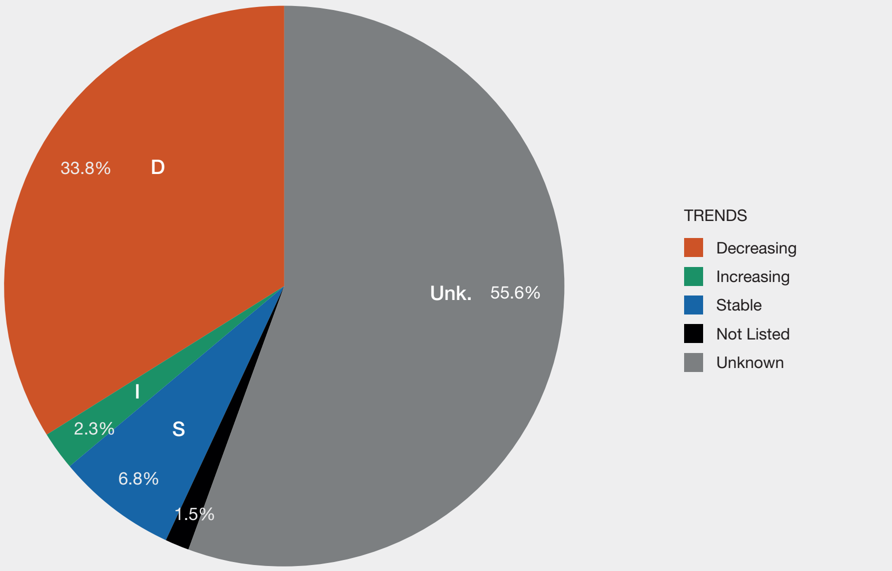


Figure 3 37 **The International Union for Conservation of Nature Red List conservation status of elasmobranch species reported in the liver oil trade.**

Source: (Macdonald & Soll, 2020) under license CC BY-NC-ND 4.0.

population depletion risks exist even when exploitation (targeted or incidental) rates are low (Simpfendorfer & Kyne, 2009). For these reasons, deep-sea sharks have been identified as a conservation priority (Dulvy *et al.*, 2014).

The cosmetics industry in Europe and the United States of America has decreased its use of shark-based squalene in recent years, under pressure from non-profit organizations and consumers. Independent tests conducted by the French organization "Bloom" determined that most cosmetics (>90% of products tested) sold in Europe or the United States of America no longer contain shark-derived ingredients, although shark-derived squalene is still commonly used in cosmetics

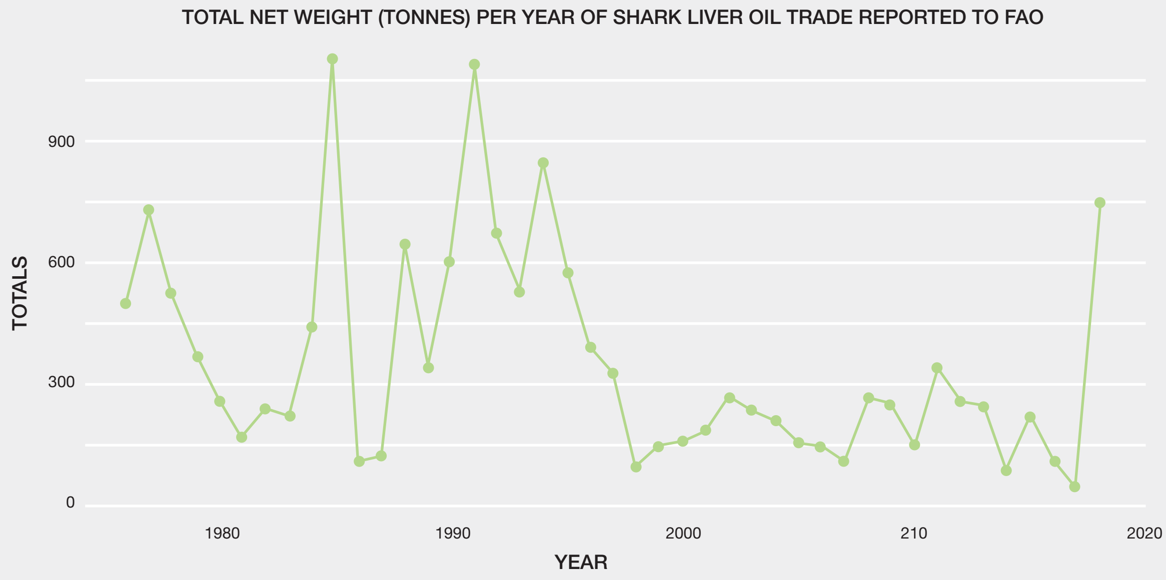


Figure 3 38 **The total reported net weight (tons) of annual trade in shark liver oil reported to the Food and Agriculture Organization of the United Nations.**

Source: (Macdonald & Soll, 2020) under license CC BY-NC-ND 4.0.

elsewhere (Ducos, Guillonneau, Le Manach, & Nouvian, 2015). The Covid-19 pandemic has reinvigorated the debate on using shark squalene-derived products in the production of potential SARS-CoV-2 vaccines (C. Macdonald & Soll, 2020).

Bioactive compounds from wild caught species and seafood processing by-products

Fish and shellfish, including crustaceans, are sources of a wide range of bioactive compounds **(Box 3.7)** that can be recovered from commercial fish processing waste (scales, shells, frames, backbones, viscera, head, liver, skin, belly flaps, dark muscle, roe, and others) and bycatch (unwanted fish and fish of poor economic value). A large corpus of grey literature promotes the use of such material for the production of nutrients, nutraceuticals and pharmaceuticals (Venugopal, 2018), however most processed fish by-products are reduced to fish meal, fish oil and fish silage (A. Jackson & Newton, 2016; Venugopal, 2018).

The potential of using these byproducts is important. Jackson and Newton (A. Jackson & Newton, 2016) estimate that the collection and processing of all byproducts not currently used for fish oil extraction would yield around 50,000 tons of EPA and DHA (eicosapentaenoic acid [20:5n-3] and docosahexaenoic acid [22:6n-3]) with around 80% coming from wild capture fisheries. This additional tonnage of EPA and DHA would increase the global supply by around 25%.

3.3.1.5.3 Recreational fisheries

Recreational fisheries are defined as the fishing of aquatic animals that do not constitute the individual's primary source of nutrition and are not sold or traded on any market (FAO, 2012b). Recreational fishing is one of the most popular leisure activities in inland waters and coastal zones worldwide, with about 11.5% of the world's population involved (Arlinghaus, Tillner, & Bork, 2015; Steven J. Cooke & Cowx, 2004; Kelleher *et al.*, 2012). In industrialized countries, this proportion can be much higher, exceeding 30% (e.g., Norway) (Arlinghaus *et al.*, 2015).

Benefits derived from recreational fisheries include substantial economic benefits in the form of expenditures and related infrastructure (Cisneros-Montemayor, Sumaila, Kaschner, & Pauly, 2010; Potts, Childs, Sauer, & Duarte, 2009), an increase in the stability of the employment buffer through increased year-round or seasonal tourism employment (Smith, Khoa, & Lorenzen, 2005), psycho-social benefits (Floyd, Nicholas, Lee, Lee, & Scott, 2006; Parkkila *et al.*, 2010), and recreational fisher involvement in conservation efforts such as habitat restoration, citizen science, and research (Copeland, Baker, Koehn, Morris, & Cowx, 2017; Tufts, Holden, & DeMille, 2015).

While commercial fisheries catch by country are documented since 1950 by the FAO, data for global marine recreational catches remains scarce. (Freire *et al.*, 2020) reported three published estimates, one of 0.5 million tons

Box 3 7 The promising potential of cone snails.

Molluscs have long been used in traditional medicine and scientists often rely on local knowledge to identify bioactive compounds with potential therapeutic applications (Benkendorff *et al.*, 2015). In this context, one of the most studied groups of organisms are the cone snails, renowned for their capacity to produce venoms used to capture their prey or deter predators (Dutertre *et al.*, 2014). Cone snails are only the tip of the iceberg: order Neogastropoda, has at least 15,000 recorded species, most of which are suspected to be venomous (Puillandre *et al.*, 2011).

Venoms produced by cone snails (termed "conotoxins") have been studied since the end of the 1970s, and constitute an inexhaustible reservoir of toxins, with more than 1,000 species and up to 200 unique toxins produced by each of them (Olivera, 2006). One toxin of cone snail has been approved to be used as an analgesic to treat chronic pain (PRIALT®). Several others are engaged at various steps of the process of drug approval, with applications such as epilepsy, cardioprotection and diabetes (Bjørn-Yoshimoto *et al.*, 2020).

Such promising applications make the cone snails (and relatives) an attractive group of organisms for pharmacological companies. However, the only source of toxins is natural populations (cone snails are highly difficult to reproduce in captivity (Perron, 1981). Researchers are now looking for sustainable solutions to preserve the biodiversity.

The Nagoya protocol regulates access to genetic resources to guarantee fair benefit sharing with local populations. This is the case, for example, with cone snails that mostly live in tropical shallow waters of emerging countries. Indeed, the highest diversity of cone snails is encountered in the Indo-Pacific (Puillandre *et al.*, 2014), specifically in the Southwest Pacific (e.g., Philippines, Indonesia, Papua New Guinea, New Caledonia), and the most studied species, such as *Conus textile* or *Conus geographus*, the latter being the only deadly species for humans, with a fatality rate of 50% (Kohn, 2018), live in these regions. There, cone snails are harvested for aesthetic reasons, and if local populations harvest common species to sell them to tourists, rare species are subject to an active international market reserved to specialists. Restrictions are applied regardless of intent. Strict application of the Nagoya protocol in a growing number of countries also affects scientific study of biodiversity. The impact of sampling in the field for scientific purpose has been claimed to be negligible compared to the impact of tourists and collectors (Duda *et al.*, 2004), the latter being itself considered to be negligible in regard to the impact of human-mediated environmental changes (Peters, O'Leary, Hawkins, & Roberts, 2016).

per year from FAO approximated recreational catches (marine and inland) based on a questionnaire answered by people in 30 mostly developed countries. A second estimate reached 10.9 million tons per year was derived from an extrapolation of Canadian recreational participation and catch rates, and included both marine and inland areas (Steven J. Cooke & Cowx, 2004). Freire *et al.* (2020) describe estimates of likely marine recreational catches for 1950–2014, based on independent reconstructions for 125 countries. Those estimates of marine recreational fisheries show that catches grew globally until the early 1980s, stabilized during the 1990s, and began increasing again thereafter, amounting to around 900,000 tons in 2014. Marine recreational catches account therefore for slightly less than 1% of total global marine catches **(Figure 3.39)**. Trends vary regionally, decreasing strongly in North America, slightly decreasing in Europe and Oceania, while increasing in Asia, South America and Africa. The derived taxonomic composition indicates that recent catches were dominated by Sparidae (12% of total catches), followed by Scombridae (10%), Carangidae (6%), Gadidae (5%), and Sciaenidae (4%). The importance of Elasmobranchii (sharks and rays) in recreational fisheries in some regions is of concern, given the life-history traits of these taxa. Preliminary catch reconstruction, despite high data uncertainty, should encourage efforts to improve national data reporting of recreational catches **(Figure 3.40)**.

In Europe, the majority of recreational and tourism fishing is carried out in the Mediterranean Sea (Antunes *et al.*, 2015; Cillari *et al.*, 2012; Lloret *et al.*, 2018; Marengo *et al.*, 2015; Mavruk, Saygu, Bengil, Alan, & Azzurro, 2018; Ulman *et al.*, 2015b; Ulman & Pauly, 2016), although some of these fishing practices take place in the Atlantic coast or its islands and archipelagos (Carvalho *et al.*, 2017; Das & Afonso, 2017). It is well established in the literature reviewed that the recreational small-scale fisheries performed in Europe is not sustainable, and only 30% of the studies reviewed show any level of sustainable exploitation of recreational small-scale fishing activities (see the data management report for Chapter 3 systematic literature review at https://doi.org/10.5281/zenodo.6452651). Unsustainability is assumed due to the lack of regulation of the recreational fishing activity in general.

The available information shows that the majority of this practice is not necessarily linked with the tourism industry (Cillari *et al.*, 2012). Instead, it is usually carried out by locals as cultural practices that maintain important connections between communities and nature. This allows for some territorial overlap, and consequently for some level of competition with other fishing practices, mainly the commercial fishing for food and feed (Carvalho *et al.*, 2017; Das & Afonso, 2017; Marengo *et al.*, 2015). Although the European regulation of fisheries in general tends to be

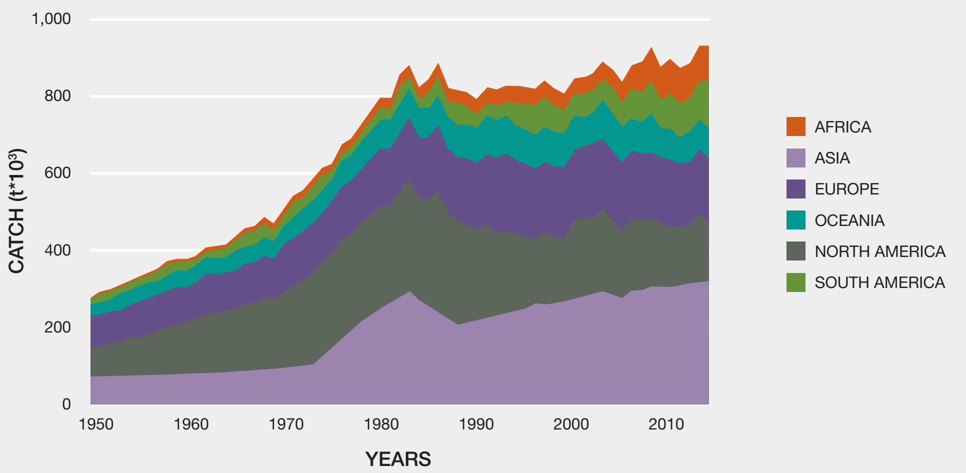


Figure 3 39 **Global marine catches from recreational fisheries by major geographic region for 1950–2014 for all countries with marine recreational fisheries.**

Source: (Freire *et al.*, 2020) under license CC BY 4.0.

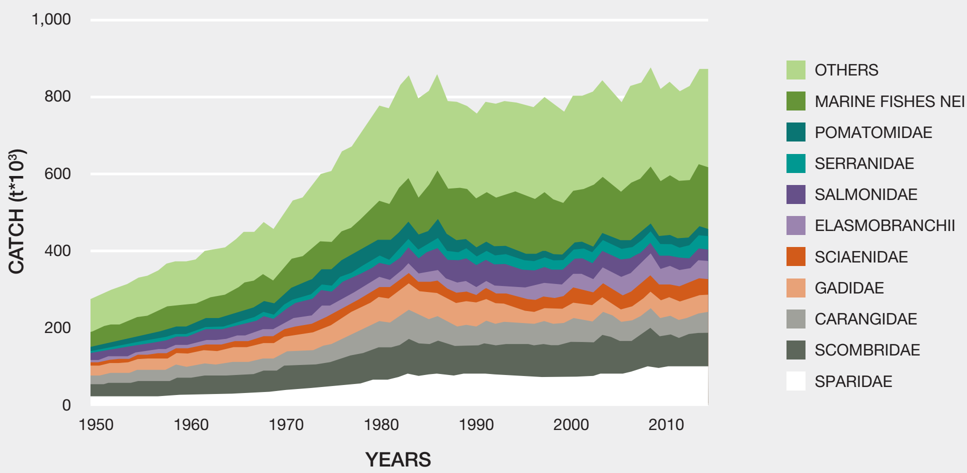


Figure 3 40 **Taxonomic composition of global recreational catches by the nine most represented families or higher groupings.**

'Marine fishes nei' (nei, not elsewhere included) comprises a large contribution of taxonomically unidentified catches; while 'Others' comprises all additional taxa with minor contributions pooled.
Source: (Freire *et al.*, 2020) under license CC BY 4.0

very widespread, most of the recreational fishing practices are not formal, and are therefore unregulated (Lloret *et al.*, 2018). On the other hand, sustainable recreational fishing practices in Europe are probably due to the use of more selective gears (Cillari *et al.*, 2012), and those that are carried out in marine protected areas or other specific areas designated by local management arrangements (Marengo *et al.*, 2015).

In Africa, despite the small number of studies on small-scale recreation and tourism fisheries, the reviewed scientific literature suggested that this type of fishing is unsustainable (Belhabib *et al.*, 2016; Leeney, 2016, 2017; Leeney & Poncelet, 2015; McCafferty *et al.*, 2012). This unsustainability is assumed due to strong fishing pressure, and lack of regulation and monitoring which means there is a relative lack of data available. The assessments cover most of the West Coast, encompassing many fish species and also a good part of the East Coast for the recreational fishing industry, mainly targeting the sawfish (see the data management report for Chapter 3 systematic literature review at https://doi.org/10.5281/zenodo.6452651). This later fishing practice is experiencing a strong decline in the last decades and in some areas the sawfish is now rarely detected.

In Latin America only two studies evaluated recreational fisheries, but in both cases these fisheries co-occur with artisanal commercial fisheries that exploit fish for food and none were considered as being fully sustainable (see the data management report for Chapter 3 systematic literature review at https://doi.org/10.5281/zenodo.6452651). The increase in recreational spear fishing caused an unsustainable decline on catches and sizes of three reef fish species in Chile (Godoy *et al.*, 2010). The tourism related to fishing, either in the form of tourists fishing for recreation or eating fish in hotels and restaurants, has increased over time and is an important economic activity in the Bahamas and other Caribbean Island countries (Smith & Zeller, 2016). However, the recreational catches related to tourism, about half of total catches in the Bahamas, have been unreported and poorly regulated, which is again assumed to compromise sustainability over time (Smith & Zeller, 2016).

In North America, most of the reviewed studies address the recreational coastal fisheries in the United States of America, especially in Florida (see the data management report for Chapter 3 systematic literature review at https://doi.org/10.5281/zenodo.6452651). Some of these studies on recreational fisheries of the bonefish *(Albula vulpes)* indicate an unsustainable pattern of decline in abundance and size of this fish species, which has suffered increased fishing effort and post-release mortality, leading to an overexploited catch-and-release fishery with negative population effects (Frezza & Clem, 2015; Rehage *et al.*, 2019; R. O. Santos, Rehage, Kroloff, Heinen & Adams, 2019). Another study showed anglers and guides are environmentally conscientious and self-aware of potential anthropogenic drivers of bonefish decline, which may have also been influenced by climate and water quality (Kroloff *et al.*, 2019). A study analyzing 22 fish species of the snapper-grouper reef fish complex in the Florida Keys reported that the majority of these species have been fished unsustainably, though overfishing appears most severe for those long-lived, slow-growing fish (Ault *et al.*, 2005). The only inland study that provided a comprehensive review of recreational and other fisheries in the region of Great Lakes and Mississippi River (United States of America and Canada). It describes internal threats such as overexploitation and bycatch/release mortality, as well as external threats such as inter-sectoral conflicts, environmental change (e.g., habitat alteration and fragmentation), water availability, and introduction of non-native species and pollution (Cooke & Murchie, 2015).

In Asia-Pacific, only four reviewed studies address recreational coastal or inland fisheries (see the data management report for Chapter 3 systematic literature review at https://doi.org/10.5281/zenodo.6452651). Although accounting for only a small fraction of the total reconstructed fisheries catches in 25 Pacific Island countries, recreational fisheries have economic relevance to local coastal communities, as these fisheries are related to tourism (Zeller *et al.*, 2015). The occurrence of whale sharks, which are not commercially exploited and are regularly sighted by fishers, indicates opportunities for the development of non-extractive tourism activities based on observation of whale sharks and promoting collaboration and use of fisher indigenous and local knowledge in Eastern Indonesia (Stacey *et al.*, 2012). The two cases of inland recreational fisheries indicate potentially overfished populations of crayfish *(Euastacus armatus)* in Australia (Zukowski, Curtis, & Watts, 2011) and more sustainable fisheries of migratory fish in the lower Mekong River basin (Mattson, 2006). Manta rays *(Manta alfredi)* have been exploited possibly at unsustainable levels for food and medicinal use in the Philippines (Acebes *et al.*, 2016).

Recreational fisheries are of concern as fishers concentrate their effort on specific areas, times, species and sizes, leading to greater impacts on targeted stocks. For instance, the nearshore zones more intensively exploited by marine recreational fishers are often critical habitats for multiple life stages of many fish (e.g., spawning, nursery), and immature life stages may be targeted in these areas (Steven J. Cooke & Cowx, 2004). Recreational fishers also selectively target larger and older “trophy” fish, often of keystone, top-predatory species, with life-history characteristics that make them vulnerable to exploitation (late age-at-maturity, low fecundity), which can lead to demographic or evolutionary effects on fish populations (Robert Arlinghaus & Cooke, 2009; Lewin, Arlinghaus, & Mehner, 2006; J Lloret *et al.*, 2020; Prato *et al.*, 2016) and community changes (e.g., successful invasion by non-native species) (FAO, 2012b). Recreational fishers can be regarded as keystone top-predators (Hilborn & Walters, 1992) with increasing efficiency, as knowledge (techniques, areas, seasons, species, etc.) is becoming more accessible and technology (GPS, sounders, braided lines, etc.) more affordable (Griffiths *et al.*, 2010).

Hence, recreational fisheries are now widely recognized as a significant component of marine capture fisheries and a potentially significant contributor to fish declines along with the commercial fleets (Agius Darmanin & Vella, 2019; Robert Arlinghaus & Cooke, 2009; Herfaut, Levrel, Thébaud, & Véron, 2013; Pawson, Glenn, & Padda, 2008). To achieve sustainable fisheries management, it appears essential to incorporate recreational fisheries stock assessments (Gordoa, Dedeu, & Boada, 2019).

3.3.1.5.4 Decorative and aesthetic

Some animal parts are used to make perfumes, mainly as a fixative substance that includes musk, ambergris, civet and castoreum. Of these four animal products, ambergris is jetsam coprolite which originates from the sperm whale (Macleod, Sinding, Olsen, Collins, & Rowland, 2020). It has been found rarely but this is in practice for centuries all over the world. It is difficult to estimate the sustainability of the ambergris gathering, as some samples have been present in the environment for about a thousand years (Rowland, Sutton, & Knowles, 2019).

The rest of this section was written following the methods used for the systematic review described in 3.3.1.3 (see the data management report for Chapter 3 systematic literature review at https://doi.org/10.5281/zenodo.6452651).

In Europe, only a fraction of the small-scale fisheries exploits aquatic animals for uses other than food. These organisms are usually benthic invertebrate species, which are not only fished for food and feed (Duncan *et al.*, 2016; Pita *et al.*, 2019), but also for a limited number of other uses. Some Porifera are traditionally used and sold as sponges for baths, for instance (Fourt *et al.*, 2020). The literature is unresolved on the sustainability of these practices. In recent decades, traditional gear was replaced by modern technologies, such as trawls (Pita *et al.*, 2019), which in combination with increased demand, led to overfishing (Fourt *et al.*, 2020). Some stocks collapsed, although when this happened is unclear. However, more recently strong control of the catch along with other introduced management measures have resulted in the sustainability of this fishing practice being slowly rebuilt (Fourt *et al.*, 2020). However, in most places the uncontrolled use of trawls is still a severe threat to the sustainability of megabenthic fauna, either for the exploited stocks or for other species of demersal fish, which are equally important for the European economy (Duncan *et al.*, 2016).

In Latin America, few studies address other uses than food, including ornamental fish to aquarium trade, decorative (handcrafts) or medicinal uses, and often these alternative uses can be made of the same organisms, some of which are also used as food (see the data management report for Chapter 3 systematic literature review at https://doi.org/10.5281/zenodo.6452651). None of the 14 studies that focused on these alternative uses addressed the sustainability of the practices. Some studies indicated partially sustainable uses of medicinal or decorative fish species on the Brazilian coast, which may occur at a local scale (low fishing pressure), but may sometimes include threatened species or be linked to trawling and by-catch (Eduardo *et al.*, 2020; Pinto, Mourão, & Alves, 2015; Rosa *et al.*, 2011; C. A. B. Santos & Nóbrea Alves, 2016). The medicinal or decorative use of parts of sharks (mostly by finning) and sawfish are regarded as unsustainable, leading to declines in the exploited species (Barbosa-Filho *et al.*, 2019; Bonfil *et al.*, 2018). Fisheries exploiting jellyfish mostly for food, but including many occasional uses as food for livestock or aquaculture, bait, medicine or aesthetic (collagen) have developed at different stages in several South American countries (Brotz *et al.*, 2017). These fisheries may be considered as partially sustainable, or potentially sustainable, given limited data on landings, potential problems of bycatch and habitat damage (depending on fishing technique) and coastal pollution from jellyfish processing (Brotz *et al.*, 2017).

In North America, no uses other than food and recreation were observed among the reviewed studies in this region (see the data management report for Chapter 3 systematic literature review at https://doi.org/10.5281/zenodo.6452651).

In Asia-Pacific, only a few studies (8) from the reviewed coastal and inland small-scale fisheries mention uses other than food, such as ornamental or decorative (see the data management report for Chapter 3 systematic literature review at https://doi.org/10.5281/zenodo.6452651). The use of marine shellfish shells as ornaments and handicrafts in Fiji is likely partially sustainable due to a recovery of exploited shellfish populations as a result of co-management (Thaman *et al.*, 2017).

3.3.1.5.5 Ceremony and cultural expression

For many small-scale fishing societies, successful fishing does not depend only on technical procedures, but rather on religious and cultural rituals and practices. Good fishing implies that propitiatory practices, such as fasting, specific diets (Teiwaki, 1988) or sex avoidance (Deb, Haque, & Thompson, 2015; Hoeppe, 2007), accompany the various stages of the technical process, including the manufacture of canoes used in fishing practice (Foale, Cohen, Januchowski-Hartley, Wenger, & Macintyre, 2011). Rituals may also be led by shamans (Ivanoff, 1992; Laugrand, 2015) or marabouts (Artaud, 2016). Thus, many taboos are meant to favour 'luck' or prevent the breach of rules and ward off the ontological imbalances resulting from a non-respect of the rules (Artaud, 2016, 2020). In fishing societies these rituals play an important part because marine species are perceived as 'partners' (Astuti, 1995; Bataille-Benguigui, 1981; D'Arcy, 2008) rather than simply as 'prey' or 'resources'. Bonds of seduction or alliances (Robert Earle Johannes, 1981; Zerner, 2003), fraternity (Grimble, 1989; Lewis, 1994), co-substantiality (Laugrand, 2015; N. Peterson & Rigsby, 2014) or consanguinity (Ivanoff, 1992) unite human and aquatic communities. Beyond these relationships, fishing rituals aim at strengthening the ties between people, social groups or clans. For instance, for the Tao people, fishing flying fish (Family Exocoetidae) is an opportunity to renew a set of cultural and identity principles and values (Berger, 2019; Fan, 2019; Gaffric, 2013). The same is true of whale hunting, which for several indigenous communities constitutes a means of regulating group relations or asserting their singularity within a State (see section 3.3.1.4.6) (Adamson, 2012; Deutsch, 2017). It is also the case for salmon fishing among the Ainu (Iwasaki-Goodman & Nomoto, 2001). Items from aquatic species, such as sea-shells, are used in some rituals, for instance in the candomblé, an Afro-Brazilian religion (Neto, Voeks, Dias, & Alves, 2012).

3.3.1.6 "Non-lethal" fishing practices and uses

3.3.1.6.1 Catch and release recreational fishing

Recreational fishing can involve a variety of gear types but catch-and-release fishing is most typically focused on fish caught by hook and line (FAO, 2012b). Therefore, this discussion is focused on angled fish.

With respect to recreational catch-and-release fishing, it is difficult to disentangle the socio-economic benefits of harvest *versus* release-oriented recreational fishing, which collectively generates over 100 billion United States dollars annually while creating opportunities for anglers to connect with nature and spend time with friends and family (Robert Arlinghaus & Cooke, 2009). Recreational fisheries certainly can and do involve harvest for personal consumption (Steven J Cooke *et al.*, 2018), but harvest rates vary markedly among regions, species, and angler typologies. To emphasize that variation, recreational harvest rates of species like muskellunge *(Esox masquinongy)* and bonefish (*Albula* spp) are around 1% while species like walleye *(Sander vitreus)* and Atlantic cod *(Gadus morhua)* have harvest rates that typically exceed 60% (Robert Arlinghaus *et al.*, 2007). In some cases, release of fish is dictated by regulations (e.g., closed seasons, bag limits, size limits) but it can also be voluntary. Where there are long term trend data available, there is evidence that fish release rates have crept up slowly over time (e.g., Brownscombe *et al.*, 2014).

The release of angled fish requires proper handling and not all fish survive (Cooke & Schramm, 2007). From a sustainability perspective, it is irrelevant whether fishing mortality arises from harvest (i.e., from an extractive fishery) or from release mortality (i.e., in a non-extractive fishery). Catch and release mortality rates are highly variable and can range from near total mortality to near total survival (recognizing that zero mortality is never attainable). Several syntheses suggest that the bulk of recreational fisheries exhibit release mortality rates that are less than 10% (Arlinghaus *et al.*, 2007; Bartholomew & Bohnsack, 2005; Muoneke & Childress, 1994). Although mortality rates are informative, alone they provide little information on the population-level consequences of release mortality (Kerns, Allen, & Harris, 2012). Information on fishing effort, life history characteristics, population status, and the role of other fisheries practices dictate whether catch and release mortality threatens the sustainability of fish populations. Mortality arising from catch and release is often cryptic and has been implicated in fisheries collapse (Post *et al.*, 2002; Schroeder & Love, 2002). There are many factors that determine whether an individual fish will survive a catch and release event. The single biggest driver of mortality is anatomical hooking location with fish hooked in the jaw region having comparatively low mortality relative to fish hooked more deeply in areas such as the gills or esophagus (Arlinghaus *et al.*, 2007).

Recreational catch and release fishing can have consequences for aquatic and coastal habitats. Issues include tackle loss (e.g., lead sinkers, hooks, line), littering, trampling of shoreline vegetation and in-water habitats (e.g., coral, gravel spawning sites), erosion, noise pollution, and hydrocarbon release from boats, and accidental or intentional release of exotic species (e.g., bait bucket transfers, stocking), among others (reviewed in Cooke & Cowx, 2006; Lewin *et al.*, 2006; McPhee *et al.*, 2002; Venohr *et al.*, 2018).

Many fishing guides and outfitters pride themselves on their operations being catch and release focused and use that in marketing. A number of non-governmental organizations focus on educating anglers on how to engage in responsible catch and release. Moreover, governments routinely apply harvest regulations as part of their fisheries management initiatives in an effort to create sustainable fisheries that benefit aquatic ecosystems and the humans that use them. Thus, catch and release activities and the associated tour operators contribute to creating responsible and sustainable recreational fisheries (Cooke *et al.*, 2019).

3.3.1.6.2 Ornamental or aquarium fish

Ornamental fish trade is a global, multibillion-dollar industry, involving over 125 countries (Evers *et al.*, 2019a) and worth billions of United States Dollars. Ornamental fisheries are divided into marine and freshwater fisheries. Some of the original representative sustainable gathering projects of ornamental fishes are losing their competitiveness due to the rise of off-site aquaculture.

The freshwater ornamental fish trade involves about 125 countries worldwide, is worth approximately 15-30 billion United States dollars (Evers, Pinnegar, & Taylor, 2019b; Penning *et al.*, 2009) and trading around 1.5 billion specimens per year (C. H. Stevens, Croft, Paull, & Tyler, 2017). Roughly 1,000 of the over 5,300 freshwater fish species traded are widely available in commercial numbers (Evers *et al.*, 2019b). A big difference is that around 90%

of freshwater ornamental fishes are farmed, usually in Asia or South America, but also in Israel, the United States of America and Europe. Although a smaller portion of freshwater ornamental fishes are still sourced from the wild, in comparison to marine ornamental fishes, it is still a challenge to determine the volume due to lack of reliable data.

The marine aquarium trade supplies public and private aquariums with a large diversity of organisms (Dey, 2016; Wabnitz, 2003). A review found that an estimated 15-30 million specimens of coral reef fishes are extracted each year from tropical coral reefs (Biondo & Burki, 2020). The review did not assess mortality rates (Stevens *et al.*, 2017), making proper harvest estimates more challenging since they cannot be based on trade data (Cohen, Valenti, & Calado, 2013; Militz, Kinch, Foale, & Southgate, 2016; Monticini, 2010; Olivier, 2001; C. H. Stevens *et al.*, 2017; Thornhill, 2012). Most marine ornamental species are being collected from the wild (Biondo, 2017, 2018; Biondo & Burki, 2019; V. Dey, 2016; Rhyne *et al.*, 2012; Rhyne, Tlusty, Szczebak, & Holmberg, 2017; Wabnitz, 2003) including species that are listed as endangered by the International Union for Conservation of Nature Red List, such as the Banggai cardinalfish *(Pterapogon kauderni)*. Of the approximately 4,000 marine ornamental fishes known to date (R. Froese & D. Pauly, 2019), about 2,500 species are in trade (Rhyne *et al.*, 2012, 2017). Of all these species only around 25 species (1%), can be bred in commercial numbers and about 300 have been bred successfully in research stages (Pouil, Tlusty, Rhyne, & Metian, 2020).

The International Union for Conservation of Nature Red List category is a starting point to warrant protection of a species, but many species of reef fishes are currently labelled 'not evaluated' and 'data deficient': 73.3% in 2014 and 44.8% in 2018, meaning that the conservation states for almost half of the species is still unknown (Biondo, 2018). Protection from international trade would come through the Convention on International Trade in Endangered Species but only few species are listed on its appendices (e.g., *Hippocampus* spp. *Cheilinus undulatus*, *Holacanthus clarionensis*) thus very little specific trade data is collected (https://cites.org/eng/app/appendices.php, (CITES, 2012).

It is estimated that over 50 countries are actively involved in the marine aquarium industry (Biondo & Burki, 2020; Rhyne *et al.*, 2012, 2017). However, this trade lacks sufficient monitoring, and the specific geographic origin of most specimens uncertain (Biondo & Burki, 2019, 2020; Biondo & Calado, 2021; Cohen *et al.*, 2013; Ploeg, 2007). The largest exporting markets are Indonesia, the Philippines and Sri Lanka (Rhyne *et al.*, 2012, 2017; Wabnitz, 2003). While some analyses have tried to estimate trading figures for large importing markets, such as the United States of America (Rhyne *et al.*, 2012, 2017), Australia (Trujillo-González & Militz, 2019), and Europe (Biondo, 2017, 2018; Biondo & Burki, 2019; Leal *et al.*, 2016), they all represent approximations and the figures presented are most likely underestimates. Japan is mentioned in the literature as a large importer, but with no recent trade figures available (Biondo, 2017, 2018; Biondo & Burki, 2019, 2020; Rhyne *et al.*, 2012, 2017; Wabnitz, 2003). Furthermore, there is no information at all for growing markets, such as those located in Southeast Asia, Africa, and South America (Biondo & Calado, 2021).

With regards to the literature focused on small scale fishing, in Latin America the nine studies addressing small-scale fisheries of ornamental fish for aquarium trade included only three studies in the Brazilian coast (Eduardo *et al.*, 2020; Monteiro-Neto *et al.*, 2003; Rosa *et al.*, 2011), while all others focus on freshwater fisheries in the Peruvian and Brazilian Amazon (Araújo *et al.*, 2020; Evers *et al.*, 2019a; Gerstner, Ortega, Sanchez, & Graham, 2006; Guzmán Maldonado *et al.*, 2017; Ladislau *et al.*, 2020; Moreau & Coomes, 2007). A study in the Brazilian coast shows an increasing trend in the number of fish (mainly native reef species) caught and traded, mostly for export, but there are no data on fishing effort or population status of exploited fish to check for the sustainability of such large trade (Monteiro-Neto *et al.*, 2003). Other studies in the Peruvian and Brazilian Amazon indicate that this activity may be unsustainable due to illegal fishing, rapid expansion of fishing effort, reduced fish abundance in more heavily fished regions compared to protected areas and synergic effects of intense exploitation, market pressure (increased sale prices) and habitat change caused by dams (Evers *et al.*, 2019a; Gerstner *et al.*, 2006; Guzmán Maldonado *et al.*, 2017). Studies addressing either coastal or inland ornamental small-scale fisheries expressed concerns on unreported and unknown fish mortality during collection and transportation (Monteiro-Neto *et al.*, 2003; Moreau & Coomes, 2007).

In Africa, only a small number of papers dealt with small-scale fisheries for ornamental trade, both in the coral reefs off the coast of Kenya. Some of the many species studied proved to be at low risk of overexploitation, mainly because there is large and disseminated use of very selective gears to capture the fish in this type of fishing. This selectivity allows the removal of mature, large (and colorful) individuals above the maturation size (Gomes, Erzini, & Mcclanahan, 2014). On the other hand, some other species are vulnerable to overfishing and other species are probably already overfished (Okemwa, Kaunda-Arara, Kimani, & Ogutu, 2016). This overfishing is due to low natural abundance and long-term intense fishing pressure. In those cases, more active management measures could mitigate threats to vulnerable species.

In Asia-Pacific, data from both fishers' knowledge and recordings of fish landings (logbooks) of seahorses

(Hippocampus comes), which are exploited as ornamental and medicinal fish in the Philippines, indicate that catch-per-unit-of-effort did not change over a period of nine years (O'Donnell *et al.*, 2012). Fishers' logbooks that included zero catches (fishing trips on which no seahorse was caught) showed the lowest catch-per-unit-of-effort values, and a previous study based on fishers' indigenous and local knowledge indicated declines of seahorse catches from 1970 to 2005 (O'Donnell, Pajaro, & Vincent, 2010). Some studies point to unsustainable rates of exploitation of sea horses (*Hippocampus* spp.) on the coast of Vietnam (Stocks *et al.*, 2019; Stocks, Foster, Bat, & Vincent, 2017). In India, nearly 50% of marine aquarium fish and corals, considered highly financially valuable species, have not been assessed for their extinction risk (Prakash *et al.*, 2017). The ornamental fisheries of corals (many species) and the coastal fish *(Pterapogon kauderni)* in Indonesia are considered to be unsustainable, due to intensive fishing pressure, habitat damage, or overestimated quotas beyond ecological capacity (Ferse *et al.*, 2012; Kolm & Berglund, 2003). A recent monitoring survey of Banggai Cardinalfish populations shows mixed trends from 2004 to 2018 among seven sites in Indonesia: recovery or partial recovery in three sites, stable in one, increase in one and decline in two sites, indicating potential effects from conservation measures in some sites and the relevance of microhabitats (sea urchins and sea anemones) to juveniles and adults of this reef fish (Wiadnyana *et al.*, 2020).

Morevoer, some marine protected areas, which were created to protect reef fish for ornamental aquarium trade in Hawaii, have increased abundance of some exploited species and thus possibly improved the sustainability of these commercially valuable ornamental fisheries (Friedlander *et al.*, 2014). The few studies on inland ornamental fisheries indicate potential unsustainable harvest, due mostly to intense fishing effort and weak regulations (see the data management report for Chapter 3 systematic literature review at https://doi.org/10.5281/zenodo.6452651). The increase on fishing effort had caused overexploitation of wild caught populations of the clown loach or tiger botia fish *Chromobotia macracanthus* in Indonesia, which directed fishers to catch fish larvae to be reared in captivity (Evers *et al.*, 2019a). In India, thousands of individuals of threatened and endemic freshwater fish species have been regularly caught and sold for high values in the export market, stimulating an intense fishing pressure (Raghavan *et al.*, 2013).

The aquarium trade of ornamental fishes is usually considered as a profitable, rapidly developing, but somewhat unpredictable economic activity, which is subjected to sudden fluctuations in the international market and may involve high operational costs, either for coastal (Monteiro-Neto *et al.*, 2003) or inland fisheries (Araújo *et al.*, 2020; Moreau & Coomes, 2007). Trade includes many dealers with large differences in prices paid between fishers and final retailers (Rosa *et al.*, 2011). For example, the well-established aquarium trade in the Negro River (Brazilian Amazon), which exploits mainly the small cardinal tetra fish *Paracheirodon axelrodi* for the international market, has experienced problems related to the productive chain, such as competition with international producers, absence of local buyers, decrease on sales and lower profits, making some fishers abandon this activity (Evers *et al.*, 2019a; Ladislau *et al.*, 2020).

The global trade of marine ornamental fishes has always lagged behind in terms of transparency, as there is a multitude of stakeholders involved from the fishers at location of capture to the (many) intermediaries and traders, the exporters and importers and the intermediaries in the importing countries (Amos & Claussen, 2009). Some attempts have been made to increase transparency in the marine ornamental fish industry. The Global Marine Data Base (GMAD) was introduced in 2002 and collected importer and exporter data but with only 41 contributing companies and unfortunately, only for one year (Green, 2003). Another attempt was the Marine Aquarium Council label that was established in 1998 to ensure traceability, good practice, and sustainable schemes of ecologically and socially responsible fishing, but has been inactive since 2008 (Dee, Horii, & Thornhill, 2014).

3.3.2 Gathering

3.3.2.1 Introduction

Wild algae, fungi and plants provide food, income and nutritional diversity for an estimated one in five people around the world, in particular women, children, landless farmers and others in vulnerable situations (Sorrenti & Food and Agriculture Organization of the United Nations, 2017). The Plant List (http://www.theplantlist.org/) and the World Flora Online (WFO, http://www.worldfloraonline.org/) list around 360,000 species with accepted names (accessed January 2021). The world checklist of vascular plants includes approximately 350,000 accepted species. With regards to fungi, 148,000 species have been scientifically identified, but it is believed that more than 90% of species remain unknown to science (Antonelli *et al.*, 2020).

Gathering is defined in the sustainable use assessment as the removal of terrestrial and aquatic algae, fungi, and wild plants or parts thereof from their habitats. This definition includes leaves and fruits of trees. Whole tree or excessive branch removal of trees is discussed under logging (see Chapter 1 for complete definitions of all practices). Gathering may, but often does not, result in the death of the organism. All wild plants, fungi, and parts of plant and fungal bodies harvested in forests, savannas, and grasslands that are not wood harvested for timber are broadly categorized as algae, fungi and plants (Sorrenti & Food and Agriculture Organization of the United Nations, 2017).

Exploitation of wild algae, fungi and plants often involves the systematic removal of biological, units or parts of units, from a population, but the level of mortality in the exploited population depends on methods of extraction and the vital parts that are removed (Ticktin, 2004). Local communities and indigenous peoples harvest wild algae, fungi and plants for primary health care, basic livelihood needs, to provide social safety nets, and subsistence income. Traditional algae, fungi and plants gathering, for either subsistence or commercial purposes, is often considered a desirable, low-impact economic activity from wild habitats, compared to alternative forms of land use that involve structural disturbance such as selective logging (Plotkin, Famolare, Conservation International, & Asociación Nacional para la Conservación de la Naturaleza, 1992). Gathering is also an important cultural and recreational activity for many, pursued by individuals and family groups even where there is no pressing financial need (Emery, 2001).

A majority of wild algae, fungi and plants gathering was considered ecologically and economically sustainable in a recent review (de Mello, Gulinck, Van den Broeck, & Parra, 2020; Stanley, Voeks, & Short, 2012). Therefore, exploitation of wild algae, fungi and plants, as such, is usually assumed to be sustainable and is viewed as a best compromise between the requirements of biodiversity conservation and those of extractive communities under varying degrees of market integration. However, commercial harvesting of wild plants has increased in recent years, for food, the pharmaceutical and cosmetic industries, as well as for artisanal herbal teas, natural dyes, and decoration. Due to the wide variation in the nature of wild algae, fungi and plants and the way they are harvested and traded, the sustainability of intensive harvesting for trade is debatable (Isabel B. Schmidt, Mandle, Ticktin, & Gaoue, 2011).

The number of people who participate in gathering provides one measure of the significance of this practice to nature's contributions to people. Data on numbers of people who gather globally are incomplete and differences in methodologies vary such that direct comparison of results across studies is difficult. The challenge of assessing numbers of people who gather are compounded by inter-annual variation in gathering, by individuals and households in response to changing needs and opportunities, and as availability of individuals with the desired characteristics ebbs and flows (Lovrić *et al.*, 2020; Watson *et al.*, 2018). With those caveats, available data suggest globally, numbers of people who engage in gathering are likely higher than those for other extractive practices.

Gathering is one of the practices most closely associated with traditional lifeways, subsistence practices, and indigenous and local knowledge in both high and low-income countries worldwide. Which wild species are edible and how they are processed, are essential elements of local and traditional knowledge. Most ethnobiological studies on gathering wild species for food consumption have documented edible species, parts, or processing methods. It is widely agreed upon in the available scientific literature that older women are the primary holders and stewards of indigenous and local knowledge, and pass on their knowledge through mother-child nexus and community sharing. Children from indigenous peoples and local communities have specialized access to specific wild resources, ones which are generally of lesser importance for adults and complement their diet. As almost exclusive harvesters of these resources, children retain their own sphere of knowledge and know-how. They are often neglected in considerations of gathering stakedholders, in spite of being full social actors in these societies and being engaged in transmission and exchange networks (Dounias & Aumeeruddy-Thomas, 2017).

Regarding trade in wild algae, fungi and plants, the International Trade Centre estimated that approximately 440 different organic wild products were identified as of 2005. Nearly all of them are wild plants, seaweed and mushrooms; more than half (253/440) of them are medicinal and aromatic plants. A total of 223,754 tons (t) of organic wild harvested products were harvested in 2005. The largest gathering areas were reported in Africa and Europe, while the highest quantity was reported harvested in Asia from a relatively small area (International Trade Centre UNCTAD/WTO, 2007). There is a large amount of trade in wild algae, fungi and plants in the informal economy with little or no records. However, formal markets for resins, tannins, pine nuts, wild mushrooms and other wild algae, fungi and plants in Europe are developing rapidly. In China formal markets around tea seed oil *(Camellia oleifera)*, Chinese chestnut *(Castanea mollissima)*, Persian walnut *(Juglans regia)*, Eucommia *(Eucommia ulmoides)* and purpleblow maple *(Acer truncatum)* are expanding (Sheppard *et al.*, 2020).

Regarding conservation and sustainable use of wild algae, fungi and plants, the International Union for Conservation of Nature Red List of Threatened Species contains over 9,600 wild food species of which 20% are considered threatened. Ironically, agriculture is the greatest threat to plants, followed by logging and gathering, which is only slightly more threatening than land use for residential and commercial development (Antonelli *et al.*, 2020). What part of the organism is gathered, its phenology, and life form, affects how susceptible the species is to over-harvesting **(Table 3.5)**. Gathering the flowers and fruits of annual-biennial plants shows the greatest susceptibility to

Table 3 5 **Susceptibility of wild plants to overharvesting.**

Note: + represents a high probability, ++ higher, +++ highest. Source: (modified from Lange, 2006) © 2017 Springer.

Life form / Plant part used	Tree	Shrub	Perennial herb	Annual-biennal
Wood	++	++	Not applicable	Not applicable
Bark	++	++	Not applicable	Not applicable
Root	++	++	+++	+++
Leaf	-	-	-	+
Flower	-	-	+(++)	+++
Fruit/seed	-	-	+(++)	+++

overharvesting. Gathering bark and roots also has a high probability of leading to overharvesting.

Removing the bark may threaten the survival of plant individual, for example when gathering the medicinal part of the African cherry *(Prunus africana)* (Fashing, 2004; K. M. Stewart, 2003), *Julbernardia paniculata, Isoberlinia angolensis* (Chungu, Muimba-Kankolongo, Roux, & Malambo, 2007), Himalayan yew *(Taxus wallichiana)* (Lanker, Malik, Gupta, & Butola, 2010) and Pepper-Bark Tree *(Warburgia salutaris)* (Senkoro, Shackleton, Voeks, & Ribeiro, 2019). There are many wild plants whose roots are harvested for medicinal use. Some of the most well-known are ginseng (*Panax* sp.), *Nardostachys grandiflora* (S. Ghimire, McKey, & Aumeeruddy-Thomas, 2005), Oshá *(Ligusticum porteri)* (Kindscher, Martin, & Long, 2019), Black Cohosh (*Actaea racemosa* L.) (Small, Chamberlain, & Mathews, 2011), *Cryptolepis sanguinolenta* (Amissah *et al.*, 2016), *Stemona tuberosa* (G. Chen, Sun, Wang, Kongkiatpaiboon, & Cai, 2019) and *Eurycoma longifolia* (Susilowati, Rachmat, Elfiati, & Hasibuan, 2019). The gathering of major parts of wild plants such as stems and bulbs is also common in herbaceous plants like orchids. These types of gathering activities may kill the plant and are therefore a focus for species conservation and sustainable management efforts. Sustainable harvest programs for gathering flowers, fruit and leaves for medicinal use have secondary benefits for habitat protection.

To study the sustainability of the use and gathering of wild plants a literature review was conducted and studies on the ecological aspect of specific species were collected, based on the parts gathered and the life form of plants used. Note that separate literature reviews were conducted on the sustainable use of wild fungi and for urban gathering (see the data management report for Chapter 3 systematic literature review at https://doi.org/10.5281/zenodo.6452651). For the literature review on wild plants, in accordance with the requirements of the systematic literature review, key word combinations were used such as: #gather/pick/collect# + #plant# + #wild# + #terms of the aim of uses + sustainable# and searched primarily in google scholar, Web of Science SCI (Science Citation Index Expanded) and CNKI (China National Knowledge Infrastructure). A total of 89,400 materials were identified, but most only described how the wild plants were used. Eight hundred and fourteen (814) relevant articles and reports went through the initial screening. Fifty-one (51) cases of specific plant species or groups met the search criteria for inclusion in the study of sustainable use by gathering. The relevant papers were carefully reviewed to determine the credibility of the conclusions of each set of research (see the data management report for Chapter 3 systematic literature review at https://doi.org/10.5281/zenodo.6452651). Cases of sustainable use by gathering wild plants are from all IPBES regions, including Africa (13), America (21), Asia and Pacific (11) and Europe and central Asia (6). Of the 51 cases of the use and gathering of wild plants retrieved, more than two-third of tree/shrub gathering were sustainably managed, while more than half of the gathered herbs assessed were considered unsustainable. For trees, the existing cases show that the gathering of bark for uses, mainly medicinal aims, are not sustainable due to the lack of management and regulatory systems. For herbs, gathering root for medicines from perennial herbs led to more unsustainability concerns **(Table 3.6)**.

Table 3 6 **Number of cases of sustainable use and gathering of wild plants through literature review.**

Note: the cases reported here represent those captured through systematic literature review. Additional material is included in the chapter text from contributing authors, personal experiences and expertise. Af.: Africa, Am.: Americas; AP.: Asia and Pacific; EC.: Europe and Central Asia. NA: Not applicable. Uns/Sus: number of unsustainable cases need solutions *versus* number of sustainable cases under specific management or regulations.

Life form / Plant part used	Tree/shrub (Uns/Sus)					Herb (Uns/Sus)				
IPBES Regions	Af	Am	AP	EC	All	Af	Am	AP	EC	All
Barks	3/1	0/1	1/1		4/3	NA	NA	NA	NA	NA
Sap/gum/resin	1/1	0/1	1/0	0/1	2/3	NA	NA	NA	NA	NA
Root/tuber/bulbs						2/0	2/2	5/1	1/1	10/4
Leaves						1/1	1/5			2/7
Flowers							1/0			1/0
Fruits/seeds	1/1	0/5	0/1	0/2	1/9		0/1			0/1
Whole						1/0	1/1	1/0		3/1
Sum	5/3	0/7	2/2	0/3	1/7	4/1	5/9	6/1	1/1	16/13

3.3.2.2 The diversity of contemporary gathering

3.3.2.2.1 Gathering in Western European and Other Group (WEOG) countries

Gathering wild algae, fungi and plants is often assumed to be an activity more prevalent in developing countries and the Global South, and less so in post-industrial countries. However, results of surveys conducted in Europe, North America, and the United Kingdom over the last 20 years suggest high rates of participation in gathering by individuals and households in many of the countries in these regions **(Table 3.7)**. In Scotland, a 2003 random sample general population survey found 18% of individuals had gathered fungi and tree or plant-derived materials in the previous 12 months, including residents of both urban and rural areas (Emery *et al.*, 2006). The Northeastern United States of America includes both the largest urban concentrations of that nation and substantial rural lands. Eighteen percent of respondents to a 2004 survey in that subnational region reported that they had gathered “tree or plant materials around woodlands: e.g., mushrooms, berries, cones, or moss” in the previous 12 months, while 26% had done so in the previous 5 years. Also in the Northeastern United States of America, 36% of respondents to a survey conducted over the five-year period 2005–2009 indicated they had picked mushrooms and/or berries in the previous 12 months (Cordell, Betz, Mou, & Gormanson, 2012; K. Watson *et al.*, 2018). A 2016 survey of households in 28 European countries found that Europe-wide, 26% of households had gathered in the previous 12 months, ranging from 4% of households in the Netherlands to 68% of households in Latvia (B. Wolfslehner, Prokofieva, & Mavsar, 2019). This study noted a general pattern of highest rates of gathering by households in Eastern Europe (Lovrić *et al.*, 2020). Unlike the surveys in Scotland and the United States of America, the European study documented gathering by households, suggesting that the percentage of individuals gathering in the region may be higher.

In Europe, changing patterns in wild plant and fungi use vary by country and region, associated with changing lifestyles, urbanization, large-scale farming, less periods of famine and economic hardship in recent years and changing outdoor recreation patterns. At the same time, large increases in immigrant populations are affecting what is harvested, by whom and for what purposes (Łuczaj *et al.*, 2012). In France, 728 algae, fungi and plants species are extracted from the wild, of which 100 are commonly used (Lescure, Thévenin, Garreta, & Morisson, 2015). Recent research in Norway found a total of 273 wild edible plants from 67 botanical families were identified by collectors, with the majority of harvested material coming from seven families and ten taxa. Fruits and berries, leaves and flowers were the most popular and important plant parts that were foraged by study respondents (Giraud, 2020).

Table 3 7 **Percent of population who gather in three Western European and Other Group (WEOG) subregions.**

Sources: (M. Emery *et al.*, 2006; Lovrić *et al.*, 2020; K. Watson *et al.*, 2018).

Survey location	Survey years	Unit of analysis	% Gathering (previous 12 months)
Scotland	2003	Individual	18
US Northeast	2004	Individual	18
US Northeast	2005–2009	Individual	36
Europe	2016	Household	26

Research suggests dozens to hundreds of wild algae, fungi and plants are gathered in urban, rural, and wilderness ecosystems throughout the continental United States of America, Alaska, Hawai'i and United States of America territories. Of these, a small subset enters into large-scale trade with maple syrup (*Acer* sp.), wild blueberries (*Vaccinium* sp.), and medicinal species such as American ginseng *(Panax quinquefolius)* noteworthy among them. Estimates of the number of United States of America residents who gather at least occasionally range from 18% to 36%, with the vast majority (>80%) gathering for personal use only. It seems likely, then, that a majority of United States of America residents who gather do so for personal use, while a few species gathered for commercial purposes account for the majority of biomass removed. Wild algae, fungi and plants gathering plays an important cultural role for many indigenous peoples and local communities in the United States of America including, but not confined to, those formally recognized as indigenous. Rights of access to wild algae, fungi and plants for subsistence purposes are provided for by law in the United States of America's state of Alaska (for rural residents of that state), Hawai'i (for Native Hawaiians), and under the terms of many treaties between tribes and the federal government (Chamberlain, Emery, & Patel-Weynand, 2018; Cordell *et al.*, 2012; M. R. Emery & Pierce, 2005; M. R. Emery, Pierce, & Schroeder, 2004; Hurley, Grabbatin, Goetcheus, & Halfacre, 2013; Robbins, Emery, & Rice, 2008).

Gatherers have different identities and sources of knowledge in gathering networks. For example, in Austria, organic certification for wild plants has been issued to three types of gatherers: regular, diversified and single-plant gatherers. Among them, regular gatherers are the principal knowledge sources of traditional and local knowledge, and the diversified gatherers who are less common and learning knowledge from formal courses or self-learning, may be more worried by the loss of traditional knowledge (Schunko & Vogl, 2018). In France, present professional gatherers are of multiple origins, urban or rural, and hold their knowledge from different sources. They care for the sustainability of the plants and ecosystems more than occasional opportunistic gatherers. Through their associations or cooperatives, they establish rules of good gathering practices (Lescure *et al.*, 2015) (Julliand, Pinton, Garreta, & Lescure, 2019).

3.3.2.2.2 Urban gathering

Urban gathering is an activity which supports biodiversity and sustainable human-nature interactions, but it is under-recognized as a global activity (McLain *et al.*, 2012; A. Russo, Escobedo, Cirella, & Zerbe, 2017; Tiwary, Vilhar, Zhiyanski, Stojanovski, & Dinca, 2020). Urban gathering promotes positive cultural, ecological, economic, and health outcomes (Shackleton, Hurley, Dahlberg, Emery, & Nagendra, 2017; Synk *et al.*, 2017). As a global phenomenon, it provides three categories of provisioning (woody biomass, food/fibre, and non-timber forest products), and it supports a 'green economy' (Shackleton, Chinyimba, Hebinck, Shackleton, & Kaoma, 2015; Tiwary *et al.*, 2020). Of the 43 studies related to urban gathering retrieved for this assessment, 70% are from the Americas, Europe and Central Asia, 20% are from Africa, and the remaining are from Asia and the Pacific. Common characteristics of gathering, such as health risks, ecological conditions, and pressures on wild algae, fungi and plants species are likely not the same between in rural and urban contexts, making further research on urban gathering a knowledge gap on the sustainable use of wild species for nature's contributions to people (Fischer & Kowarik, 2020; Rupprecht, Byrne, Garden, & Hero, 2015; Shackleton *et al.*, 2017; Short Gianotti & Hurley, 2016). The use of Geographic Information Systems (GIS) and spatial modelling in digital platforms and apps shows promise in quantifying urban natures as baselines for this additional research (Arrington, 2021; Moss, Voigt, & Becker, 2021).

Dozens to hundreds of feral and wild plant and fungi species are gathered for food, medicine, firewood, decoration, and cultural practices in urban ecosystems (Kaoma & Shackleton, 2015; Landor-Yamagata, Kowarik, & Fischer, 2018; Łuczaj, Wilde, & Townsend, 2021; McLain *et al.*, 2012; McLain, Poe, Urgenson, Blahna, & Buttolph, 2017; Palliwoda, Kowarik, & von der Lippe, 2017; Poe, LeCompte, McLain, & Hurley, 2014; Shackleton *et al.*, 2017; Shackleton *et al.*, 2015; Somesh, Rao, Murali, & Nagendra, 2021). In some cases, for example in Uganda, New Zealand, French Guiana, Haiti and India, wild plants are primarily gathered for medicinal purposes (Dejouhanet & de Bercegol, 2019; Mollee, Pouliot, & McDonald, 2017; Tareau, Dejouhanet, Odonne, Palisse, & Ansoe, 2019; Wehi & Wehi, 2010). However, in major urban spaces in these countries gathering wild edible plants and fungi was most commonly for food, followed by medicinal uses and personal enjoyment (Amato-Lourenco *et al.*, 2020; Garekae & Shackleton, 2020a; Landor-Yamagata *et al.*, 2018). Wild edibles, including berries, fruits, nuts, greens, and young shoots, were by far the most frequently mentioned type of product, contributing to diversifying urban diets (Garekae & Shackleton, 2020a; McLain, Hurley, Emery, & Poe, 2014; Sardeshpande & Shackleton, 2020a; Shackleton *et al.*, 2017; Somesh *et al.*, 2021).

Urban green spaces where gathering happens are promising pathways towards biodiversity conservation in cities because they facilitate interactions between people and nature which support physical and mental health (Palliwoda *et al.*, 2017). Equitable access to cultural ecosystem services from urban green space helps overcome sociocultural barriers, strengthens social relationships, maintains knowledge and traditions of families and communities, increases shares in the management of goods and services, and increases healthy food intake and personal participation in healthy behaviors (Askerlund & Almers, 2016; Jennings, Larson, & Yun, 2016; Landor-Yamagata *et al.*, 2018; McLain *et al.*, 2012; Šiftová, 2020; Tiwary *et al.*, 2020). Urban gathering can also support identity, place attachment, or mobility and agency of people and communities in the city (Poe, LeCompte, McLain, & Hurley, 2014).

Gender and income level affect urban gathering activities differently in different regions. They may be evenly distributed along gender or income categories in the United States of America, Germany and the United Kingdom of Great Britain and Norhtern Ireland (Fischer & Kowarik, 2020; Łuczaj *et al.*, 2021; McLain *et al.*, 2012; McLain *et al.*, 2014). Urban gathering in developing countries tends to be more female-dominated in some countries (Garekae & Shackleton, 2020a; Somesh *et al.*, 2021) and male-dominated in other countries (Garekae & Shackleton, 2020b). Residents with lower income and predominantly living or growing up in rural areas or peri-urban areas are more likely to be urban foragers (Garekae & Shackleton, 2020b, 2020a; Mollee *et al.*, 2017; Short Gianotti & Hurley, 2016).

Most urban gathering in the developed world is not commercially oriented; products are mainly for personal consumption and gifting (Charnley, McLain, & Poe, 2018; Rebecca J McLain *et al.*, 2014). In countries in the Global South, rapid urbanization, unplanned settlements, and poor service delivery mean that it remains vital to gather for self-provisioning and income. A substantial contribution of total household income can be generated from urban gathering, particularly in poorer households (Borelli *et al.*, 2020; Dejouhanet & de Bercegol, 2019; Kaoma & Shackleton, 2015; Somesh *et al.*, 2021). However, the potential of urban gathering to affect food sovereignty and security is not evenly distributed across socioeconomic strata (Bunge, Diemont, Bunge, & Harris, 2019).

Most gatherers acquire and pass on knowledge about gathering practices through family and friends or gathering trips (Garekae & Shackleton, 2020b, 2020a; McLain *et al.*, 2014). Oral transmission, amateur society outings, professional scientists, books, and field guides help counteract the decline in more traditional outdoor gathering activities (Łuczaj *et al.*, 2021; McLain *et al.*, 2014; Palliwoda *et al.*, 2017). Stakeholders exchange information on the nature of green spaces, species and ecosystems and allied activities. City managers can make use of gatherers' extensive local ecological knowledge to inform more formal management practices and support the overall management of urban natural areas (McLain *et al.*, 2017; Sardeshpande & Shackleton, 2020b).

Voluntary codes of conduct may be the best way to manage urban gathering to prevent over-harvesting (Charnley *et al.*, 2018; McLain *et al.*, 2017). Urban gatherers usually select common wild plant species and plant parts that have little impact on the reproduction of plants (Schunko, Wild, & Brandner, 2021). Many gatherers have adopted the "principles of practice" and appropriate techniques for preventing or limiting negative ecological impacts; meanwhile, they teach and promulgate sustainable and responsible harvesting (Łuczaj *et al.*, 2021; Schunko *et al.*, 2021).

Despite these benefits, urban gathering is not extensive enough to be considered as a solution to multiple challenges within the food system (Nyman, 2019). With some exceptions (e.g., cities in the Pacific region (Borelli *et al.*, 2020)), the average contribution of wild algae, fungi and plants to diets is low (Shackleton *et al.*, 2017) due to lower tree density in urban spaces, the relatively low proportion of edible parts, or both (Bunge *et al.*, 2019; Estela, Ghermandi, & Margutti, 1995). There are also concerns and potentially physical health risks from eating wild plants or fungi grown on contaminated urban land (McLain *et al.*, 2012; A. Russo *et al.*, 2017), the spraying of chemical herbicides and pesticides (McLain *et al.*, 2014), and mistaking potentially toxic species with edible species (Fischer & Kowarik, 2020). For example, the wild edible food gathered along freeways and arterial roads often have concentrations of lead exceeding safety levels for human consumption (Amato-Lourenco *et al.*, 2020; von Hoffen & Säumel, 2014).

Tensions sometimes exist between urban gatherers and land managers, and between gatherers and other citizens over gathering, particularly in public spaces (McLain *et al.*, 2012). This varies by region. Gathering in many African cities, for example, is permissible in open urban areas, with tacit support from policy and land managers (Sardeshpande & Shackleton, 2020b). However, in many cities in Europe and North America urban gathering is not widely recognized or encouraged, although it is happening. Many cities have some form of regulations that prohibit or discourage urban foraging (Landor-Yamagata *et al.*, 2018; Ortez, 2021; Shackleton *et al.*, 2017).

Urban gathering is growing in popularity. Many scholars agree that more people would like to gather wild algae, fungi and plants (Fischer & Kowarik, 2020), but safety concerns, lack of knowledge, perceived social stigma, and lack of access remain significant barriers to urban gathering for many (Ortez, 2021; Somesh *et al.*, 2021). Conservation practitioners had a negative or ambivalent view about the desirability of allowing or encouraging more foraging, particularly in parks or natural areas (Wehi & Wehi, 2010). Risks to biodiversity seem manageable as overharvesting has not been documented (Landor-Yamagata *et al.*, 2018), and in fact many urban greenspaces conserve considerable biodiversity (Rupprecht *et al.*, 2015). Fruit gathering was likely to be least damaging (Sardeshpande & Shackleton, 2020b), and more abundant species are collected more frequently (Fischer & Kowarik, 2020). Even among those favoring gathering, sustainability assessment and adoption of appropriate rules was a precondition (Sardeshpande & Shackleton, 2020b).

Gathering may support invasive species management in urban ecosystems (Arrington, 2021; McLain *et al.*, 2017). Although most utilized species are native (Charnley *et al.*, 2018; Palliwoda *et al.*, 2017), a species' status as invasive or non-invasive can influence gathering practice (McLain *et al.*, 2017). Since many invasive wild plants have a history of cultivation as food, medicine, and materials, providing some socio-economic values, the gathering and use of edible weeds as a complementary resource has promising possibilities. For example, bracken fern *(Pteridium aquilinum)*, a native plant in the Pacific Northwest region of the United States of America, has been classified and gathered as an edible 'weed' Poe, LeCompte, McLain, & Hurley, 2014).

An emerging approach is to consider urban forests as nature-based solutions in the urban environment and include them in city management and planning (Roeland *et al.*, 2019). Trees are welcomed for their products and regulating services like shade and windbreaks, also their less tangible aesthetic and cultural values (Shackleton *et al.*, 2015). Urban gathering creates ties between people and the surrounding nature, in fact encouraging people to see urban vegetation and green space as natural (Landor-Yamagata *et al.*, 2018). Urban planners may consider these benefits of green spaces and issues of access to nature in the city (Charnley *et al.*, 2018; Shackleton, Drescher, & Schlesinger, 2020).

In summary, the combination of edible green infrastructure and urban beautification contributes to urban food production, as well as co-benefits nutrition, socioeconomics, and environment (Russo *et al.*, 2017). Ecosystem services provided by urban green space create urban gardening and gathering opportunities that contribute to healthy lifestyles (Jennings *et al.*, 2016). Traditional tropical home gardens serve as a model for biocultural diversity in small-scale urban green spaces (Hemmelgarn & Munsell, 2021; Sardeshpande & Shackleton, 2020a). The forest garden helps urban children develop environmental, scientific, and possibly other values (Askerlund & Almers, 2016). The use of edible green infrastructure areas and gardens are playing an important role in the COVID-19 pandemic and post-lockdown period as people have spent more time at home and demonstrated an increased awareness of the need for self- reliance and resilience to emerging threats (A. Russo & Cirella, 2020).

3.3.2.2.3 Gender trends

Gathering wild products is a gendered activity in many parts of the world, depending on cultural rules, on the type of harvested wild algae, fungi and plants and the places where they are harvested. In many countries, women perform the bulk of the labor for gathering and processing wild plants for food, medicine, fuel and handicrafts, as well as for other subsistence purposes, and often sell wild products at local markets (Howard, 2003). Some gathering activities are specific to men, some others are conducted equally by men or women, as well as children, or involve the whole family. Today, commercial gathering is done by men and women who make it their primary profession (Julliand *et al.*, 2019). In low-income households, women are often responsible for gathering for self-consumption and to sell (Sabater, 2020).

A range of examples show a variety of gender dynamics in gathering around the world. In the 1980s, farmers in the mountains of central France, men and women, harvested wild plants and mushrooms for their own consumption, to share with family and for commercial purposes. Children, teenagers and elders dedicated more time to gathering than adults, the latter being busy with agricultural activities (Larrère & La Soudière, 1985). In Turkey, gathering practices between men and women differ in that woman prefer to gather in social groups, and distribute some of the wild plants such as edible greens that they have gathered as gifts to friends and neighbors (Ertug, 2003). In the tropical forests of French Guiana, the Maroon Ndjuka women gather wild plants close to the village and the fields, while men gather wild plants in the deep forest (Tareau *et al.*, 2019). In the savannas of central Brazil, the Xavante women gather wild plants while the men hunt, but men sometimes join them (Flowers, 2014). Extractivism with long expeditions in the forest is usually practiced by men, for instance rubber or piassava collectors in the Amazon (Schmink & García, 2015), eaglewood (*Aquilaria* sp.) collectors in Borneo or Papua; in these last regions, some traders even organize expeditions where they drop a group of several men in the middle of the forest from a helicopter (Mittelman, Lai, Byron, Michon, & Katz, 1997). In the dry and semi-dry areas of Africa, gums and resins such as gum arabic *(Acacia senegal, A. seyal)*, myrrh *(Commiphora myrrha)* and frankincense (*Boswellia* spp.) are usually gathered by men, pastoralists who fulfil this activity while taking their cattle to graze (Mugah, Chikamai, Mbiru, & Casadei, 1997). Tapping resins in general is a male task, especially when it is necessary to climb on trees. Batak benzoin tappers in North Sumatra, Indonesia, describe the benzoin tree *(Styrax paralleloneurum)* as a woman who gets pregnant of the resin after the tapping, a symbolic sexual act (Esther Katz, García, & Goloubinoff, 2002).

3.3.2.3 Uses of wild plants, algae, and fungi, including the leaves and fruits of trees

Unlike the case for some of the other practices, where only selected uses are relevant, all of the uses outlined in Chapter 1 of the assessment are relevant for gathering practices. In fact, in several subsections of 3.3.2.3 the diversity of species gathered for the various uses are so extensive that additional subdivisions have been created. The sections are as follows: Ceremony and cultural expression (3.3.2.3.1); decorative and aesthetic (3.3.2.3.2) with subsections on ornamental, natural cloth and dyes, handicrafts, and perfume and incense; energy (3.3.2.3.3); food (3.3.2.3.4) with subsections on nuts & seeds, starchy fruits, juicy fruits, beverages, syrups, gums, and resins, wild edible mushrooms, and wild vegetables; medicine and hygiene (3.3.2.3.5); recreation (3.3.2.3.6); science and education (3.3.2.3.7); and materials and shelter (3.3.2.3.8). Importantly, the text is not an inventory of all species gathered for various practices. Rather, the focus is on those species of particular interest in relation to sustainable use which emerged through the systematic literature review and those that were highlighted through various rounds of expert discussion and review.

3.3.2.3.1 Ceremony and cultural expression

The world's major cultures and ritual practices observe conservation of species and nature as essentials for human well-being. Cultural expression may take the form of song, stories, dances, art, designs, crafts, rituals, ceremonies, and more. Many wild species, especially wild plants and fungi, perform critical roles in ceremonies of various cultures around the world. They are harvested for use in spiritual observances and practices, and are highly valued for their role in maintaining cultural identity in formal ways (Hamilton, 2004). The Millennium Ecosystem Assessment highlighted that impeding religious and social ceremonies by denying people access to required wild plants or fungi could harm social relations as "many cultures attach spiritual and religious values to ecosystems or their components" (Millennium Ecosystem Assessment, 2005).

Research on gathering for ceremony and cultural expression focus more on the cultural dimensions, such as the types of rituals (e.g., marriage, birth, death, important memorial points and specific religious rituals) than they do on the sustainable use of the species *per se*. Wild species are sometimes mixed with horticultural plants, and used for decoration, smoking, dyeing, as non-pharmacological medicine or for energy and nutrition. Flowers and incenses made out of dried plants or resins such as frankincense or myrrh are often used in rituals. It is difficult to make a complete list of species used for ceremonies, as many ethnobotanists make inventories of dozens to hundreds of wild plants from local surveys (Barceló, Butí, Gras, Orriols, & Vallès, 2019; Des, Rizki, & Fitri, 2019; Yanfei Geng *et al.*, 2017; Rangel-Landa, Casas, García-Frapolli, & Lira, 2017).

Some of the wild species used for rituals are unusual and rare (Naegel, 2004; Rangel-Landa *et al.*, 2017). Gatherers give them as presents to the organizers of the ceremony or communitarian feast, and commercialization is uncommon (Barceló *et al.*, 2019; Rangel-Landa *et al.*, 2017). However, because of important traditional culture, there is often concern about the disappearance of these particular species in studies of national culture. Rare species are harvested at levels just enough to satisfy the needs of the community (Rangel-Landa *et al.*, 2017), and in some cases substitutions are developed (Des *et al.*, 2019). The ritual practices of Naxi people in Yunnan, China for example, pay high respect to conserving natural resources, although these beliefs and cultural expressions receive less attention from younger generations (Yanfei Geng *et al.*, 2017).

Hallucinogenic plants and fungi harvested in the wild are used by shamans or mediums, in religious or curing ceremonies, in particular on the American continent, but also in Siberia *(Amanita muscaria)* and Africa (iboga, *Tabernante iboga*, in Gabon). For the Jotï, an indigenous group in the Venezuelan Amazon, mushrooms play a central role in their religious and spiritual beliefs and are fundamental to their cosmology (Zent, 2008). Among the most known in the Americas are ayahuasca (*Banisteropsis* sp.) from the Amazon, and *Psylocybe* mushrooms and peyote *(Lophophora williamsii)* from Mexico and the United States of America Southwest (Furst, 1972; Heim & Wasson, 1958; Schultes & Hofmann, 1979). In the 1960s, after Wasson's discovery of hallucinogenic *Psylocybe* mushrooms in Mexico, "hippies" rushed to that country to experience these fungi. There has also been a development of shamanistic tourism among the Mazatecs in Mexico (Demanget, 2010) and in the Peruvian Amazon (Fotiou, 2016). Peyote is still traded in the United States of America (Feeney, 2017). Overall, 216 species of fungi are thought to be hallucinogenic, and of these 116 species belong to the Genus *Psilocybe* (Willis, 2018).

Since rituals are not a daily need, there are few relevant management measures that are directly applied to species specifically relating to ceremonial use, and it is recommended that maintaining traditional and cultural practices can complement management strategies (Kideghesho, 2009). In fact, conserving biodiversity based on cultural and religious faiths may be often more efficient and sustainable than government legislation or regulations given peoples' long-term relationships with the particular species.

3.3.2.3.2 Decorative and aesthetic

Wild species are harvested for crafts and decorative use for personal consumption, as gifts, and for sale as raw or value-added items (M. R. Emery, 1999). The gathering of wild species like orchids, Bromeliads, succulents, and wild fungi are important sources of money and livelihood for collectors at local and regional scales and may also enter into global trade. Hence the sustainability of their wild populations, habitat, economies and communities is a subject of concern. Many wild species harvested for crafts are usually listed in inventories as parts of general ethnobotanical research. It can be challenging to distinguish among uses at the local level, as one collection may result in the gathering one species for food, medicine, ritual decoration, and transplant into the home garden. There is a lack of research on the sustainability of this kind of mixed use.

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When the acquisition is part of the organism and managed well, gathering wild plants for the use of decoration may not have too many negative effects. For example, although there is lack of conservation assessment of the 80% of wild harvested Indian plants to make potpourri, the gathering of such 455 species provides a supplementary income to rural poor and is considered as a sustainable use (Cook, Leon, & Nesbitt, 2015). In Minas Gerais, central Brazil, the gatherers of everlasting *(sempre-vivas)* flowers of the Serra do Espinhaço Meridional enrich the native pastures where the flowers grow with the seeds fallen from the collected flowers and stimulate their growth by fire management (Monteiro *et al.*, 2019), demonstrating a form of traditional management and care which supports sustainable use. Their agro-extrative system was recognized by FAO in 2020 as a Globally Important Agricultural Heritage System (GIAHS) (GIAHS, 2020). Trade in exotic wild plants increased in North America and Europe after the Second World War and demand for wild plants increased pressure on wild populations and even drove the extinction of some rare species in the late 1970s (Lavorgna, Rutherford, Vaglica, Smith, & Sajeva, 2018). Twenty-two European countries reported the total value of "ornamental plants" at almost 1,400 million euros, which amounts to 49.6% and the highest of the marketed plant products from forests (Forest Europe, 2020). The Convention on International Trade in Endangered Species of Wild Fauna and Flora has listed more than 32,000 species of ornamental plants in its Appendices, most in Appendix II **(Table 3.8)**.

The gathering for sale of cut flower or foliage of bromeliads, or ornamental plants like aloe and orchids are considered to negatively affect species survival (Flores-Palacios, Bustamante-Molina, Corona-López, & Valencia-Díaz, 2015; Mondragón Chaparro & Ticktin, 2011; Mondragón, Méndez-García, & Morillo, 2016; Negrelle & Anacleto, 2012; Phelps & Webb, 2015; Sakai *et al.*, 2016). Many of these species are also cultivated, but no data was available at the time of this assessment on the share of global market sales from wild *versus* cultivated plants. Sale prices vary between species (Mondragón *et al.*, 2016), but the origin of the plants (wild *vs* farmed) did not affect price, since cultured plants have better physical variables than wild-harvested plants (Elps, Carrasco, & Webb, 2014). Some researchers believe that the supply-side measures to ensure the sustainable use may lack effectiveness. Consumer preferences may help to reduce the market driven push to overharvest (Elps *et al.*, 2014).

More than a half of all cactus species (57%) are used by people. Cacti are prized for their aesthetic qualities. The most common use is for ornamental horticulture (674 species), which in most cases is related to gathering wild plants and seeds for specialized collections. Cacti comprise about 130 genera and 1,500 species distributed mainly in North and South America; however, several species of *Rhipsalis* (mistletoe cactus) occur in tropical Africa. Some species of *Opuntia* (prickly pear) have been introduced in Africa, Australia and South Asia (India). Nearly all genera are cultivated as ornamentals; some of the more common are *Opuntia* and *Carnegiea* (giant saguaro), *Cereus* (hedge cactus, cereus), *Echinopsis* (sea-urchin cactus), *Epiphyllum* (orchid cactus), *Hylocereus* (night-blooming cereus), *Mammilaria* (pincushion cactus), *Melocactus* (Turk's cap cactus), *Rhipsalis*, and *Schlumbergera* (Christmas

Table 3 8 **Ornamental wild plants listed under the Convention on International Trade in Endangered Species of Wild Fauna and Flora.**

Source: Species+ data (UNEP, 2021) (The Species+ Website. Nairobi, Kenya. Compiled by UNEP-WCMC, Cambridge, UK. Available at: www.speciesplus.net. [Accessed 01/March/2021])

Common name	Family/Genus	Appendix	Number of listed species in the taxa
Agaves	*Agavaceae*	I and II	4
Snowdrops	*Galanthus* spp. and *Sternbergia* spp.	II	21 + 9
Cashews	*Operculicarya* spp.	II	3
Elephant trunks	*Pachypodium* spp.	I and II	23
Ponytail palms	*Beaucarnea* spp.	II	11
Bromelias	*Tillandsia*	II	3
Cacti	*Cactaceae*	I and II	1532
Zygosicyos	*Zygosicyos*	II	2
Tree-ferns	*Cyathea* spp. and *Dicksonia* spp.	II	686 + 46
Cycads	*Cycadaceae* spp. and *Zamiaceae* spp.	I and II	109 + 228
Alluaudias	*Didiereaceae* spp.	II	12
Elephant's foot,	*Dioscorea deltoidea*	II	1
Venus' flytrap	*Dionaea muscipula*	II	1
Succulent spurges	*Euphorbia* spp.	I and II	709
Ocotillos	*Fouquieria*	I and II	3
Aloes	*Aloe* spp.	I and II	483
Pitcher-plants	*Nepenthes* spp. and *Sarracenia* spp.	I and II	112 + 29
Orchids	*Orchidaceae* spp.	I and II	27,924
Palms	*Palmae*	I and II	13
Poppy	*Meconopsis regia*	III	1
Passion-flowers	*Adenia sp.*	II	3
Sesames	*Uncarina*	II	2
Lewisias, portulacas, and purslanes	*Anacampseros* spp., *Avonia* spp. and *Lewisia serrata*	II	25 + 11 + 1
Cyclamens	*Cyclamen* spp.	II	27
Stangerias	*Stangeria eriopus* and *Bowenia* spp.	I and II	3
Grapes	*Cyphostemma* spp.	II	3

cactus) (Judd, 1999). Native people of the Americas propagate branches, seeds or transplant complete individuals from the wild to their agroforestry systems and home gardens (Casas & Barbera, 2002). People occasionally harvest useful parts of several species of cacti for use in traditional medicine. Cactus pears from *Opuntia stricta* are also considered as a potential source of natural colorants (Casas & Barbera, 2002; Goettsch *et al.*, 2015).

Due to their popularity and the commercialization of so many wild species, poaching entire plants from the wild is a growing problem. Most species are regulated by the Convention on International Trade in Endangered Species of Wild Fauna and Flora **(Table 3.8)**. Among the threatened cacti species, 64% are utilized by humans in some form and 57% (236 species) are used in horticulture (Goettsch *et al.*, 2015). There is growing concern that a high proportion of cactus species may be threatened with extinction in the near future, mainly due to growing illegal trade.

Orchids are a prominent group of the global horticultural trade. While large numbers of orchids are grown commercially, there are still large numbers taken directly from the wild. Over-harvesting of wild orchids associated with floral and medicinal trade is a serious concern for their long-term survival (Hinsley *et al.*, 2018). Cross-border trade of orchids is well recognized as a threat to orchid conservation and regulated by the Convention on International Trade in Endangered Species of Wild Fauna and Flora. However, domestic trade may not be regulated or poorly enforced in some orchid-rich countries (Phelps & Webb, 2015; Tamara Ticktin *et al.*, 2020; Wong & Liu, 2019). This legal and illegal domestic trade of wild orchids can be larger than cross-border trade and can also pose serious threats to species survival, but receive far less attention from orchid conservationists (Phelps & Webb, 2015; Tamara Ticktin *et al.*, 2020; Wong & Liu, 2019).

Snowdrops (*Galanthus* sp.) is a relatively small genus of perennial herbaceous plants distributed throughout Europe and central Asia, threatened in the wild due to habitat destruction, illegal gathering and climate change. A cherished garden plant with beautiful flowers blooming in winter and early spring, *Galanthus* is the world's most traded wild-sourced ornamental bulb genus. To implement the Convention on International Trade in Endangered Species of Wild Fauna and Flora regulations, Turkey sets annual export quotas of wild bulbs at 2.5-5.0 million for *G. elwesii* and 2-4 million *G. woronowii*. Georgia sets an export quota of wild *G. woronowii* at 15 million a year to ensure the trade and gathering do not endanger the survival of wild populations (Rønsted, Zubov, Bruun-Lund, & Davis, 2013; UNEP, 2021, p. 2021).

Natural cloth and dye

Numerous wild plants, lichens, and mushrooms have been used as natural dyes for centuries. Some of them, such as Brazil wood *(Caesalpinia echinata)*, were traded across continents. Most natural dyes were substituted by chemical dyes from the 19th century on, but some remained in use in local arts and crafts, and have been revived recently. Some species are not only on textiles but also in the cosmetic and food industries. For instance, the lichen *Rocella canariensis* is used as a food coloring known as E121 (Cardon, 2007).

Cotton, linen, silk, wool and artificial fiber and dyes have replaced many wild sources. Uganda bark cloth was derived from the wild fig or mutuba tree *(Ficus natalensis)* and has been recognized by UNESCO as a masterpiece of the 'Intangible Cultural Heritage of Humanity'. The production process requires collaboration among local laborers, specific skills and specially designed tools. In recent years bark cloth has been explored as a sustainable fashion luxury textile, providing jobs to local communities (Venkatraman, Scott, & Liauw, 2020). The use of bark facilitates scattered planting of mutuba trees in agroforestry systems, which in turn protects crops and soil from erosion on windy hill slopes. World Overview on Conservation Approaches and Technologies (WOCAT) has developed a guide on the use and propagation of the tree. This example highlights the value of this specialized knowledge. However, traditional knowledge on unique dying sources and processes is vanishing fast, and represents a knowledge gap which may become impossible to address in the near future.

Many wild fungi and lichens are also harvested for use in dye making. For example, Emery, Martin and Dyke (2006) found that of the over 200 species harvested from the wild in Scotland, 76 of them were non-vascular species. Of these, 16 were harvested for crafting purposes such as the production of dyes for homespun wool. A group of lichens known collectively as 'orchil' has been used as a dyestuff since the Bronze Age in Europe. Trade in orchil declined as manufactured, synthetic and cheaper alternatives were found. It continues at low levels for artisanal use (Wolfslehner *et al.*, 2019). Some firms specialized in plant dyes aim at meeting standards of environmentally and socially responsible manufacturing and have applied to a certification, but as of 2010 this issue remained unresolved (Cardon, 2010).

Handicrafts

The following is not meant to be an exhaustive inventory of all wild algae, fungi and plants used for handicrafts. Rather, it is a review of the wild species of interest with regards to sustainable use which appeared in the systematic literature searches.

A wild plant material called golden grass *(Syngonanthus nitens)* is used to produce golden handicraft articles in Brazil. Rural communities harvest, process and knit the scapes of *Syngonanthus nitens*, which has been an important source of income for them since the late 1990s. The survival of plant populations was once affected by the increase in community demand for scapes. The Brazilian federal environmental agency (Ibama) has proposed management techniques to prevent overexploitation of the species. For example, the harvest time was set precisely to ensure the removal of inflorescences after seed production or full maturation. Furthermore, returning the capitula of inflorescences used in handicraft to the field represents another important tool for the sustainable management of golden grass (Oliveira, Cruz, Sousa, Moreira, & Tanaka, 2014; I. B. Schmidt, Figueiredo, & Scariot, 2007; I. B. Schmidt & Ticktin, 2012).

There are several types of wild plants in the United States of America called Sweetgrass, that can be used to make handcrafts. *Hierochloe odorata* is native to Northern North America and is commonly used as incense and fragrance by Native Americans. It is used traditionally to craft or decorate baskets and bowls (Leif, 2010). In South Carolina, gulfhairawn muhly *(Muhlenbergia filipes)* is also called Sweetgrass. Its leaves are gathered by the Gullah community, descendants of enslaved Africans, to make a form of coiled basketry. The Gullah basket is now recognized as an artform and a major source of income for the local people (USDA & NRCS, 2009). This native coastal grass on which the basket makers depend has become increasingly scarce due to urbanization and limited access to the resource. Basket makers have to develop social-economical strategies, such as purchasing raw materials from other states, or negotiating access to the grass to maintain the traditional artform and their livelihood (Grabbatin, Hurley, & Halfacre, 2011; Hurley *et al.*, 2013; USDA & NRCS, 2009).

Many species of wild fungi are harvested for craft purposes. Turkey tail mushrooms *(Trametes versicolor)* grow throughout North American forests, and also across Europe and Asia. Turkey tail is a very colorful bracket fungus that grows throughout the year on dead or rotting wood. Pieces of the fruiting body are often harvested for use by artists and jewelry makers, who most commonly use them in earrings and necklaces (Spahr, 2009). *Ganoderma applanatum* (commonly known as the artist's conk) is also a bracket fungus with a cosmopolitan distribution. It is sometimes used as a medicinal tea, but it is most commonly known in North America for its use as an artist's canvas of sorts, where burning or carving into the underflesh of a dried polypore leaves behind brown markings to create images (Wetzel, Duchesne, & Laporte, 2006). In this case, while some mycelia live on in the dying or decaying wood medium, polypores take so long to grow that when the fruiting body is harvested, functionally almost the entire the organism is harvested.

The long-term sustainability of wild mushroom, wild fungi and wild lichen gathering varies depending on several factors. First, how much of the organism is harvested is paramount. In most cases, it is actually only the fruiting body that is taken, leaving the mycelium behind in its substrate. However, if the fruiting body is harvested before the spores are released the reproductive potential is essentially removed. Despite variation across species and regions in what is harvested and how, there is general agreement that most fungi harvested for crafts purposes are harvested at sustainable levels.

Bark is a popular handicraft item. Otomi people in Mexico use barks of *Trema micrantha* and several Ficus species for handmaking paper crafts. With the color paintings by the Nahua people, Amate bark paper has been traded nationally and internationally. Bark harvesters include indigenous and non-indigenous peoples, often of low-income. From the 1980s to 1995, the bark supply increased dramatically from only 4 main harvesters to around 200 people in an area of 1500 km^2. As the main source and preferred species of bark paper, Trema micrantha are fast growing, occur within all vegetation and can be harvested throughout the year. The species is recommended for amelioration of degraded lands. It is planted as a shade tree in coffee plantations. When it reaches five to eight years of age it is removed as part of the management of the coffee plantation. With the expansion of the harvest area, including the above factors, this use of bark to make Amate handicrafts is considered to be growing and sustainable (López, 2005). Birch bark is also harvested throughout central North America and northern Europe and used for a variety of handicrafts including baskets and ornaments. According to Emery *et al.* (2014), "Paper birch *(Betula papyrifera)* is a cultural keystone species for the Anishinaabe in the United States of America Great Lakes region" specifically because of its bark.

Perfume and incense

Aromatic plants often have medicinal values and face the same stress and sustainability problems as medicinal plants. Numerous resins are used as incense around the world, either for local use and small-scale trade or for international trade, such as frankincense (*Boswellia* spp.) or myrrh (*Commiphora* spp.). Frankincense and myrrh products also have wide ranges of other industrial uses such as for food and beverages, and are used as traditional medicines in China. The first two quality grades of final products are sold in international markets and the least quality graded items are for domestic use like in churches, coffee ceremonies, etc. Tapping and gathering of frankincense is carried out around the dry season. It follows a specific pattern including shaving a thin layer of the bark, the moderate widening of the wound one month later, and then the gathering the gum. An average of 500 g of frankincense is obtained from each tree each season after three to four months of continued tapping (W. Tadesse, Desalegn, & Alia, 2007).

Total world export demand is estimated at around 2500-10,000 tons/year with much uncertainty, since the European Union and the United States of America have a broader classification of natural gums and resins in the harmonized system code. The principal exporters are Ethiopia, Kenya, Somalia and Eritrea (Coppen, 2020b; Wubalem Tadesse, Dejene, Zeleke, & Desalegn, 2020). *Boswellia papyrifera* which is the main source (70% of the Ethiopia's natural gum and resins production) is declining at alarming rates, due to expansion of agricultural lands, overgrazing, population increase, growing demand for construction and fuel wood, forest fires, and pests and diseases. Recent increases in demand of frankincense have also led overharvesting. The lack of traceability in the supply chain and the ineffectiveness of organic certification also affects populations of substitute frankincense species. Studies suggested cultivation and substitution to mitigate the impact and sustain this historical activity (Brendler, Brinckmann, & Schippmann, 2018; S. Johnson *et al.*, 2019; Wubalem Tadesse *et al.*, 2020).

The Spikenard, also called *Jatamansi*, is made from the rhizomes of *Nardostachys jatamansi* distributed in the Qinhai-tibet Plateau and Himalayas in Asia. It is vulnerable to harvesting and on the verge of extinction due to overexploitation and habitat destruction in some areas. It was evaluated as critically endangered in India but is common in Himalayas of China and Nepal. Sustainability of harvest is related to the harvesting practices. The sensitivity is higher in outcrop than in meadow habitats. Positive effects are possible with low harvesting levels under strict management conditions (Ghimire, Gimenez, Pradel, McKey, & Aumeeruddy-Thomas, 2008; Ghimire *et al.*, 2005; Kamini & Raina, 2013; Larsen, 2005).

3.3.2.3.3 Energy

As renewable sources of bioenergy, wild plants and fungi have a huge contribution to make to reducing both carbon emissions and energy poverty. Many African countries have high proportions of fuel species. In East Africa, the indigenous tree species *Croton megalocarpus* supports a sustainable seed oil industry that provides biofuel for electricity. One microenterprise, EcoFuels Kenya, sources more than 3,000 tonnes of wild-harvested nuts each year. Fungi, in particular, have much unexplored potential within the bioenergy sector. Microbial fuel cells can be run on fungal enzymes, such as those from baker's yeast *(Saccharomyces cerevisiae)*, to generate electricity from plant biomass (Antonelli *et al.*, 2020).

Switchgrass *(Panicum virgatum)* is native to North and Central America, can grow in many different soils, has low fertilizer requirements and can in some cases promote biodiversity depending on the land use being displaced (Cheng & Timilsina, 2011). It can be used as a biofuel source and has potential economic benefits especially in the United States of America. Despite this potential, the environmental consequences of converting to crop grasslands and large land use needs must be addressed (Barney & DiTomaso, 2010; R. A. Brown, Rosenberg, Hays, Easterling, & Mearns, 2000). Switchgrass has been shown to have the potential to decrease soil erosion rates 30 times during the first year of growth, and up to 600 times during the second and third years when the root system has been established (McLaughlin *et al.*, 2002; Williams, Inman, Aden, & Heath, 2009). Werling *et al.* (2014) found that perennial grasslands that contained switchgrass and prairie plantings have significantly higher biodiversity than maize lands, as arthropods, grassland birds, soil-living methanotrophic bacteria and pollination-insects were found, among others.

Two other interesting wild plant species are *Miscanthus* spp., which is native to Southeast Asia, and Bermudagrass *(Cynodon dactylon)*, native along the United States of America coast. All three grass species are very interesting as biofuel plants, as they grow in the wild but can also be cultivated (Cheng & Timilsina, 2011). The grass genus *Miscanthus* is among the first crops for which bilateral agreements have been developed under the Convention on Biological Diversity to guide breeding of new varieties from wild germplasm collections from Asia (Antonelli *et al.*, 2020; Grace *et al.*, 2020). Certain natural grasslands are found in some climate zones and it may be beneficial for future biofuel production to come from grassland as the root system in the soil can prevent erosion.

Jatropha is a group of non-edible plants found mostly in America that includes 66 species (Dehgan, 1984; Goel, Makkar, Francis, & Becker, 2007). The most common species, *Jatropha curcas*, is a multipurpose plant species useful to control soil erosion, improve soil infiltration, reclaim wasteland and phytoremediation of contaminated soil, and prepare green manure (Subedi, Chaudhary, Kunwar, Bussmann, & Oaniagua-Zambrana, 2021). The species has a high core nonvolatile oil content, between 25 and 35% (Díaz *et al.*, 2017; R. S. Kumar, Parthiban, Hemalatha, Kalaiselvi, & Rao, 2009), and is the most domesticated species of *Jatropha* used today. It was created through a combination of systematic selection, inter-hybridization (between *J. curcas* and *J. integerrima*) and breeding programs and has a higher oil content (Sujatha & Prabakaran, 2003), but *Jatropha* is still a wild plant grown as live fence around agricultural fields (Becker & Makkar, 2008; R. S. Kumar *et al.*, 2009) and is regularly used by indigenous people Subedi *et al.*, 2021). The other plant with oil content—*Croton megalocarpus*— is native to eastern Africa and can have a seed oil content of 30-45% on a mass basis (Aliyu, Agnew, & Douglas, 2010; Hines & Eckman, 1993).

Another interesting wild plant rich with oil is the Beauty Leaf Tree *(Calophyllum inophyllum),* which can carry 10,000 fruits per tree a year and the seeds contain up to 60-70% useful oil (Friday & Okano, 2006; Jahirul *et al.*, 2013). The tree is native to Australia but has been introduced to Southeast Asia and India and started to use as biofuel plant at small-scale (Friday & Okano, 2006). Brock *et al.* (2018) noted the gold-of-pleasure *(Camelina sativa)*, which is an old-world oilseed crop that went out of use in the mid-20th century but has now gained renewed interest as a biofuel source.

There are various studies about wild-living plants and crops and even Yang *et al.* (2013) have studied possible wild plants for biofuel production and to avoid competition of using of edible plants for food industry. They studied wild plants from salt-alkali wastelands, which often occur in many arid and semi-arid regions of the world. They note that "[…] the direct competition with food production should be avoided and a much wider range of plants possible sources of biomass should be made or screened so that they are able to be grown on marginal lands. The non-edible biofuel plant species with fewer inputs, higher tolerant are required so that the diesel plants can be planted in the desert or on the saline-alkali land." They listed several wild herbaceous plants rich in oil from stems and leaves in China: *Euphorbia heyneana* (15.01%), *Ricinus communis* (13.9%), *Cirsium setosum* (12.5%), *Euphorbia nutans* (11.02%), *Cirsiu japonicum* (9.27%), *Metaplexis japonica* (8.27%), *Taraxacum officnala* (7.75%), *Lactuca raddeana* (7.63%), *Euphorbia humifusa* (6.88%), *Euphorbia thymifolia* (6.81%), *Euphorbia* *esula* (6.57%) and *Aster tataricus* (5.64%). It is possible to develop a method to extract biofuel from these herbaceous plants and at the same time use the semi-alkali wasteland as possible cultivation land and avoid competition with crops for food production.

3.3.2.3.4 Food and beverage

Food consumption is the most common form of use for gathering wild species. Foraging is the oldest productive activity of people, but it keeps being practiced, in rural as in urban environments (Svizzero, 2016). Information on wild species used for food historically came from ethnobiological/ethnobotanical inventories. It is more recently increasing in the scientific literature due to renewed interest in gathering and sustainable use. The most important sources of human food are almost all vascular plants (flowering plants, conifers and other gymnosperms, ferns, horsetails and clubmosses), accounting for 7,014 species of the 7,039 included in the reviews cited. The remainder are bryophytes (mosses, liverworts and hornworts), and green and red algae (Antonelli *et al.*, 2020; Ulian *et al.*, 2020). In agricultural and forager communities in Asian and African countries, the mean use of wild foods is 90-100 species per location, and in indigenous communities there are an estimated 120 wild species used as food in communities in both industrialized and developing countries (Bharucha & Pretty, 2010).

With economic and social development, the acquisition of wild food through gathering has been gradually marginalized. In some places, the harvest and consumption of wild foods is considered antiquated behavior and may even be denigrated and abandoned (Garcia, 2006; Łuczaj *et al.*, 2012). For example, islands of Western Oceania are particularly rich in native fruit and nut trees; in Vanuatu, out of 40 of these native species, 30 are not cultivated; they used to play an important part in local diets but presently are often substituted by industrial food (Walter & Sam, 1999). In places where gathering persists, it has been suggested that some people consider it an optimal alternative to farming. This may include trading foraged goods with farmers. This is recognized to be the case in places where gatherers refrain from practicing agriculture for cultural, social, or institutional reasons. (C. Tisdell & Svizzero, 2015). Nevertheless, it is now valued again in some countries as health food and in haute cuisine (Łuczaj *et al.*, 2012; Doyon, 2019). There is also a growing demand for wild plants in the food and aromatics industry (Lescure *et al.*, 2015).

In addition to being a food source, evidence shows that for some indigenous peoples and local communities, during times of food shortage wild foods provide nutritional supplements of important vitamins and minerals (Harris & Mohammed, 2003). This finding extends to urban dwellers in developed countries. Gathering wild foods sustains dietary traditions and supports community livelihoods. Trends in consumption of wild foods in Europe vary according to regions and countries, and according to categories of species. One study found that across European Union countries at least 27 species of mushrooms and 81 species of vascular plants are harvested and consumed as wild food (Schulp, Thuiller, & Verburg, 2014). Gathering for food is not a static process; some wild plants are consistently gathered, others are forgotten or re-emerge after periods of unpopularity (Łuczaj *et al.*, 2012).

Scientific studies have focused on the analysis of dietary conditions related to human health, such as the nutritional content of wild species, toxic side effects, heavy metal concentration (mainly fungi) and health risk assessment. Recorded indigenous and local knowledge combined with scientific analysis, is promoting new resources for crop development, the protection of crop wild relatives, and the provision of new solutions or ideas to address global hunger and protein sources. Because the number of wild plants (and fungi) gathered for food is so extensive, we have further divided this section into sub-sections.

Wild fruits are important source of nutrition, medicine, materials for cosmetics, crafts, fiber, and fuel and are the most widely used wild algae, fungi and plants. Clement (2006) distinguishes three types of fruits: (i) nuts and seeds, which contain oil and are rich in proteins and so can play an important part in the diet, (ii) starchy fruits rich in oil and starch (such as palm fruits) and (iii) juicy fruits, such as berries, rich in vitamins. In the United States of America alone, permitted harvest volumes of edible fruits, nuts, and berries were as follows: 303, 748 gallons and 670,726 pounds (Chamberlain, Emery, & Patel-Weynand, 2018). While these figures represent the best available data, they likely do not represent total harvest of popular species black walnut (*Juglans negra* L.), pine nuts, and low-bush blueberries.

A recent literature review on wild edible fruits found that studies have increased over the last three decades, a majority of it reports ethnobotanical and taxonomic descriptions with relatively few studies on their landscape ecology, economics, and conservation. Among them, a third of retrieved articles were based on studies in Africa and a quarter were from South America. (Sardeshpande & Shackleton, 2019).

Different fruit species respond differently to harvesting and other disturbances, such as fire and herbivory. Although the review by Stanley *et al.* (2012) concludes that the majority of case studies surmise that wild algae, fungi and plants harvests are ecologically sustainable, Sardeshpande and Shackleton (2019) found 14 of the 25 studies explicitly addressing harvest sustainability illustrated overexploitation beyond recovery to optimal vitality. In some cases, extraction in a commercial scale is the attempt to make benefits to avoid tree logging and deforestation, such as the marula *(Sclerocarya birrea)* fruit, the Brazil nut (*Bertholletia excelsa*, Lecythidaceae) and the bush mango (*Irvingia gabonensis* Baill. ex Lanen.). When harvest is lethal to the plant or market demand is high which drives to intensive production, the species of wild edible fruits is domesticated and cultivated as tree crops. Certification is considered to ensure the sustainability of gathering under the influence of the trade chain and to promote socio-economic conditions for harvesters and forest communities (Sardeshpande & Shackleton, 2019). The following examples collate several species of wild edible fruits are a complement to the aforementioned review that are mainly gathered from the wild and also support a certain scale of trade in a medium term.

Nuts and Seeds

In the United States of America, pine nuts (*Pinus monophyla* Torr. & Frém.) are highly prized nuts harvested primarily from natural stands on public lands in the western half of the country. These forests are usually not actively managed for pine nut production and in fact are a forest complex (pinyon-juniper) that was historically seen as without much value and eradicated in favor of range lands. However, pine nuts have a long history of use among indigenous peoples and local communities in the southwestern United States of America, where pine nuts continue to be harvested for local markets and for export. While the United States of America exported approximately 20,000 United States Dollars worth of pine nuts in 2007, it imported about 54 million United States Dollars worth (Chamberlain, Emery, & Patel-Weynand, 2018), suggesting that the majority of pine nuts harvested are for personal and local use. Also in the United States of America, approximately 25 million pounds of black walnut (*Juglans nigra* L.) were harvested from natural populations in 1998, although it is unknown if this constitutes a sustainable harvest amount (Chamberlain, Emery, & Patel-Weynand, 2018).

The Brazil nut tree is an iconic tree occurring in *terra firme* (non-flooded) forests throughout the Amazon basin. It can reach up to 50 meters tall and live for hundreds of years. Brazil nut seed harvesting from natural forests is a cornerstone algae, fungi and plants economies in Amazonia. Brazil nuts are the only globally traded seed gathered from the wild by tens of thousands of rural households and are an integral component of the extractivist culture of many indigenous peoples and local communities in the area. In the Brazilian Amazon alone, over 45,000 tons of Brazil nuts are gathered annually, with sales of over 33 million United States dollars (Guariguata, Cronkleton, Duchelle, & Zuidema, 2017; Peres *et al.*, 2003; Wadt, Kainer, Staudhammer, & Serrano, 2008).

Brazil nut is organized in a concession system and the supply chain includes three certification schemes: (i) organic certification; (ii) Fairtrade certification; and (iii) Forest Stewardship Council certification. It is considered a model of the use of wild species for promoting "conservation-through-use". Extensive research suggests the Brazil nut tree reacts robustly to the type and level of extraction currently practiced in the medium term (Guariguata *et al.*, 2017).

Given the importance of this species to the local economy, this assessment highlights some specific concerns that may affect future status and trends of Brazil nut production. Without active management, in the past extensive and intensive exploitation led to insufficient juvenile recruitment to maintain populations, and harvested populations went into a process of senescence and demographic collapse. Rainfall is also a key factor in determining tree performance and demography and the forecasted declines in pollinator diversity may threaten the long-term resilience of the Brazil nut trees. Climate change therefore could potentially negatively impact *B. excelsa* populations (Peres *et al.*, 2003; Thomas *et al.*, 2017). Changes in human use of the forested landscape are also an immediate concern. Brazil nut extraction is accompanied by unsustainable forestry activities outside the gathering seasons in a given year. Due to development pressures, Brazil nut forests have been gradually destroyed and transformed into market-oriented agricultural areas to support global beef markets. Land conversion in the basin has also sparked violent conflicts and led to decreased sustainable management of Brazil nut producing areas. Some of these challenges are being addressed in Brazil, Bolivia and Peru (Bertwell, Kainer, Cropper Jr, Staudhammer, & de Oliveira Wadt, 2018; Escobal & Aldana, 2003; C. S. Simmons *et al.*, 2019; Wadt *et al.*, 2008).

Starchy Fruits

At least 30 Amazonian palm species are used as food, most of them for their fruits (*Attalea* spp., *Euterpe* spp., *Mauritia flexuosa, Oenocarpus* ssp.), consumed raw, cooked or processed into drinks (Kahn, 1997). *Oenocarpus bataua* is the seventh most abundant tree in the Amazon and one of the most used palms in neotropical forests in the Americas. Once, felling adult palms was the most common technique used to harvest fruits, which negatively affected the demography of its population. Inconsistent regulations on *O. bataua* harvesting across different countries contributes to confusion and threatens sustainable use of this species. Colombia has a harvest quota; Ecuador requires management plans; Peru and Bolivia forbid killing the tree. However, in all cases enforcement is difficult. To support sustainable use, in some villages, adult palms are climbed when they are not too tall to cut racemes with ripe fruits, and such non-destructive harvest techniques may meet the increasing demand and maintain the populations.

Pequi *(Caryocar brasiliense)* is a native fruit from Brazil, found in the Amazon, Caatinga, Cerrado, and Atlantic rainforest regions, and has high potential for sustainable use (Guedes, Antoniassi, & de Faria-Machado, 2017). Pequi was harvested in 265 municipalities in the Cerrado ecoregion, which produced approximately 76 thousand tons. Finally, 42 thousand tons of pequi were harvested from 2012 to 2017.

Juicy Fruits

Berries and juicy tree fruits are harvested all over the world for personal, informal economic and formal economic use. In the United States of America people commonly harvest wild low-bush blueberries, wild raspberries, wild strawberries, and less commonly serviceberries (*Amelanchier* spp.), chokecherries *(Prunus virginiana)*, and other species of wild cherry (Chamberlain, Emery, & Patel-Weynand, 2018).

Lingonberry or cowberry (*Vaccinium* vitis-idaea) is one of the most popular berries in American and European Nordic countries, and it is widely used in the human diet. It is a perennial evergreen shrub distributed in circumboreal regions of northern Eurasia and North America. Lingonberries are most commonly harvested by hand with berry rakes. Lingonberry is an important element of coniferous forests understories in terms of nature's contributions to people, and it also has cultural and economic importance, linked to a rural lifestyle. Major lingonberry-exporting countries are Sweden, Finland, and the Russian Federation (Padmanabhan, Correa-Betanzo, & Paliyath, 2016; Pouta, Sievänen, & Neuvonen, 2006; Woziwoda, Dyderski, & Jagodziński, 2020). A set of criteria and indicators were involved in assessing the commercial supply chain of bilberry in Finland, and suggested a lack of social sustainability due to decreasing involvement and consultations with forest owners and the local communities (Hamunen, Kurttila, Miina, Peltola, & Tikkanen, 2019).

Lingonberries are most commonly harvested by hand with berry rakes. In Finland, 11-26 million kg of bilberries and lingonberries were gathered in the 1990s. It is estimated that over half of the population still participates in berry picking based on the Nordic *allemannsretten* or "everyman's right", which is a long-standing right to move through and share resources on both private and public lands, including the right to pick berries and mushrooms in communal areas.

Some estimates suggest utilization rates of the two most common berries, bilberry (*Vaccinium myrtillus* L.) and lingonberry are low (4–15% of the total annual yield of wild berries), making this a very sustainable activity. One study found that approximately 32% of the total harvested of berries were for commercial sale (Turtiainen, Salo, & Saastamoinen, 2011). However, the demand for so called "super foods" has accelerated exports for global markets, and the volume of the Nordic wild berry harvest has doubled during the past two decades. Along with an increase in the market demand, lingonberry has been domesticated and commercially cultivated in several locations across Europe, Scandinavia, and also recently in the United States of America (Forest Europe, 2020; Padmanabhan *et al.*, 2016).

The land area covered in bilberry *(Vaccinium myrtillus)* bushes and locally harvested berries have declined in Central Italian Apennine regions in recent last decades (Nin, Petrucci, Del Bubba, Ancillotti, & Giordani, 2017). Regular gatherers of bilberry in Estonia use clearly delineated picking areas, and typically do not share their areas outside close family relations (Remm, Runkla, & Lohmus, 2018). Bilberries are also a popular wild food in the Czech Republic, where the number of households involved in the gathering of wild fruits has increased in recent years. The ratio of participants and yield of bilberries are the highest in wild fruit (Wolfslehner *et al.*, 2019). There is also a high demand on bilberries in France. The boom started in the late 1960s. At that time some gatherers in the Massif Central area increased the quantity of berries they were gathering by 500%. This gathering is regulated for non-residents (Larrère, 1982).

Most cacti produce edible fruit for humans but prickly pears of *Opuntia* species and fruits from *Stenocereus, Cereus,*

Box 3 8 **The many lives of a single plant.**

Based on (Paye, 2000, p. 142; Yetman *et al.*, 2020, p. 69).

The Saguaro cactus *(Carnegiea gigantea)* grows in desert of Arizona, California, and Mexico, and can reach up to height of 15 meters (50 feet) with a life span of about 200 years (Yetman *et al.*, 2020). The fruits are harvested by O'odham Indians, who cook the pulp to make jam, candy, syrup, and wine, but the wild plant also plays an important role in the lives of many other organisms in these environments of which humans are a part. It takes 50 years for this cactus to bear flowers and fruits. The cactus provides shelter and food for numerous organisms throughout its life span. Carpenter birds and elf owls make nests in the fleshy body of the cactus, and Harris's hawks build nests in the branches. Bats, doves, butterflies and bees enjoy the nectar when the cactus blooms during May. Many animals such as curved bill thrashers, horned lizards, coyotes, and javelin pigs also eat the fruits. As the cactus nears the end of its lifespan, aquatic beetles swim through the decomposing plant flesh. When the cactus is dead, it is home to termites, spiders, giant centipedes, banded geckos, cactus mice, and spotted night snakes.

Carnegiea and *Pachycereus* species are the most important for numerous peoples of the arid Americas **(Box 3.8)**. Fruits are gathered from the wild, semi-cultivated and cultivated stands. People commonly make use of a tool called "chicole" which is a long stick with a kind of basket in the top. With "chicole" people reach and pull fruits without damaging them or injuring themselves or the wild plant. The fruits harvested are stored in a basket or bucket for transporting them to homes and markets. In agroforestry systems, people sometimes leave fruit-producing cacti because they favor their propagation and take special care of these valued plants. Some species, mainly *Stenocereus, Cereus, Lemairocereus*, are cultivated, and processes of domestication and generation of varieties associated to human selection have been documented in Mexico (Casas & Barbera, 2002; Casas, Otero-Arnaiz, Pérez-Negrón, & Valiente-Banuet, 2007).

Beverages

Mead, Hyssop, Salep, teas, and wild coffees from dandelion greens and chicory are some of the many beverages people make from wild plants. The English term "tea" refers the infusion made of the leaves of *Camellia sinensis* but there are kinds of aromatic and refreshing beverages around the world. In Europe, 142 taxa of plants belonging to 99 genera and 40 families are reported the use of recreational tea (Sõukand *et al.*, 2013). In China, 759 plant species have documented for use as teas, and a market survey identified an additional 23 species used as herbal tea (Fu *et al.*, 2018). The majority of wild plants used are perceived as medicinal plants in local folk medicine or "folk functional foods". The status of the use of herbal tea is dependent on access to the natural resources, cultural and social contexts, and the habit of its use in the region and personal preferences of the consumer.

Salep is a beverage made from orchid tubers in Europe and central Asia. Harvesting wild orchid tubers for this purpose dates back to the medieval period. Six species of orchids are named as components of Salep. Tuber gathering for Salep has been cited as a cause of orchid population decline and causes conservation concern in Turkey and neighboring countries (Charitonidou, Stara, Kougioumoutzis, & Halley, 2019; Ghorbani, Gravendeel, Naghibi, & de Boer, 2014; Kreziou, de Boer, & Gravendeel, 2016; Masters, van Andel, de Boer, Heijungs, & Gravendeel, 2020). Scientists and conservationists recommend cessation of wild orchids harvest for this purpose (Ghorbani *et al.*, 2014).

Syrups, Gums and Resins

Indigenous tribes in Eastern North America know the sap of maples (*Acer* spp.) and call it "sweet water." When the first European explorers and colonists arrived, they learned of maple sap and boiling the sap down to produce syrup or sugar. Sugar maple *(Acer saccharum)* is the species most frequently tapped for sap production. Under the best conditions, sugar maples reach a tappable size in about 40 years and can continue to produce sap for a century (Ciesla, 2002). During the maple sugaring season, which lasts about six weeks in spring, an average maple tree will yield between 35 and 50 liters of sap, which will produce between 1 and 1.5 liter of pure maple syrup (Ciesla, 2002).

Maple syrup is produced only in the Eastern United States of America and Canada. Maple syrup production is a hobby that connects people to nature, provides supplementary income for farmers, and is an important cultural practice for indigenous peoples (Weiss *et al.*, 2019). As a large-scale commodity, maple syrup is a luxury item consumed worldwide **(Figure 3.41)**. The largest market for syrup is in the United States of America. Since the late 19th century, maple production in the United States of America has declined while that in Canada has increased. With sugar maple (*Acer saccharum* Marshall) often distributed throughout the region's forests, only a small percentage of potentially tappable trees are in use for maple syrup production (an estimated 0.4% in the United States of America; Ciesla 2002, Farrell and Chabot 2012). Maple syrup production is weather dependent and expected to be heavily affected by climate change, with the potential for it to be eliminated in southern reaches of its current distribution peaked in the 19th century, reaching a record 25,032,928 liters of maple syrup in 1860. (Iverson & Matthews, 2018).

In Europe, the main sources of tree sap are silver and downy birch trees (*Betula pendula* Roth and *Betula pubescens* Ehrh). Birch sap is colorless or slightly opalescent. It is used as a traditional drink, in traditional medicine, in veterinary medicine and as a cosmetic product. Gathering sap from birch and other trees was more widespread in earlier times. In Russia, Ukraine, Belarus, Estonia, Latvia and Lithuania it remains a more common practice. The most productive silver birch trees for sap gathering are those taller than 28 m. A birch tree can produce 36l gallons of sap in nine days. Experiments conducted in Estonia in the 1970s showed that the profit gained from the sap was six times the profit gained from timber. More recently birch sap is becoming a more commercial product, and is of interest to pharmaceutical companies (Grabek-Lejko, Kasprzyk, Zagu\la, & Puchalski, 2017; Mingaila *et al.*, 2020; Svanberg *et al.*, 2012).

Gum Arabic or acacia gum is a tree gum exudate gathered from a number of *Acacia* species and has been an important part of commerce since ancient times. Gum Arabic is used in food and drink industries, in pharmaceuticals and in printing and textile industries as thickening, stabilizing, binding and sizing agents. Gum resin products are harvested from natural exudates by herdsmen, women and children while herding and doing other activities. Yields of gum Arabic from individual trees are very variable.

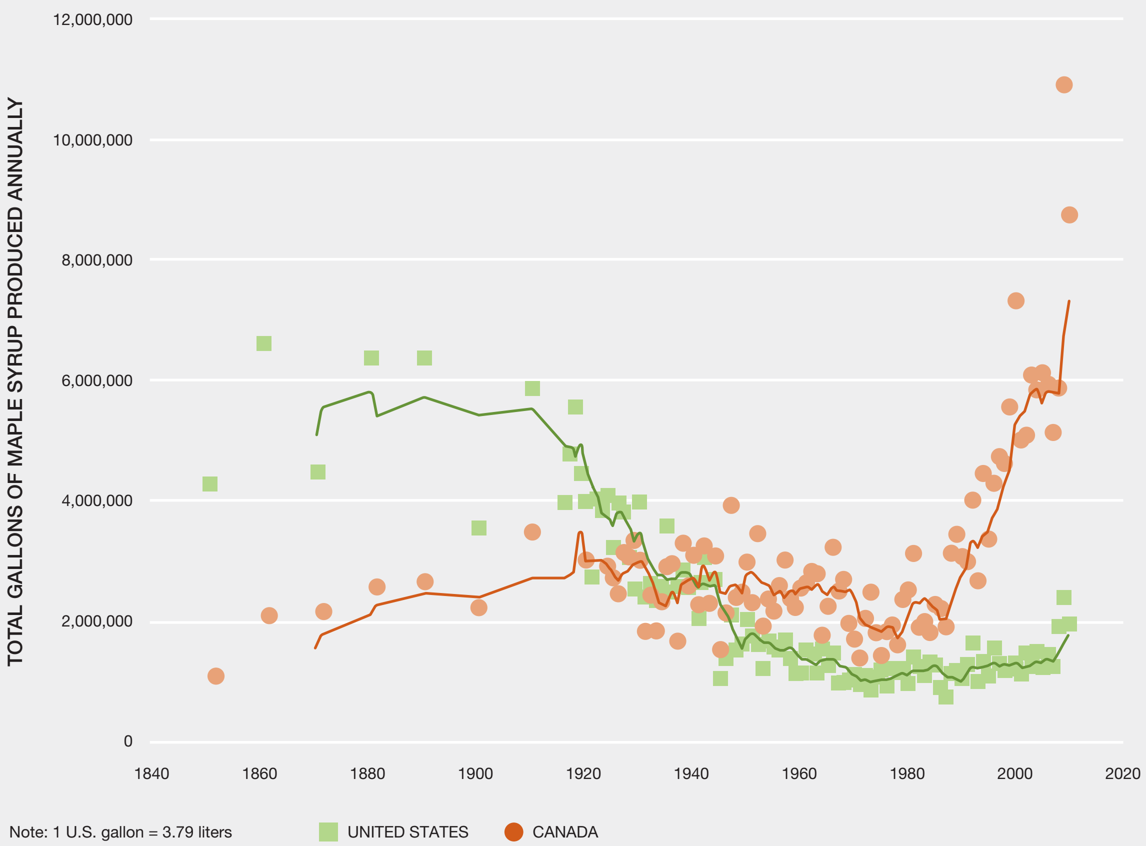


Figure 3 41 **Maple syrup production in the United States of America and Canada 1860–2010.**

Source: (Freire *et al.*, 2020) under license CC BY 4.0.

A tree yields an average of 250g of gum per season. Very small proportions of gum enter the local market, but some is directly sold as a road-side snack in some West African countries, in Niger for example (E.S. Barron personal observation). African countries export about 100,000 tons of gum Arabic annually, and demand was previously projected to reach 150,000 tons by 2020. The European Union is responsible for 80% of global trade in gum Arabic, worth around 125 million euros. Sudan is one of the biggest gum Arabic producers in the world and produces more than 80% of the total world gum Arabic (Wubalem Tadesse *et al.*, 2020; B. Wolfslehner *et al.*, 2019) **(Table 3.9)**.

Karaya gum is produced as an exudate from the genus *Sterculia* including *Sterculia urens* tree found in India and *Sterculia setigera* found in Africa and is used for many industries. World demand for karaya gum is about 7,000 tons, and Senegal is the leading exporter in Africa. The population of karaya trees once markedly declined due to crude traditional tapping methods which lead to the death of the tapped trees and over exploitation. Scientific tapping and proper harvesting methods are now priorities (Nair, 2004; Wubalem Tadesse *et al.*, 2020).

Wild edible mushrooms

More than 350 species coming from 18 orders of fungi are commonly eaten as food (Willis, 2018). The number of used wild edible mushrooms is likely much higher than that based on lists and assessments from individual countries, e.g., over 1000 species of edible mushrooms are listed in China (Wu *et al.*, 2019), 371 in Mexico (Moreno Fuentes, 2014) and 268 species are traded in Europe (Peintner *et al.*, 2013). The last comprehensive global assessment was conducted in 2004 (Boa, 2004), and given the high rates of taxonomic discovery among fungi, including of useful species (Dentinger & Suz, 2014; Willis, 2018; F. Wu *et al.*, 2019), a re-evaluation is overdue. Wild mushrooms are harvested for food in over 80 countries worldwide (Pieroni, Nebel, Santoro, & Heinrick, 2005a). Among wild-harvested fungi, most commonly consumed and traded are Chanterelles (*Cantharellus* spp.), Porcini (*Boletus* spp.). Truffles (*Tuber* spp.) Morels (*Morchella* spp.), Brittlegills (*Russula* spp.), Milkcaps (*Lactarius* spp.), Button mushroom (*Agaricus* spp.), and Matsutake (*Tricoloma* spp.). Wild edible mushrooms can be found in over 200 genera, and grow in a wide variety of habitats (Boa, 2004). Many of the most popular used species form symbiotic relationships, making them difficult if

Table 3 9 **Exports of gum Arabic (tons) from different African countries 2001–2010.**
Source: (Wubalem Tadesse *et al.*, 2020) under license CC-BY 4.0.

Country \ Year	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010
Sudan	7949	34382	13217	27444	33079	23149	n/a	37860	36636	48598
Nigeria	0	0	0	n/a	n/a	1314	14463	14124	40862	34780
Chad	12891	9161	9672	12044	14188	17816	11860	16219	9417	9509
Ethiopia	830	875	381	234	111	317	956	614	622	909
Tanzania	843	693	1252	1361	1169	965	1031	935	631	824
Cameroon	571	592	338	264	371	413	310	151	520	510
Senegal	121	0	0	213	323	475	610	836	935	330
Mali	482	750	704	52	28	17	29	1308	703	275
Burkina Faso	2	0	21	18	81	n/a	90	57	63	83
Kenya	23	0	92	23	32	28	75	165	41	75
Eritrea	n/a	n/a	116	49	495	38	688	419	350	51
Somalia	26	12	4	70	714	92	473	513	50	47
Niger	2	20	38	43	42	73	67	66	44	44

not impossible to cultivate. For example, all of the above-listed genera (with the exception of button mushrooms) form ectomycorrhizal symbioses with trees, while *Termitomyces* spp. which are widely consumed across Africa and Asia are symbionts of termites (Boa, 2004). Popular saprotrophic species include button mushrooms, straw mushrooms (*Volvariella* spp.), shitake *(Letinula edodes)* and oyster mushrooms (*Pleurotus* spp.), although these species are cultivated at large scale (Boa, 2004), they are also frequently harvested in the wild, for example in Malaysia (Fui, Saikim, Kulip, & Seelan, 2018), Benin (Codjia & Yorou, 2014), Mexico (Haro-Luna, Ruan-Soto, & Guzmán-Dávalos, 2019) or Italy (Pieroni, Nebel, Santoro, & Heinrick, 2005b).

To assess status and trends of wild useful fungi, literature searches were conducted via a variety of search engines (Google Scholar, EBSCO Host and SCOPUS). To this end a Google Scholar search with the terms "(gathering OR collecting OR picking OR hunting OR foraging) AND (mushroom OR lichen OR fungi) AND sustainable AND wild" served as the basis and variations in the combinations of these terms, as well as supplementation with the different use categories (e.g., ceremonial, medicinal, food) were used until 50% saturation of articles already in the database were reached. In total, 112 sources were reviewed (see the data management report for Chapter 3 systematic literature review for the gathering of fungi at https://doi.org/10.5281/zenodo.4659811).

The extent of usage of different species varies widely. Typically, ethnomycological studies report the use of tens to hundreds of species where a majority is harvested for personal use, gifting or barter (based on 20 articles from the literature review). A smaller number of popular species is sold at local and regional markets, while select species, often global commodities, are sold on to middlemen and traders to enter national and international markets (based on 9 articles from the literature review). This phenomenon is particularly well-documented in Mexico. For example, the Mazahua people use 31 species of wild mushrooms, of which 18 are sold in local or regional markets (Farfán, Casas, Ibarra-Manríquez, & Pérez-Negrón, 2007). The less popular species are also sometimes sold in mixed species bags, while a handful of highly-prized species including *Amanita caesarea* complex, porcini, morels, chanterelles and matsutake are targeted for export (Montoya, Hernández, Mapes, Kong, & Estrada–Torres, 2008; Pérez-Moreno, Martínez-Reyes, Yescas-Pérez, Delgado-Alvarado, & Xoconostle-Cázares, 2008). A similar imbalance in usage among taxa also exists at larger geographical scales as indicated by a comparison among European guidelines and legislations, where on lists from 24 countries with an average length of 55 taxa and a total of 268, only two taxa were listed in all countries: porcinis (*Boletus edulis* complex) and chanterelles *(Cantharellus cibarius)*. A further five (*Lactarius deliciosus, Morchella esculenta, Boletus badius, Agaricus campestris* and *Craterellus cornucopioides*) were listed in more than 70% of countries, while 134 (about 50%) were listed in only one or two countries (Peintner *et al.*, 2013). Finally, species preferences and use may shift over time as is highlighted by *Russula virescens* which was highly appreciated in the Southwest of France in the 18th century but is no longer consumed nowadays, while the chanterelle increased in popularity in this region (Duhart, 2012).

The use and appreciation of different mushroom species is deeply cultural, and whether a species is used and what for is often due to a multitude of factors, including language, geography, cultural and culinary traditions (Comandini & Rinaldi, 2020). For example, regions in Europe with similar occurrence of mushroom species (e.g., Southeast Europe *versus* Southwest Europe, or Eastern Europe *versus* the nordic countries) favor different species and the use of species is more strongly influenced by local tastes, traditions and commerce with neighbors than climatic variables or vegetation (Peintner *et al.*, 2013). In line with this, usage frequently reflects cultural interactions, for example in Finland, gatherers in Eastern parts of the country with stronger cultural influence from Russia prefer milk caps (*Lactarius* spp.), while those in Southwestern regions where French cuisine permeated through Swedish influence prefer porcinis and chanterelles (Comandini & Rinaldi, 2020). Immigrant populations often bring culinary traditions and preferences to their new homes, nicely illustrated in the Western United States of America, where a culture and tradition of gathering, along with different species preferences, was established by early immigrants from Europe, Asia and Russia (Arora, 2008a; Parks & Schmitt, 1997). Another salient example illustrating, fine-grained, context dependence of use are the false morels *(Gyromitra esculenta)*, which are consumed at quantity in Finland (Turtiainen, Saastamoinen, Kangas, & Vaara, 2012), and *Gyromitra infula*, which is harvested both in Nepal (M. Christensen, Bhattarai, Devkota, & Larsen, 2008) and Mexico (Pérez-Moreno *et al.*, 2008). These species are largely considered toxic and safe consumption rests on the knowledge of correct preparation (Peintner *et al.*, 2013), highlighting the importance of indigenous and local knowledge in shaping use of individual species.

The trade of edible fungi has been valued at 42 billion United States dollars in 2018 (Willis, 2018). However, this estimate includes mostly cultivable species and only two (porcinis and morels) out of the nine species evaluated are exclusively gathered in the wild, while other economically important wild taxa such as truffles, chanterelles and matsutake are omitted. Data on trade volumes also is often aggregated at higher levels that include both taxa from cultivation and wild gathering. For example (de Frutos, 2020) estimated international trade for edible fungi at 1.2 million tons for 2017 based on United Nations Comtrade data (https://comtrade.un.org/), using harmonized customs codes that include all species, except the genus *Agaricus. Agaricus* spp. constitute approximately 30% of the cultivated mushroom trade volume, so this figure is likely still influenced by the other four taxa cultivated at large scale [*Pleurotus, Lentinula, Auricularia* and *Flammulina;* (Royse, 2014)]. FAOSTAT aggregates data for all fungi into a “mushrooms and truffles” category, yielding a production of 10.9 million tons for 2017 (http://www.fao.org/faostat/en/#data/QC; Accessed 27.02.2021). Comparison with the international trade figure suggests that as much as 90% of trade volume may be based on cultivated fungi, although again it is unclear what proportion can be attributed to wild species. The FAO estimates also include *Agaricus* data and other cultivated species. The heterogeneity in taxonomic granularity of data accumulation and aggregated reporting for both cultivated and wild species makes it challenging to produce meaningful estimates of production and trade volumes of wild edible fungi. This problem constitutes an active area of work within the FAO, as reflected by the introduction of new harmonized system codes for widely traded wild plants algae and fungi coming into effect in January 2022 (World Customs Organization, 2019). Nevertheless, a body of literature focusing on specific regions or target species clearly highlights the economic importance and development potential of wild mushroom trade, especially for rural areas.

Our literature review yielded 24 studies that highlight a contribution of gathering and selling wild fungi to incomes of rural populations worldwide (3 Africa, 5 Americas, 8 Europe and Central Asia, 7 Asia Pacific). China, and Yunnan in particular, provides an excellent example of how the gathering of wild edible fungi can fuel economic development in rural areas. Yunnan harbors a large diversity of edible fungi and is the center of the wild edible mushroom industry in China (R. Hua, Chen, & Fu, 2017; Dongyang Liu *et al.*, 2018). Especially in the more remote areas of Yunnan, the contribution from gathering of wild fungi can reach up to 90% of annual household income (Arora, 2008b; R. Hua *et al.*, 2017; Huber, Ineichen, Yang, & Weckerle, 2010). In Nanhua county alone, the yearly production of wild fungi amounted to 7677 tons, valued at 80 million United States dollars (Dongyang Liu *et al.*, 2018). In 2015, the total yield of wild edible fungi for the whole province amounted to 0.17 million tons, with Yunnan being a major supplier of porcini (*Boletus* spp.) which are primarily exported to Europe and matsutake *(Tricholoma matsutake)* which are exported to Japan (R. Hua *et al.*, 2017; Huber *et al.*, 2010).

In all papers surveyed, gathering wild fungi was a supplemental activity to other forms of subsistence, primarily agriculture due to the seasonality of mushroom fruiting and year to year fluctuations in abundance and price. However, due to the highly perishable nature of the product that requires fast processing, the establishment of mushroom supply chains has led to lasting economic diversification in rural areas with the involvement of middlemen, mushroom traders and processing facilities (Arora, 2008b; Huber *et al.*, 2010; Dongyang Liu *et al.*, 2018). In Shangri-la, Diqing Tibetan Autonomous Prefecture, matsutake are often bought and sold several times before leaving the city, spreading the income not only to middlemen, who often do not have access to matsutake habitats themselves, but also to shops, restaurants and other facilities that were established near the mushroom markets (Arora, 2008b). In Mexico mushrooms are also sold to traders and middlemen, although here there was a greater emphasis on sale at local and regional markets for generation of income (Farfán-Heredia, Casas, Moreno-Calles, García-Frapolli, & Castilleja, 2018; Pérez-Moreno *et al.*, 2008).

None of the papers reviewed mentioned commercial export of fungi from Africa, but highlighted informal, local sale as the main form of generating income (e.g., Osarenkhoe, John, & Theophilus, 2014; Wendiro, Wacoo, & Wise, 2019; Yorou *et al.*, 2014). However, small scale export of porcini to Italy and the United States of America, primarily from Southern Africa, were indicated based on personal communication (Boa, 2004; Sitta & Floriani, 2008). Besides direct contributions to household income, wild mushrooms provide a rich source of protein and can help to bridge periods of food scarcity which often fall into the rainy season, e.g., in Ethiopia (Dejene, Oria-de-Rueda, & Martín-Pinto, 2017), West Africa (Yorou *et al.*, 2014) and Mexico (Farfán *et al.*, 2007).

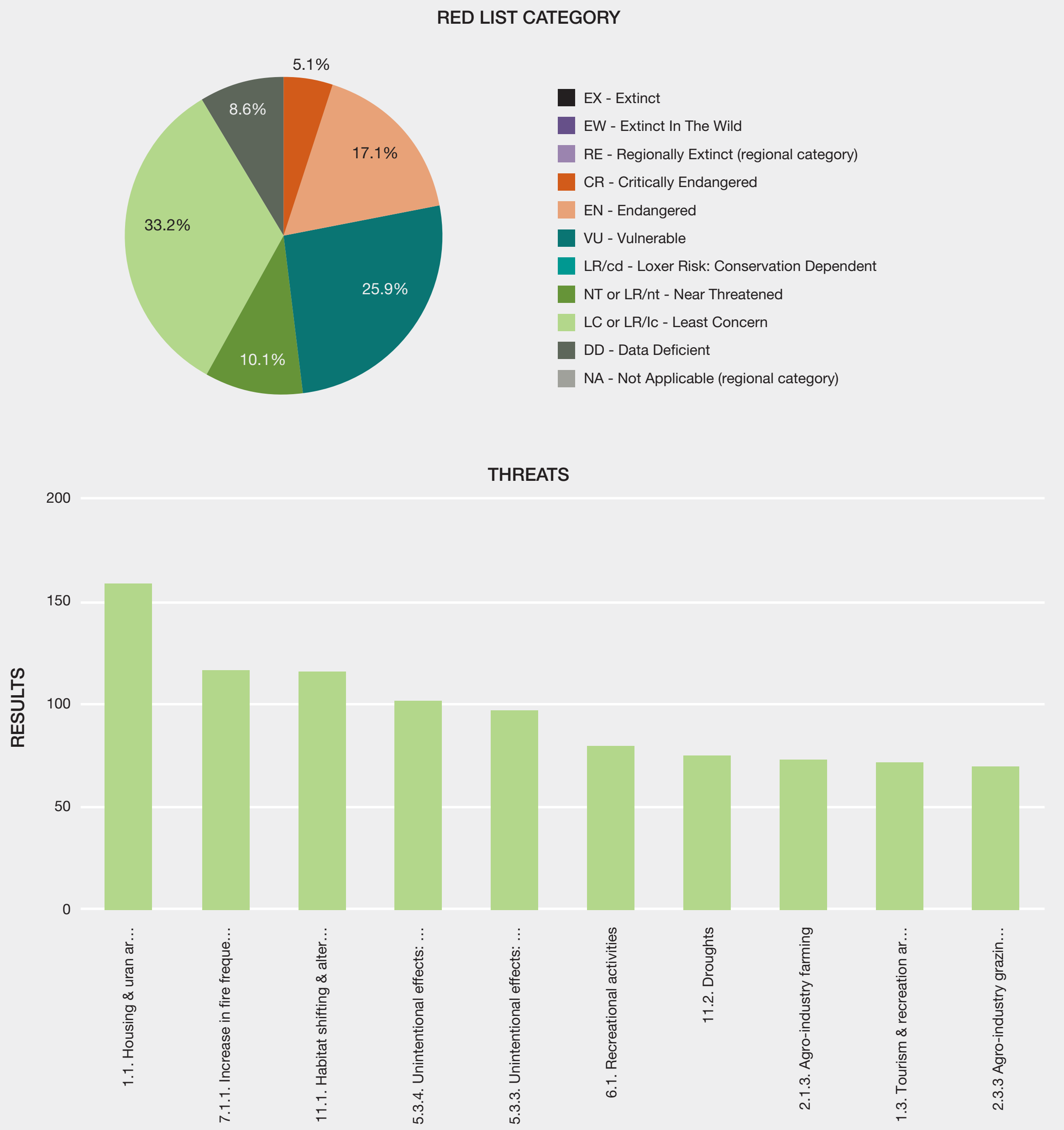


Figure 3 42 **The threatened status and threats of all assessed fungal species.**

At the top, the threatened status of all assessed 545 species and at the bottom, the threats of all assessed 545 species. Source: (IUCN, 2020b) © IUCN Red List Data. This figure was made using the International Union for Conservation of Nature website https://www.iucnredlist.org/search/stats, by selecting "Fungi" in the tab "Taxonomy".

Table 3 10 **Distribution of edible fungi assessed by the International Union for Conservation of Nature Red List in each IPBES region.**

Abbreviations: CR: Critically Endangered, EN: Endangered, VU: Vulnerable, NT: Near Threatened, LC: Least Concern. Source: (IUCN, 2020b) © IUCN Red List Data.

IPBES regions \ IUCN status	CR	EN	VU	NT	LC	Total
Americas	1	3	3	3	35	45
Asia Pacific		1	5	5	37	48
Africa			1	1	7	9
Europe and central Asia			7	5	36	48
All	1	4	9	5	41	60

In addition to the 27 sources mentioned above, a further ten studies were found where wild mushrooms were important contributors to a healthy diet and subsistence of people in economically marginalized positions. The use of wild mushrooms was also reported among several indigenous groups in the Amazon, most prominently the Jotï (Zent, Zent, & Iturriaga, 2004; Zent, 2008) and the Yanomami people (Fidalgo & Prance, 1976; Sanuma *et al.*, 2016). Recently, the Yanomami in Brazil started trading some mushrooms as a niche market (Sanuma *et al.*, 2016).

Although often considered the “meat of the poor” or emergency foods that can cover protein nutritional needs (Christensen *et al.*, 2008; Guissou, Lykke, Sankara, & Guinko, 2008; Oyetayo, 2011; Redzic, Barudanovic, & Pilipovic, 2010), this view diminishes the cultural importance of wild edible fungi. In some communities in Mexico, for example, mushrooms are considered delicacies with great flavor that are superior to meat (Farfán-Heredia *et al.*, 2018; Haro-Luna *et al.*, 2019). The strong appreciation and deep cultural traditions associated with gathering and consumption of fungi are reflected in the fact that many papers explicitly mention recreation, social bonding and stress release as a major reason why people gathered wild mushrooms (9 sources). Gifting and exchange of gathered fungi or products prepared from them among friends, family and members of the community were also mentioned several times (Garibay-Orijel, Cifuentes, & Estrada-Torres, 2006; Haro-Luna *et al.*, 2019; Pieroni *et al.*, 2005b).

Gathering and eating wild foods and engaging in culinary traditions provides a sense of place, identity and a connection with nature that is celebrated at festivals, e.g., in Spain (Fusté-Forné, 2019) or China (Dongyang Liu *et al.*, 2018) and has developed into a sizeable foraging tourism industry worth 800,000 euros per year in Spain (Fusté-Forné, 2019). Finally, a study comparing rural populations in Sweden, Ukraine and Russia showed that a high proportion of people engaged in gathering, irrespective of economic status (Stryamets, Elbakidze, Ceuterick, Angelstam, & Axelsson, 2015). Instead, the importance of commercial harvesting to supplement income increased or decreased inversely proportional to the standard of living and employment opportunities, highlighting that the social importance of gathering wild fungi may often be masked by economic necessity and reasons for gathering can shift over time.

In the International Union for Conservation of Nature Red list, 116 of 545 species of evaluated Fungi are used as human food, and 16 species of edible fungi are evaluated as threatened. With the exception of Africa, the distribution of the species assessed is relatively balanced in the other three IPBES regions (IUCN, 2020b) (**Figure 3.42**; **Figure 3.43**; **Table 3.10**). Less than 14% of edible fungi are threatened, which is lower compared to the overall level of the species assessed by the International Union for Conservation of Nature red lists (28%). However, due to the limited number of fungi assessed, the figure may not be representative of the global status. With regards to **Figures 3.42** and **3.43**, it is important to note that the majority of fungal conservation related inventory and monitoring has historically been based in Europe, hence the density of data from that region (Barron, 2011).

In a regional assessment, take China as an example, the threatened species list of China’s macrofungi assesses the overall threat status of 9302 species and 1.04% of the total number of species (97 species) is assessed as threatened (Yijian *et al.*, 2020). Among the 97 threatened fungi, there are 13 species used as food, 8 species are medicinal use, and other 8 species are used both for food and medicine **(Figure 3.44)**.

Based on the literature survey, land use change (10 sources), timber harvesting, deforestation (8 sources) and climate change (8 sources) were listed as the most common ecological threats that likely affect a broad range of

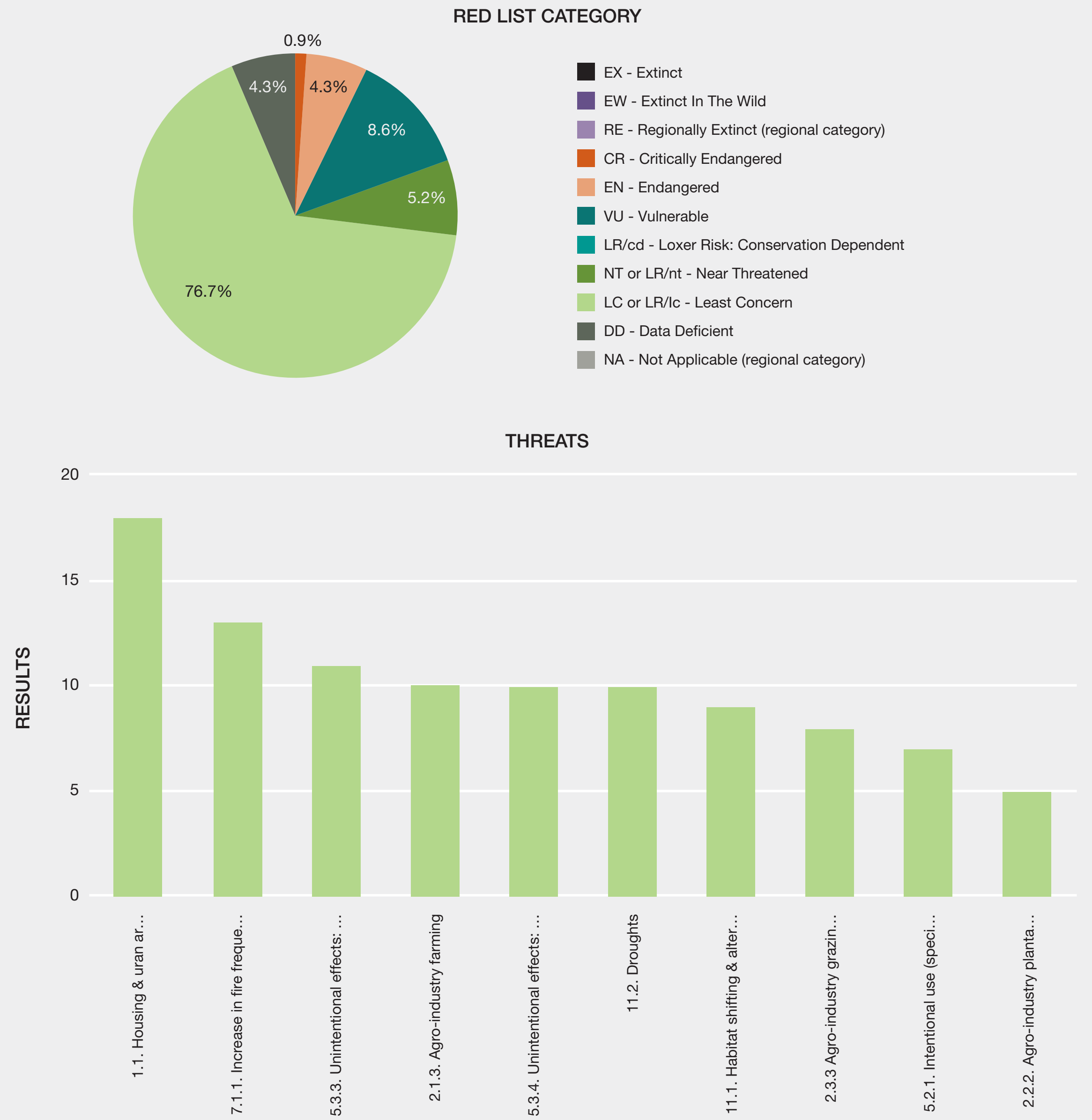


Figure 3 43 **The threatened status and threats of edible fungal species.**

At the top, the threatened status of all assessed 116 species and at the bottom, the threats of all assessed 116 species. Source: (IUCN, 2020b) © IUCN Red List Data. This figure was made using the International Union for Conservation of Nature website https://www.iucnredlist.org/search/stats, by selecting "Fungi" in the tab "Taxonomy" and then by selecting "Food-human" in the tab "Use and Trade".

edible fungi irrespective of their economic importance. Overharvesting was primarily reported on in the context of species gathered for commercial purposes (8 sources), with a particular focus on matsutake (Martínez Carrera, 2002; J. S. Brooks & Tshering, 2010; Dongyang Liu *et al.*, 2018) and truffles (Garcia-Barreda *et al.*, 2018; Radomir, Mesud, & Žaklina, 2018). Long-term scientific studies monitoring the effect of different harvesting techniques (picking *versus* cutting) showed no adverse effects of gathering fruitbodies on future production of epigeous (aboveground) fruitbodies using either technique, but instead identified trampling associated with gathering activities as reducing the number of fruitbodies (Egli, Peter, Buser, Stahel, & Ayer, 2006). Another study focused on harvesting techniques of the American matsutake *(Tricholoma magnivelare)* and also found no adverse effects of gathering on the number and weight of fruitbodies produced when mushrooms were picked using best practice methods (no soil removal, careful plucking of fruitbodies using a small tool) over the course of ten years (Luoma *et al.*, 2006) **(Box 3.9).** However,

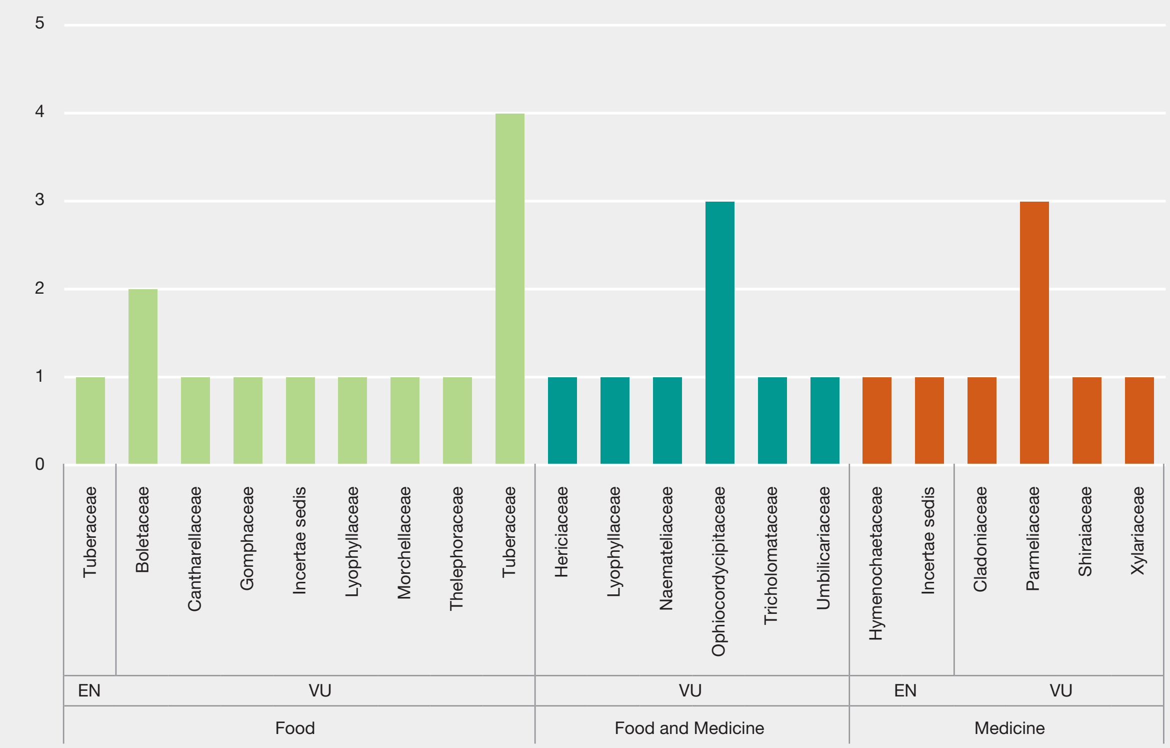


Figure 3 44 **China threatened fungi used as food and medicine.**

Based on (Ministry of Ecology and Environment of the People's Republic of China & Chinese Academy of Sciences, 2018). Abbreviations: EN: Endangered, VU: Vulnerable. See data management report for the figure at https://doi.org/10.5281/zenodo.6453079.

more disruptive harvesting methods using raking and soil removal resulted in fewer and lighter fruitbodies in the nine years following treatment, especially if soil was not replaced. This is in line with reports by Yi gatherers who expressed concerns about younger gatherers uprooting entire fruitbodies instead of using the more careful traditional gathering techniques (Dongyang Liu *et al.*, 2018). Overall, however, the long-term studies reconcile reports of overharvesting with those where no influence was reported despite commercial scale gathering (Arora, 2008b; Christensen *et al.*, 2008; Huber *et al.*, 2010), indicating that a good balance of commercial development and sustainable use is achievable with appropriate management practices. Species most at risk appear to be those subject to disruptive gathering practices such as matsutake and truffles, both of which are developing belowground, although structured research for a larger variety of species is currently lacking.

Indigenous peoples and local communities are both the main sources of knowledge with respect to status and management approaches (7 out of 8 articles reporting overharvesting) and key stakeholders in the use of wild edible fungi. Integrative research articulating local scale indigenous and local knowledge with other sources of knowledge that can incorporate the large year to year fluctuations in fruiting due to climatic variables and the impact of other environmental factors are required to better understand the multidimensional drivers influencing sustainable use. Consequently, the erosion and loss of indigenous and local knowledge also presents a major threat to sustainable use of wild fungi. This was reported in twelve studies reviewed and across all IPBES regions (5 Africa, 2 Americas, 2 Europe and Central Asia and 3 Asia Pacific). Indigenous and local knowledge is usually transmitted orally within families, often while engaging in gathering activities, so factors such as increased urbanization and associated cultural changes can decrease interest in gathering and the opportunity to do so (M. R. Emery & Barron, 2010). In three cases societal changes coincided with decline of wild edible fungi through deforestation and land use change and scarcity was one of the major reasons cited why people did not engage in gathering, e.g., in Burkina Faso (Guissou *et al.*, 2008) or Nigeria (Oyetayo, 2011; Uzoebo *et al.*, 2019). One of the latter also cited social stigmas associated with gathering, which all together can lead to a situation of rapidly declining indigenous and local knowledge.

Case studies from the United States of America and Europe highlight policies that are rooted in different philosophies

Box 3 9 **Matsutake and sustainable management.**

Matsutake *(Tricholoma matsutake)* and the closely-related species *T. magnivelare* and *T. caligatum*) are subject to some of the richest literature available with regards to management practices for wild edible fungi (Tsing, Satsuka, & for the Matsutake Worlds Research Group, 2008). Matsutake are highly appreciated in Japan where productivity has been in decline since the 1940s. *T. matsutake* grows as an ectomycorrhizal symbiont of Japanese red pine, a pioneer species that is commonly found around settlements (Saito & Mitsumata, 2008). For centuries people have been coppicing the *satoyama* (village forests) to harvest wood for fuel and other uses which created a favorable habitat for matsutake. A low point in the matsutake production was reached in the 1970s when many households switched to propane gas and oil fuels, which was considered the main reason for the decline in productivity (Saito & Mitsumata, 2008).

Due to the high market prices, especially for Japanese matsutake which can reach over 400 United States dollars per kg (2006), research has focused on silvicultural approaches for habitat improvement to increase matsutake yields (Saito & Mitsumata, 2008). In a comparison of different land management practices, the most successful one was rooted in the traditional *irai* system, where wild algae, fungi and plants are considered a communal resource of the village. This management practice involves joint habitat improvement sessions and days where everyone can gather which not only improved matsutake production, but also provided community-building social activities and ultimately a virtuous cycle where increased matsutake production and the social aspects leads to increased interest in participating in management activities (Saito & Mitsumata, 2008). Similar community-based management practices have proven successful in Shangri-la, Diqing Tibetan Autonomous Prefecture, China where matsutake harvest rights are organized by village and overharvesting and competition between gatherers are mitigated by instituting “rest days” of 3 to 5 consecutive days where gathering is prohibited once the quantity of matsutake sales declines (Arora, 2008b). Although the measure was implemented as a means to maximize profit, it also prevents harvest of very young specimens and may thereby benefit the reproductive potential of the fungus. Conflict among gatherers was further minimized by charging high fees for gathering permits for outsiders. Despite matsutake contributing the majority of household income in the region, there were no concerns voiced about declining numbers of mushrooms (Arora, 2008b).

of nature and perspectives on resource management (Tsing *et al.*, 2008). When market demand for wild edible mushrooms increased in the United States of America in the 1980s, restrictive gathering laws were put into place, either requiring permits for gathering, selling and buying mushrooms (Rebecca J. McLain, 2008), or forbidding gathering outright (Arora, 2008a). This led to *de facto* criminalization of gathering and gatherers that often had a long history of engaging in this activity but were not involved in the process of developing meaningful regulation. Consequences were increased volumes of mushrooms sold via black and grey markets (Arora, 2008a; Parks & Schmitt, 1997) and a reframing of gathering from a family activity as work or an outright illegal activity, threatening transmission of indigenous and local knowledge (Arora, 2008a; Rebecca J. McLain, 2008).

Similarly, mushroom gatherers and traders were not included as stakeholders in the development of the Forest Development Strategy in Serbia, where only commercial entities can apply for permits to gather wild plants, algae and, fungi (Radomir *et al.*, 2018). Serbia houses a rich variety of popular edible mushrooms, most prominently black truffles *(Tuber melanosporum)* which can fetch a market price of up to 4000 euros per kg. Due to high taxes levied on gathering and selling truffles, and above-mentioned restrictions on permitting, there is a flourishing black market and the majority of truffle export is purportedly going through illegal routes (Radomir *et al.*, 2018), a situation that neither benefits stakeholders nor allows for a realistic assessment of how to balance gathering activities with a healthy forest ecosystem.

In Southern Europe, wild truffle populations have been in decline, largely due to habitat degradation and climate change (Büntgen *et al.*, 2012; Garcia-Barreda *et al.*, 2018; Pieroni, 2016). However, an assessment of policies and regulations relating to truffle gathering in Spain suggest that lack of appropriate management strategies may further exacerbate this trend (Garcia-Barreda *et al.*, 2018). In Spain, gathering rights in public forests are auctioned off for terms of two to six years to private gatherers, which creates little incentive to invest in long-term strategies to maintain harvests. The situation becomes more acute as productivity declines and bidding becomes economically unattractive to commercial entities. In this case harvesting rights are purchased by municipalities which often leads to overexploitation, excessive trampling and damaging picking techniques as a larger number of gatherers face stiff competition with each other (Garcia-Barreda *et al.*, 2018). Nevertheless, overall truffle production in Spain has increased in recent years due to the contribution of truffles grown in plantations, whose share rose from 10% in 1998 to 60% in 2012 (Garcia-Barreda *et al.*, 2018), indicating that cultivation and silvicultural approaches are important avenues towards sustainable use for highly prized and highly commercialized species.

Wild vegetables

Wild vegetables are an important part of the human diet. The gathering and consumption of wild vegetables are of great value in three ways. (i) They contribute to food security and famine, (ii) they are playing an increasingly important role as health food, and (iii) diets including wild vegetables pass on traditional flavors and cultural influences. Wild fruits and mushrooms are more frequently gathered than wild vegetables, and many wild vegetables have been forgotten. Herbicides have also contributed to their disappearance. Some species are regaining popularity as gourmet or health foods (Łuczaj *et al.*, 2012).

Wild vegetables were commonly eaten in the past, especially in times of scarcity. In non-famine times, they diversified monotonous diets. Children ate some wild vegetables with an acidic taste *(Rumex, Oxalis)* as snacks (Łuczaj *et al.*, 2012). Some species are still gathered, such as *Asparagus acutifolius* or *Scolymus hispanicus*, as a part of traditional diets **(Table 3.11)**. The Mediterranean diet is a model of healthy dietary patterns and has been recognized on the UNESCO Representative List of the Intangible Cultural Heritage of Humanity for Italy, Spain, Portugal, Morocco, Greece, Cyprus, and Croatia. Cultural and historical factors diversify the use of wild vegetables (Łuczaj, Zovko Končić, Miličević, Dolina, & Pandža, 2013; Łuczaj & Dolina, 2015; Łuczaj, Łukasz, Jug-Dujaković, Dolina, Jeričević, & Vitasović-Kosić, 2019; Geraci, Amato, Di Noto, Bazan, & Schicchi, 2018).

Wild vegetables can be important local commodities and are sold at high prices in local and regional markets. However, if they are not gathered as wild vegetables, they are often considered weeds because they need little attention or management and are gathered from the wild, agricultural or disturbed spaces. Wild vegetables are often associated with traditional production systems and a long history of local selection and usage. In France, at least until the 1980s, people in some rural areas ate wild vegetables at the turn of the seasons, bitter salads in spring to purify the blood, and diuretic vegetables in autumn to prevent winter rheumatism (Fédensieu, 1988; Schaal, 1993).

Local and traditional knowledge is an important factor in maintaining the sustainability of wild vegetable gathering, cooking and consumption. This knowledge of wild vegetables may serve as baseline data for sustainable use (Ahmad, Ahmad, & Weckerle, 2013; Konsam, Thongam, & Handique, 2016; Maroyi, 2013; Wujisguleng & Khasbagen, 2010). Knowledge on food plants is, however, eroding in various parts of the world. In Mexico, rural indigenous and mestizo populations commonly eat wild greens called *quelites*, mainly gathered when weeding the fields; the most common species are *Amaranthus hybridus, Chenopodium berlandieri, Anoda cristata, Porophyllum ruderale* (Bye, 1981). Perceived as poor people's food, they are disappearing from peoples' diets, but there are actions to promote them (Mera Ovando, Castro Lara, & Bye Boettler, 2011).

Weeds from rice fields are especially consumed in Asia and still play an important part in the diet in Northern Thailand (Cruz-Garcia & Price, 2011) and Laos (Kosaka *et al.*, 2013). There was little evidence of wild greens consumption in South America. In the Amazon, most people are not keen on greens; the few wild species occasionally consumed are *Phytolacca rivinoides* and *Talinum* spp. (Katz *et al.*, 2012). In Africa, a large number of indigenous or naturalized vegetables, such as baobab leaves or spider plant *(Cleome gynandra)*, contribute to dietary diversity and food security, but have been neglected in some areas (Towns & Shackleton, 2018).

Two widely consumed and popular wild vegetables in the United States of America are fiddlehead ferns and ramps (wild onions). Fiddleheads are newly emerging and immature fronds of the ostrich fern (*Matteuccia struthiopteris* (L.) Todaro), which occurs throughout temperate areas of the country with high soil moisture. For many they are an early food of spring, are also part of rural, local economies and can sometimes be found in larger grocery store chains in regions where they are popular. Total yields are estimated at 100,000 pounds annually, which is believed to be a sustainable yield. Ramps *(Allium tricoccum)* are a spring ephemeral species popular in the Eastern and central northern United States of America. Like fiddleheads, they

Table 3.11 **Comparison of the use of wild vegetables among Mediterranean countries.**
Source: (Geraci *et al.*, 2018) under license CC-BY 4.0.

Numbers \ Country	Italy	Spain	Turkey	Morocco	Croatia / Herzegovina	Cyprus / Greece
Families	40	53	36	37	32	23
Genera	162	158	97	98	74	57
Taxa	299	277	151	158	98	76

are an early spring wild crop that is highly prized and celebrated as part of the return of spring. There is a long history of local and subsistence use of this species, which became nationally recognized in the 1990s due to a growing interest in it as a specialty food product. Now sold nationally in restaurants and health food stores, the accompanying market expansion has led to concerns regarding sustainable harvesting. Total quantities harvested are undocumented (Chamberlain, Emery, & Patel-Weynand, 2018).

Seaweeds, or "ocean vegetables", are collected throughout coastal areas all over the world. Historically, coastal people have been gathering and using seaweeds and seagrasses for a variety of purposes, including food, feed, fertilizer, medicine, fibers, biofuel and materials; they are included here as food is a primary reason for collection. Globally, total macroalgal production has increased by approximately 5.7% per annum (including harvest of wild species and cultivation) (FAO, 2014c; Rebours, Friis Pedersen, Øvsthus, & Roleda, 2014). By volume, production is dominated by aquaculture (>96%), which resulted in 27.3 million tons of annual global production in 2014 (Lotze, Milewski, Fast, Kay, & Worm, 2019; Mac Monagail, Cornish, Morrison, Araujo, & Critchley, 2017).

Despite the large scale of production from aquaculture, wild seaweed harvesting still plays an important role in many cultures. Thirty-two countries report active harvesting of seaweeds from the wild, with over 800,000 tons harvested annually from natural beds. Methods, regulations and management regimes vary widely across species and countries. European, Canadian and Latin American seaweed production still comes from harvesting wild populations (Buschmann *et al.*, 2017; Rebours *et al.*, 2014). Chile, China and Norway lead in exploitation of wild seaweed stocks. The Chilean harvest by artisanal fishers has been around 400,000 tons over the last 10 years, and there is concern about the environmental impacts of kelp removals. The marine license vetting committee of Ireland grants licenses to mechanically harvest seaweed and considers the potential negative impact on the marine environment (Mac Monagail *et al.*, 2017). Seaweed has been harvested in Brittany for several centuries, where this activity became industrial in the 18th century (Arzel, 1987) **(Box 3.10)**.

In Hawai'i seaweeds (Limu) are used for food, medicine, and ceremony as a traditional wild green. In recent years, more young Hawaiian men than women reported gathering wild seaweeds, indicating a cultural shift from pre-Contact Hawai'i, when women were the predominant gatherers and consumers of limu. Knowledgeable adults report a decline in the abundance of wild seaweeds driven by over-picking and pollution (Hart, Ticktin, Kelman, Wright, & Tabandera, 2014).

Box 3 10 Seaweeds harvest in Brittany (Western France).

The tip of the Brittany peninsula is particularly rich in seaweed, where over 330 species of macroalgae have been reported. There are two types of seaweed harvesting (Garineaud, 2017). Kelp is harvested from the sub-tidal sea in the archipelago of Molène-Ouessant (off the tip) and on the northern coast of the tip (from Le Conquet to Roscoff). This activity is locally considered as part of small-scale fisheries, with environmental knowledge transmitted within the family (Garineaud, 2015). Two types of kelp are harvested: *Laminaria digitata* (40 000 tons/year) and *Laminaria hyperborea* (25 000 tons/year) (Mesnildrey, Jacob, Frangoudes, Reunavot, & Lesueur, 2012). They are used to produce alginate, a gelling-thickening agent used in industry, especially the food industry. Two companies buy 95% of the harvest. This exploitation is considered sustainable (Frangoudes & Garineaud, 2015) because it is followed and controlled by a scientific institution, IFREMER (Institut Français de Recherche pour l'Exploitation de la Mer), in collaboration with the kelp collectors and industrial companies. However, the economic dependence on two companies and the lack of diversification of trade makes the practice vulnerable.

About 30 species are harvested on shore, in quantities of a few kilograms to several tons per year, reaching a total of about 10,000 tons. Around 300 collectors are involved in this activity, with different status, from seasonal workers to small processing companies (Garineaud, 2017). The most harvested species are Fucales and edible seaweeds such as *Palmaria palmata, Himanthalia elongata* or "pioka" (*Chondrus crispus* and *Mastocarpus stellatus*). The seaweeds are harvested by hand, or with scissors when clinging to a rock. Then they are dried, either preserved in salt or processed and sold fresh, depending on the species, the use and the collector. They are mainly used in food, industrial and pharmaceutical products. It is difficult to analyze this exploitation because of the diversity of harvested species, outlets and stakeholders. The lack of scientific knowledge, follow up, and control of this activity makes it vulnerable to changing conditions. It is difficult to establish administrative frameworks, exploitation regulations and labels matching with the stakeholders and their practices. The main risk with regards to sustainable use would be to turn this small-scale exploitation into a more intensive, more industrial and less diversified trade. Climate change is also likely to have an impact on seaweed harvesting and increase variability of the resource. Some species have already been displaced (Gallon *et al.*, 2014; Raybaud *et al.*, 2013). It is unclear how companies will adapt to variability and changing environmental and social conditions (Garineaud, 2017). Finally, the lack of information, transparency and accessible data makes understanding the social dimensions more difficult.

Protista and blue-green algae

The terrestrial cyanobacterial species *Nostoc flagelliforme*, commonly called Fai-Cai (Fat Choy), lives in desert or semi-arid grasslands in the Asia Pacific region, and is used as a vegetable in Chinese cuisine (Dai, 1992; Gao, 1998; YL Geng & Jiang, 1991). Herders scrape the vegetable with rakes. Indigenous and local knowledge suggests one must forage over approximately 10 acres of grassland to harvest 100g of dry fat choy. The raking can destroy the delicate grasslands and accelerate desertification. Therefore, the species was up-listed into the Class I of state key protected wild plants (even though it is not a plant) in 2000 and harvest and trade were banned at that time (But, Cheng, Chan, Lau, & But, 2002).

Nostoc commune or Ge-Xian-Mi (Rice of Immortal Ge) is the second edible species of Nostoc, originally listed for use in the The Compendium of Materia Medica (S. Li, 1596) by Shi-Zhen Li (1518–93?) of the Ming Dynasty. The name of Ge-Xian-Mi is related to Ge Hong (AD 284–364), a Taoist theoretician of the Eastern Jin Dynasty, who used *N. commune* as food during periods of famine and later introduced it to the emperor. It is used for health food and herbal medicine however the wild type of *N. commune* has been decreasing as a result of recent increases in market demand and environmental pollution. Artificial culture of the blue green algae generates economic benefits (Diao & Yang, 2014; Nazih & Bard, 2018). Nostoc species are still consumed, not only in China, but also in various countries such as the Philippines, Thailand, Japan, Fiji, Peru, Ecuador, Mexico, Mongolia, and Siberia (Borowitzka, 2018).

High-quality agar and agarose for bacteriology and pharmaceuticals originated from wild harvested *Pterocladiella capillacea*. A report reveals a decline in biomass coupled with a peak in wholesale prices, which have resulted in overharvesting in some countries in due to this increased economic exploitation (Patarra, Iha, Pereira, & Neto, 2019). Ongoing unsustainable commercial harvest of the algae could result in further marine ecological damage; thus, the future of the industry is uncertain.

3.3.2.3.5 Medicine and hygiene

Humans use wild plants and fungi for medicinal purposes all over the world. Gathering wild species for medicines is motivated by a range of factors. These include poverty or difficulty accessing medical assistance, traditional knowledge and beliefs, cultural heritage, or for profit due to commercialization. There is also a growing demand for products produced at least in part from wild harvested plants and fungi, to complement chemical medicines in many high-income countries (Lamrani-Alaoui & Hassikou, 2018; Lanker *et al.*, 2010; H. Liu, Luo, Heinen, Bhat, & Liu, 2014; Nekratova & Shurupova, 2016; L. Petersen, Reid, Moll, & Hockings, 2017; K. M. Stewart, 2003).

A large number of ethnobotanical studies have generated inventories and analysis of medicinal and hygienic uses of wild plants. Online databases summarize information on medicinal plants. For example, the Kew royal botanical garden has established the Medicinal Plant Names Service (https://mpns.science.kew.org), the Africa Museum in Brussels runs the Prelude Medicinal Plants Database (https://www.africamuseum.be), and databases like Native American Ethnobotany (http://naeb.brit.org/), the Indian Medicinal Plants Database (http://www.medicinalplants.in/) and the China National Genebank (https://db.cngb.org) all include information on medicinal uses.

These inventories of medicinal plants outline the threat level to the species, conservation status, or priority of conservation for further actions. In South Africa for example, 2,062 indigenous plant species (10% of the total national flora) have been documented for use as traditional medicine. Of these, 82 wild medicinal plant species (0.4% of the total national flora) are considered threatened with extinction at a national level (V. L. Williams, Victor, & Crouch, 2013). Thirteen percent of Myanmar medicinal plant species are considered threatened in the International Union for Conservation of Nature Red List of Threatened Species (DeFilipps, Krupnick, & Krupnick, 2018). These data suggest possibilities for future research, conservation programs, sustainable harvesting projects, management and regulations.

When and how wild plants or plant parts are gathered have important effects on their medicinal value. Each category of medicinal plant has its specific collection time to maintain not only efficacy, but also sustainability. In this regard, Kletter and Kriechbaum (2001, p. 12) remarks “a plant has medicinal value when it is harvested at the right time, but is mere grass when harvested during the wrong season”.

Local and traditional knowledge is a key to the sustainable gathering of wild medicinal plants. Of the articles retrieved in the Web of Science published between 2000–2020, more than one third (n=117/349) mentioned “traditional knowledge”. By its very nature, traditional knowledge is holistic in nature, thus in these articles it was not always distinguished as being specifically for medical use, and could also be related consumption for food or aromatic uses. This is consistent with the fact that many wild medicinal plants have multiple uses at the same time. Like in Angola, 35% of the 127 Leguminosae plants are only used medicinally by the local communities, while the remaining species were reported to have many other uses (S. Catarino, Duarte, Costa, Carrero, & Romeiras, 2019).

Wild plant species are chosen for pharmaceutical studies through different methods. One method what has come to be known as bioprospecting: the investigation of indigenous uses of wild plant species based on indigenous local knowledge that can offer strong clues to the biological activities of those plants. There are many examples of this knowledge being used by companies who either do not financially compensate local people at all, or do not do so in proportion to the value of their resultant profits. This is commonly known as biopiracy, and is a major issue is many developing countries and with indigenous communities around the world (Benjaminsen & Svarstad, 2021; Shiva, 2007). In relation to the definitions of sustainable use reviewed in Chapter 2, this form of exploitation is considered as a form of unsustainable use. Scientific experimentation is another method through which medicinal knowledge of natural products has over the last few centuries (D. A. Dias, Urban, & Roessner, 2012). About a quarter of all Food and Drug Administration and/or the European Medical Agency approved drugs are plant based (Thomford *et al.*, 2018). From 1981 to 2002, around 49% of the small-molecule new chemical entities that were introduced were from natural products or based on natural-products. The utilization of natural products in order to discover and develop new drugs is an active area of research (Koehn & Carter, 2005; Newman & Cragg, 2007, 2007).

Medicinal Fungi

Fungi are also widely used for medical purposes, especially in the Asia Pacific Region. Our literature review yielded 33 studies that detailed the use of medicinal fungi from all IPBES regions (Africa 8, Americas 7, Europe and Central Asia 8 and Asia Pacific 8). Of these, 90% also reported on species used for food, so many of the aspects pertaining to sustainable use are shared with wild edible mushrooms (see section 3.3.2.3). All studies reporting on wild species and their uses (12 in total) reported fewer medicinal species than species used for food and often species were used both as food and medicine. The largest number of medicinal species was reported from China, with 692 species with medicinal properties with 277 species considered as both food and medicine (Wu *et al.*, 2019). Mexico also hosts a large variety of medicinal fungi with a survey reporting the use of 70 species to treat over 40 different conditions, again many with dual use as food and medicine (Guzmán, 2008). Medicinal fungi also have a long history of use in Europe, where interest in traditional medicines has been increasing again recently after a decline in use in the 20th century (Comandini & Rinaldi, 2020).

Box 3 11 **Status and trends of caterpillar fungus in the Nepalese Himalayas.**

Ophiocordyceps sinensis (Berk.) G.H.Sung, J.M.Sung, Hywel-Jones & Spatafora, (Hypocreales, Ophiocordycipitaceae) is a high-altitude fungus reported only from the alpine meadows in Nepal, India, Bhutan and China. Locally called Yar-tsa-gunbu (summer grass, winter insect), it occurs from 3,540 m to 5,050 m above sea level across 24 different northern districts in Nepal (S. Devkota, 2008) and up to 5,200 m in Bhutan (Cannon *et al.*, 2009). It is an entomopathogenic fungus that parasitizes over 50 species of Thitarodes (Hepialidae) moth larvae (X.-L. Wang & Yao, 2011).

In the gathering season (May – July) and particularly when the snow melts, gathering is extensive. As many as 70,000 collectors (men, women, and children) have been reported across 25 principal gathering pastures in a single district (Dolpa of Nepal), living in temporary tent camps for about two months (S Devkota, 2009). The fungus provides a substantial source of cash income for many households: 21.1% contribution to the total household income and 53.3% to the total cash income among rural inhabitants and helping to fund childrens' education, food purchasing, household construction and debt repayments (Pouliot, Pyakurel, & Smith-Hall, 2018; Shrestha & Bawa, 2014). Apart from this, subsidiary incomes in mountain communities come from farming, animal husbandry, collection and trade of other medicinal and aromatic plants (Olsen & Larsen, 2003).

The global annual collection of caterpillar fungus is roughly estimated at 85-185 metric tons (Winkler, 2008). Indigenous peoples and local communities living in the Nepalese Himalayas use it for the treatment of different diseases like diarrhea, headache, cough, rheumatism, liver disease, and also as an aphrodisiac and tonic (S. Devkota, 2006). However, the main market is China, where there are several reasons behind increasing demand. Many consider the species as valuable medicinal fungi in accordance with traditional Chinese medicine. It is traded as the "Himalayan Viagra" and prices have exceeded 140,000 United States dollars per kg for the best quality in Chinese markets, depending upon size, color, aroma, and region of origin (Shrestha & Bawa, 2014). The high number of collectors, their trampling effects on fragile subalpine and alpine landscapes, wild species poaching, improper garbage disposal and annual large harvested volumes have raised several sustainability concerns (Byers, Byers, Shrestha, Thapa, & Sharma, 2020; S Devkota, 2009; Pouliot *et al.*, 2018).

The Chinese government has supported been thoroughly making efforts to reduce dependence on wild Ophiocordyceps sinensis through cultivation and fermentation technologies (Yue, Ye, Lin, & Zhou, 2013). Advanced biotechnology is being applied to cultivate Paecilomyces hepialid (fermentation mycelium) with active ingredients from the natural caterpillar fungus as well as compounds of its equivalent medicinal value (Ji *et al.*, 2020). There has been intensive focus on the artificial cultivation of the caterpillar fungus which has yielded successful approaches for its propagation and breeding (X. Li *et al.*, 2019). The emergence and application of culture-based techniques as a substitute for wild caterpillar fungus and the development of artificially bred varieties are a promising path towards protection and sustainable use of wild caterpillar fungus resources.

The most common medicinal fungi include *Ophiocordyceps sinensis* (caterpillar fungus), *Ganoderma lucidum* (lingzhi or reishi), *Lariciformis officinalis, Lentinula edidodes* (shitake), *Trametes versicolor* (turkey tail), *Schizophyllum commune* (the split gill) and *Pleurotus* spp., especially *Pleurotus tuber-regium* which is used medicinally across Africa (Milenge Kamalebo, Nshimba Seya Wa Malale, Masumbuko Ndabaga, Degreef, & De Kesel, 2018; Oyetayo, 2011).

Medicinal fungi produce a range of active compounds, many of which have been shown to have anti-oxidant, anti-tumor or anti-microbial properties (Wu *et al.*, 2019). To this end, *G. lucidum* is probably the most intensively studied species. It produces over 400 bioactive compounds and has been dubbed "the mushroom of immortality" in China where it has been used for over 2,400 years (Cör, Knez, & Knez Hrnčič, 2018). Nowadays it is widely used to supplement cancer treatment both in China and Western countries. Several records indicating the medicinal use of lichens in Spain and Nepal were also found (Shiva Devkota, Chaudhary, Werth, & Scheidegger, 2017; González-Tejero, Martínez-Lirola, Casares-Porcel, & Molero-Mesa, 1995). Perhaps the most valuable species globally is the illusive caterpillar fungus *(Ophiocordyceps sinensis)*, which grows only in the Himalayan mountains **(Box 3.11)**.

Gathering of caterpillar fungus *(Ophiocordyceps sinensis)* has dramatically increased over the last 20 years. A short seasonal and rotational approach for gathering is useful for its sustainability. Caterpillar fungus extraction provides up to 72% of household income in the area, and estimates of households involved in the short seasonal gathering range from 52% to 98%. Understanding of local commercial harvest and trade supports sustainable management (J. He, 2018; Kuniyal & Sundriyal, 2013; Woodhouse, McGowan, & Milner-Gulland, 2014).

Seeds, Leaves and fruits for medicinal use

The gathering of seeds, leaves and fruits for medicinal use is usually non-lethal and seasonal. In some cases, the average annual harvest is high but the population size is consistent, such as with *Aloe ferox* in South Africa and *Euphorbia antisyphilitica* in Mexico (Martinez-Balleste & Mandujano, 2013). These species were once included in the Convention on International Trade in Endangered Species of Wild Fauna and Flora, but after assessing the sustainability of harvest and trade, their products have been exempted from strict control. In order to sustain the trade, improving the techniques of wax extraction and promoting fair trade pricing structures that benefit local harvesters have been suggested (Martinez-Balleste & Mandujano, 2013).

Certification schemes can support management for sustainable use. For example, harvesting of the fruits of *Schisandra sphenanthera* in Chinese forests meet sustainable wild harvesting standards, with an incentive to maintain habitat outside formal protected areas based on FairWild Standards (2010) and Giant Panda Friendly Products Standards (2012) (Brinckmann *et al.*, 2018). In the absence of effective management, gathering of leaves, seeds and fruit is stressful for some sensitive species such as *Aloe peglerae*, *Cola nitida* and *C. millenii* which are endemic to Africa and currently endangered. Studies suggest developing silvicultural techniques to improve domestication through *ex situ* cultivation in gardens and orchards (Chungu *et al.*, 2007; Lawin *et al.*, 2019; Pfab & Scholes, 2004; Savi *et al.*, 2019).

Barks and stems

Bark harvesting for medicinal purposes is widespread in Africa as a form of local and free medicine. *Julbernardia paniculate* and *Isoberlinia angolensis* are two species severely negatively affected by bark removal. Traditionally, there have been measures to reduce injuries to the tree. One form of local tree protection is to cover the wound site with mud, which protects the tree from wood deterioration and insect damage (Chungu *et al.*, 2007). In addition to practical measures, domestic legislation can also offer local protection. For example, *Warburgia salutaris* is endangered and overexploited in many regions and deemed threatened throughout its range. South Africa's environmental legislation now prohibits the harvesting of protected wild plants or plant parts (e.g., the bark and leaves of *Warburgia salutaris*) and recommends the use of alternative species (Rasethe, Semenya, & Maroyi, 2019; Senkoro *et al.*, 2019).

Harvesting bark to meet medicinal demands is becoming less sustainable for some species due to increasing demand. *Prunus africana* in Africa and The Himalayan yew *(Taxus wallichiana)* are greatly threatened with unsustainable harvest. Wild-gathering of barks of *Prunus africana* is no longer sustainable. The population has been declining over much of its geographical range in sub-Saharan Africa in recent decades. Only recently have existing standing crop inventories and scientifically based annual quotas being determined (Fashing, 2004; K. Stewart, 2009; K. M. Stewart, 2003). The Himalayan yew *(Taxus wallichiana)* is very slow growing species with poor natural regeneration. Most wild populations in Asia Pacific forests are threatened with extinction and are endangered in the Himalaya due to over-harvesting of their barks and leaves in combination with low seed production and germination. *In situ* conservation and management and artificial regeneration using efficient biotechnological tools have been proposed (Lanker *et al.*, 2010). Both of these species are included in the Convention on International Trade in Endangered Species of Wild Fauna and Flora to restrict international trade, with the intention to develop tools and methods for sustainable gathering or promote alternative source including cultivation.

Roots, Rhizome, Tuber and Bulbils

Gathering roots, rhizome, tuber and bulbils usually does harm to plants. It is fatal to dig out all the roots and tuber. Such gathering, if not managed properly, is unsustainable. Destructive overharvesting is the key threat to *Stemona tuberosa, Gymnadenia conopsea* in Asia and *Siphonochilus aethiopicus* and *Dioscorea bulbifera* in Africa driven by high market demand (G. Chen *et al.*, 2019; Ikiriza *et al.*, 2019; Kala, 2009; Shao *et al.*, 2017; Xego, Kambizi, & Nchu, 2016). The increasing demand on stems and roots coupled with non-sustainable harvesting methods has resulted in a substantial decline of *Cryptolepis sanguinolenta* in its wild populations in Africa forests. The development of domestication protocols has been suggested as one way to protect the species and decrease rates of decline (J. He, 2018; Kuniyal & Sundriyal, 2013; Woodhouse *et al.*, 2014).

Panax quinquefolius (American wild ginseng) is a highly valued wild root collected extensively in the United State of America s. Populations have declined significantly over

Box 3 12 The sustainable use of wild orchids in traditional Chinese medicine.

In China, orchids are traded for both ornamental and medicinal purposes (Hong Liu *et al.*, 2020). About a quarter of all Chinese wild orchid species are considered traditional Chinese medicine and market demands for some of them have been extremely high (H. Liu *et al.*, 2014). Wild populations of some traditional Chinese medicinal orchids, such as those in the genus of *Dendrobium*, have either been extirpated or reduced to small, isolated populations. Augmentations or reintroductions are required to bring these populations back to a healthy state.

Recognizing the issue of high demand on exhausted natural resources, the Chinese government has embraced the conservation intervention to increase supply by farming (Hong Liu, Gale, Cheuk, & Fischer, 2019) and has been very successful in encouraging massive shade house commercial cultivation of threatened traditional Chinese medicinal orchids. The total shadehouse products of *Dendrobium officinale*, one of the most used medicinal orchid species in China, was more than 6.4 billion United States dollars in 2011 (H. Liu *et al.*, 2014). However, it appears as though large commercial shadehouse cultivations have not alleviated pressure on wild populations. One reason for this is related to the public perception that cultivated products are considered to be less potent than wild harvested orchids, and so wild harvested products are considered to be of higher quality and are sold at premium prices (H. Liu *et al.*, 2014). In addition, orchids growing in industrial shade houses are subject to synthetic fertilizers and pesticides, which also make the cultivated product less desirable.

A semi-wild cultivation approach in which specimens are outplanted into native wooded areas specifically for harvesting has been implemented in a few places in southern China, such as Renhua County in northern Guangdong province, Xingyi County in Southwestern Guizhou province, and Leye County in northwestern Guangxi province. These areas are relatively undeveloped compared to the Pearl River Delta area in southern China and are within the native ranges of several medicinal Dendrobium orchids. These cultivation operations are a hybrid between commercial cultivation and population restoration because farmers can harvest certain number of stems (pseudobulbs) without killing the plants, and allow some plants to flower and fruit. Seeds produced from these plants are potential sources of population recovery and thus this form of outplanting is called "restoration-friendly cultivation" (H. Liu *et al.*, 2014). The market share of these semi-wild products is unknown.

This semi-wild or restoration-friendly cultivation approach has been suggested for other epiphytic orchids that are harvested for cultural and religious festivals and ceremonies in Latin American Countries (Tamara Ticktin *et al.*, 2020). For example, Mexico has more than 1,300 species of orchids (Hágsater *et al.*, 2015) and among these more than 300 orchid species in 90 genera were used for religious and cultural celebrations (Menchaca García, Lozano Rodríguez, & Sánchez Morales, 2012; Tamara Ticktin *et al.*, 2020). About a dozen of these species (e.g., *Laelia speciosa, Euchile karwinskii, Barkeria vanneriana*) are traded in high volumes legally and illegally (Tamara Ticktin *et al.*, 2020). There have been attempts to establish a rural community nursery system in the relevant areas, in which farmers were encouraged to plant these orchids in their backyard and adjacent community forests. The nurseries were then registered as Environmental Management Units (UMA, for their acronym in Spanish) (Menchaca García *et al.*, 2012). The nursery system was intended to promote sustainable harvesting in rural communities, as non-lethal harvesting can be done sustainably from the nurseries which then in turn allow wild populations to recover, as shown in population viability simulation models in Ticktin *et al.* (2020).

Semi-wild or restoration-friendly cultivation operations have positive impacts on sustainable use, but are not widespread in comparison with harvest quantities. Ecological and socioecological infrastructure needs to be developed and supported to achieve orchid conservation and support livelihoods (H. Liu *et al.*, 2014). For example, mass reproduction centers coordinated with farmers to deliver enough plants for semi-wild planting requires support (Menchaca García *et al.*, 2012). These centers can also provide technical support on growing and harvesting and marketing support. Restoration-friendly cultivation can directly facilitate the recovery of threatened species, encourage protection of natural forests, and benefit marginalized rural communities. However, it is unclear exactly what ecological growth conditions, harvesting regimes, and market conditions are suitable to achieve population restoration while generating enough income for participants, and what policies are needed to enable marginalized rural small holders to engage with the centers.

time in six northern states in the United States of America and harvest pressure is now restricting harvestable stocks. Annual wild ginseng harvests decreased from the high point in the late 1980s to early 1990s, but subsequently increased after 2005. Natural rates of population recovery are slow. Market prices for this species do seem to operate on a supply and demand logic such that quantities supplied are negatively related to prices, theoretically providing economic incentives for forest retention. A federal regulation has banned exports of roots from plants under five years old in effect since 1999. Management includes stewardship-oriented harvest restrictions such as delays in the opening of the permitted harvest season by two weeks, self-limits on harvest intensity, and planting ginseng seeds at the time of harvest (Burkhart & Jacobson, 2009; Burkhart, Jacobson, & Finley, 2012; Case, Flinn, Jancaitis, Alley, & Paxton, 2007; Frey, Chamberlain, & Prestemon, 2018; J. Schmidt, Cruse-Sanders, Chamberlain, Ferreira, & Young, 2019).

Some perennial wild plants can tolerate a certain degree of gathering activities. Populations of *Neopicrorhiza scrophulariiflora* were heavily exploited in one area of the alpine Himalayas but appear more resilient to extraction than other commercially exploited populations (S. Ghimire *et al.*, 2005; Poudeyal, Meilby, Shrestha, & Ghimire, 2019). In the American highlands, the local risk index for conservation status of *Oxalis adenophylla* was medium, driven by changes in its environment and not directly related to gathering. In fact, gathering of leaves and roots of this species is thought to promote its conservation through the understanding of its sensitivity to harvesting (Ochoa & Ladio, 2014).

The rhizome of *Hydrastis canadensis* (goldenseal) is widely harvested in America's woodlands and has been used for traditional medicine by native peoples. Regeneration time varies between populations, making it difficult to predict overall abundance. Late-summer and fall are the ideal periods when goldenseal rhizomes are traditionally gathered (Albrecht & McCarthy, 2006; Burkhart & Jacobson, 2009; D. L. Christensen & Gorchov, 2010).

In the majority of cases, proper management and predictable harvest volumes are required to ensure that root gathering meets the need of regeneration and renewal, but habitat conditions are also critical. For example, black cohosh *(Actaea racemose)* is highly responsive to harvest intensity in the United States of America. Low to moderate harvest intensities and/or longer recovery periods will be necessary for prolonged and sustainable harvests (Small *et al.*, 2011). A low harvest rate, for example 50% of mature plants every 10 years, may be sustainable for the harvest of osha or wild parsnip *(Ligusticum porter)* in America's highlands (Kindscher *et al.*, 2019). Harvesting of *Rheum acuminatum R. australe* and *Rhaponticum carthamoides* in central Nepal can be considered sustainable under optimal management. Predictable exploitable reserves and volume of harvesting, however, partly differ between species and strongly depends on habitat conditions (Nekratova & Shurupova, 2016; Rokaya, Münzbergová, & Dostálek, 2017). Management including wild cultivation can also protect habitats. Micro-propagation can aid in re-establishing plants in their natural habitats (Ikiriza *et al.*, 2019; Kala, 2009). Overexploitation for traditional medicine and health food sipplements, combined with habitat destruction, has resulted in the rapid decrease of *Dendrobium* sp. in Asia. However, epiphytic orchids planted in natural forests as part of *in situ* cultivation are facilitating more sustainable harvesting (H. Liu *et al.*, 2014, 2014; Shao *et al.*, 2017) **(Box 3.12)**.

3.3.2.3.6. Recreation

Many of the other uses covered throughout section 3.3.2 include some sort of recreational component. Only a few examples are provided here that stand out in terms of their recreational value.

A trend has been observed in recent years to promote forest management and sustainable use by combining gathering of wild algae, fungi and plants with non-extractive practices such as tourism. For example, mycological tourism is growing in popularity, often associated with amateur societies in North America and Europe, where people go mushroom gathering, harvest wild mushrooms and then identify them with the help of professionals (Barron, 2010). On one hand it is considered professional exploitation of wild resources, on the other hand it is a form of forest management (Jiménez-Ruiz, Thomé-Ortiz, Espinoza-Ortega, & Vizcarra Bordi, 2017). In fact, amateur mycological associations continue to grow and are considered a valuable resource by mycologists for everything from taxonomic assistance to data collection (Barron, 2011).

Many cultural services and values support recreational gathering of wild species. In the Northeastern United States of America, gathering wild edible huckleberries has been related to maintaining social relations, recreational use and commercial purposes (Carroll, Blatner, & Cohn, 2003). In Spain, while the gathering and consumption of wild edible plants is generally decreasing, there is an increase in the harvest of foods with high cultural value (Reyes-Garcia *et al.*, 2015). In Austria, interviews in 2008–2009 reveal the multiple motivations for gathering wild plants; women, older respondents and home gardeners gather wild plants more often for fun (Schunko, Grasser, & Vogl, 2015).

3.3.2.3.7 Science and education

Around the world, gathering wild specimens continues to generate information of scientific value. This includes dried plants and fungi for herbaria and fungaria, living plants and fungal cultures grown by botanical gardens and mycological institutes, and seeds stored in seed banks (Antonelli *et al.*, 2020; Paton *et al.*, 2020). The world's preserved botanical and mycological collections mostly date back to the late 1800s and early 1900s. There are 3,324 active herbaria in the world, containing 392,353,689 specimens. Norther America, Europe and temperate Asia (including Russia and China) have the highest number of herbaria (Antonelli *et al.*, 2020; Paton *et al.*, 2020; Pearce *et al.*, 2020). The Millennium Seed Bank Partnership conserves high-quality propagules. It has involved 96 countries and territories, and 32% of taxa (representing half of the collections) have at least one identified use for humans (U. Liu, Breman, Cossu, & Kenney, 2018).

Regarding live wild plants gathering, analysis of the PlantSearch database hosted by Botanic Gardens Conservation International indicates that 107,340 accepted species grow in botanic garden collections, representing 31% of vascular plant species. However, 93% of these species are held in temperate parts of the world. As a result, a temperate species has a 60% chance of being cultivated within the botanic garden network, whereas a tropical species has only a 25% chance.

Collection for scientific purposes, however, is on the decline (Heberling, Prather, & Tonsor, 2019), and there have been recent calls for more "holistic sampling" to maximize the usefulness of collections to protect individuals in the wild (Heberling *et al.*, 2019; U. Liu *et al.*, 2018). Good photographs, non-lethal harvest techniques, and the sharing of specimen information or molecular methods (Minteer, Collins, Love, & Puschendorf, 2014; D. Russo, Ancillotto, Hughes, Galimberti, & Mori, 2017) all represent good alternatives to lethal harvesting for scientific use. The Global Biodiversity Information Facility (GBIF) provides access to more than 1.4 billion records (including observations, preserved samples, fossils and living specimens) of all types of life on Earth in nearly 53,000 datasets supported by 1,600 institutions. The data of observation-based occurrences is surpassing the harvest of specimen-based occurrences in the Global Biodiversity Information Facility (Troudet, Vignes-Lebbe, Grandcolas, & Legendre, 2018). However, African countries, Central, South and Southeast Asian countries and East European countries have been poorly represented in harvest of vascular plants species aggregated in the Global Biodiversity Information Facility and data in the World Checklist of Vascular Plants are also poor (Antonelli *et al.*, 2020; Paton *et al.*, 2020).

Rocha *et al.* (2014) have argued that halting the collection of voucher specimens by scientists would be detrimental. Scientists believe that in order to describe the earth's biodiversity and understand wild species, museum collections should increase by 600%, while still being collected responsibly following best practices (Henen, 2016). Continued gathering would also support herbarium-based publications, which have dramatically increased in the past century (Heberling *et al.*, 2019). To that end, regulatory authorities could develop quotas for specimen harvest that are based on scientific guidelines (Maya & Gómez, 2016). Scientific gathering practices also face a series of economic and social pressures, including budget cuts and shortfalls in university and museum settings (Suarez & Tsutsui, 2004), high gathering costs (Enrique, Daniela, & Fernando, 2020), ethical considerations, and effects of regulations like the Convention on International Trade in Endangered Species of Wild Fauna and Flora for the cross-border exchange of specimens (Roberts & Solow, 2008). To promote scientific research on species conservation and materials sharing between scientists, the Convention on International Trade in Endangered Species of Wild Fauna and Flora established the registered scientific institute scheme, and encourages Parties to register scientific institutes. So far, 74 Parties have registered a total of 857 scientific institutions and individuals with the Secretariat of the Convention on International Trade in Endangered Species of Wild Fauna and Flora (Antonelli *et al.*, 2020; C. Williams *et al.*, 2020).

From Kew's dataset, there are more than 7,039 known species of edible wild plants, but only 417 (5.9%) are considered food crops by the FAO (Antonelli *et al.*, 2020; Ulian *et al.*, 2020). Crop wild relatives are sources of genetic diversity useful for developing more productive, nutritious and resilient crop varieties, and thus contribute to global food security. In 2016, the most important discovered species with potential for new food sources were 11 new Brazilian species of *Manihot* which are relatives of the highly valued food plant *Manihot esculenta* (cassava). *Manihot esculenta* is the third most important food after maize and rice, and it offers more food security than cereals. (Antonelli *et al.*, 2020; Castañeda-Álvarez *et al.*, 2016; Vincent *et al.*, 2013).

The Crop Wild Relatives Project (cwrdiversity.org) used the Harlan and de Wet (1971) gene pool concept to set up an inventory of globally important crop wild relatives' taxa for 173 priority crops. It contains 1667 taxa, divided between 37 families and 108 genera. The region with the highest number of priority crop wild relatives is Western Asia with 262 taxa, followed by China with 222 and Southeastern Europe with 181 (Vincent *et al.*, 2013). However, the diversity of crop wild relatives is poorly represented in gene banks. Kew's Millennium Seed Bank includes 688 crop wild relatives among its over 78,000 accessions. Over 70% of taxa are identified as high priority for further gathering in order to improve their representation in gene banks. The most critical gathering gaps occur in the Mediterranean and the Near East, Western and Southern Europe, Southeast and East Asia, and South America (Antonelli *et al.*, 2020; Castañeda-Álvarez *et al.*, 2016; Vincent *et al.*, 2013) **(Figure 3.45)**. A discussion of crop wild relatives is relevant for this assessment in relation to the sustainable use of collecting

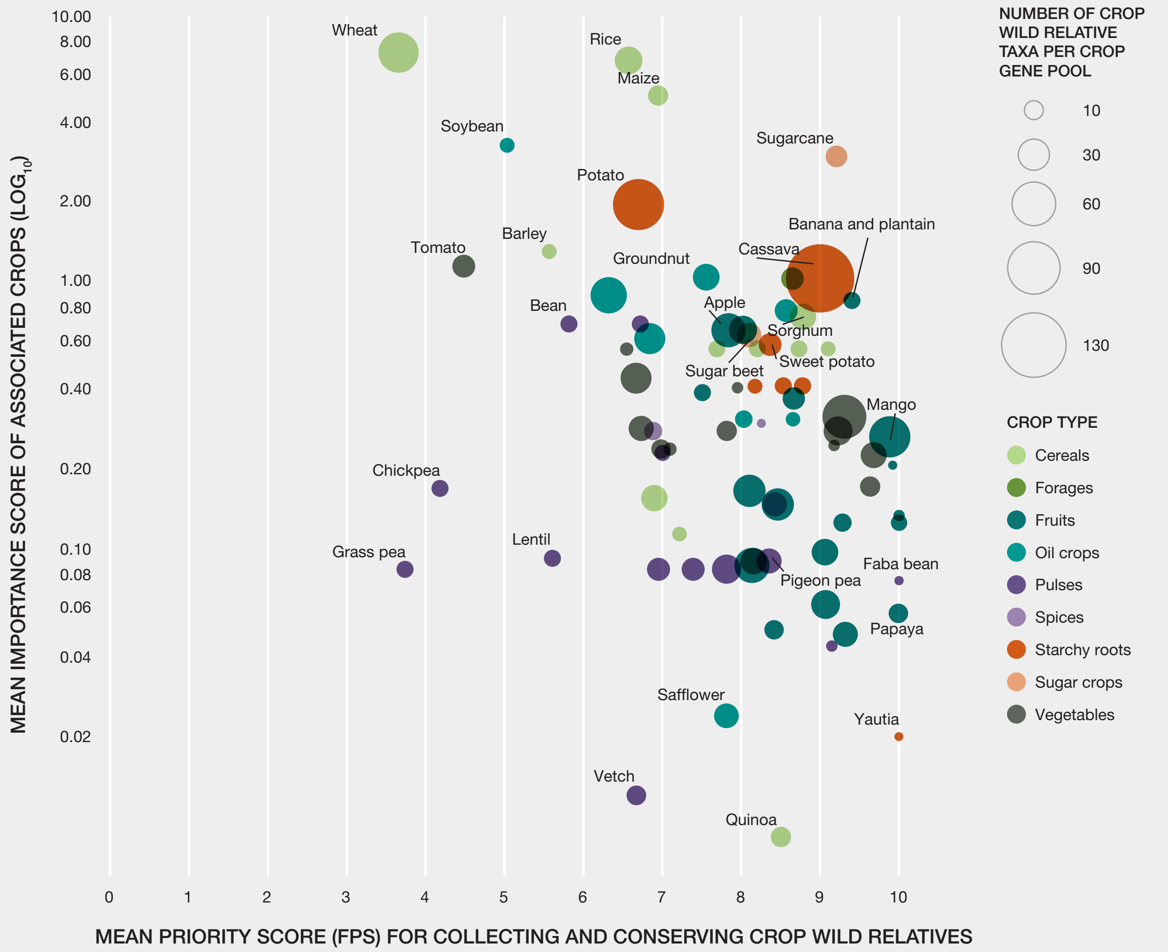


Figure 3 45 **Gathering priorities for crop wild relatives and the importance of associated crops.**

Crop types which occur in the upper portion of the graph are major global good stuffs; those on the far right are in greater need of the genetic diversity that crop wild relatives can provide. For example, of the three major global grain crops, wheat, rice, and corn, corn is in greatest need of genetic diversification from wild relatives in order to enhance food security Source: (Castañeda-Álvarez *et al.*, 2016) © 2016, Macmillan Publishers Limited under licencse CC BY-NC-SA 4.0.

specimens. However, analysis of the role of crop wild relatives in supporting crop diversity and providing genetic resources is beyond the scope of the current assessment.

3.3.2.3.8 Materials and shelter

Artificial materials have replaced many wild sources, but in some remote areas materials from wild species are more readily available and commonly used (**Box 3.13**; **Box 3.14**). Other than wood and bamboo, organic materials in tropical areas used for material and shelter include natural fibers, thatch, grass, reeds, sisal fiber, coir waste, elephant grass and straw (Bengtsson & Whitaker, 1988).

Sisal fibers are long natural fibers derived from Agave *(Agave sisalana)* leaves native to Mexico. In the 1960s the global production was 640 (metric) kt/year (UNIDO/CFC, 2005), but has since declined due to the rise of synthetic fibers. Sisal is grown mainly in Brazil, East Africa and China and has low requirements for fiber production and thus high potential for environmental sustainability (Broeren *et al.*, 2017). The FAO recommends natural fibers as future fibers, such as coir waste derived from coconut palm *(Cocos nucifera)*, Abaca extracted from the leaf sheath around the trunk of the abaca plant *(Musa textilis)* and Jute extracted from the bark of the white jute plant *(Corchorus capsularis)* (http://www.fao.org/economic/futurefibres). Similar to elephant grass *(Pennisetum purpureum)*, the source of these natural materials is shifting due to agriculture.

Palm leaves are an important source of roof thatch for rural communities in many parts of the tropics (Svenning & Macía, 2002). A total of 194 useful palm species and 2,395 different uses throughout northwest South America,

Box 3 13 Bamboo, a plant of many virtues.

Sources: (Laws, 2010; Paye, 2000).

There are over 1,400 species in the world, and they can thrive at high altitudes and low plains. Bamboo is one of the fastest-growing plants on the planet, and its influence has been widely felt: aside from rice, no other plant has played such as important role in the history as bamboo.

Besides being edible, it has medicinal, commercial and practical values: taken together, they yield more than 1,000 different products from their stems and leaves. Many uses of bamboo include preparation of waterproof coat and hat, each wrought out of leaves; agricultural implements; the fishing net, baskets of diverse shapes, arrows, paper and pens, grain-measures, wine-cups, water-ladles, chopsticks, tobacco-pipes, etc. In Asia, the bamboo symbolizes virtues, humanity, and resistance to hardship, and it has played an important role in Asian arts, including in ink drawing and painting.

Use of bamboo is the most common by indigenous and local communities of the world and every year people use over three billion cubic meters of wood worldwide to construct buildings, boats, furniture, and fences. Wood and steel have been the main materials for production in the modern construction industry. As deforestation intensifies, fast-growing bamboo is considered as an alternative to wood, easily used as an alternative in flooring, roofing, and even steel-reinforced buildings in Africa.

Box 3 14 Case study: neotropical palms.

Species or group

Palms are one of the critical elements in the floristic composition of tropical rainforests (Abensperg-Traun, 2009; Montufar & Pintaud, 2006). The Family includes 181 genera and c. 2,450 species distributed in the tropical region worldwide, with some species that extend into subtropical areas in both hemispheres (Baker & Dransfield, 2016). The South American continent hosts a wealth and diversity of palms and the Amazon contains 70% of the genus of palms of this region (Pintaud *et al.*, 2008).

Human uses and practices

Palms are renowned for their extraordinary usefulness for human communities (*Borchsenius*, n.d.), providing basic sustenance, construction materials, tools, and medicines. Palms are also often part of symbolic activities of indigenous communities (Macía *et al.*, 2011). They provide valuable income for rural inhabitants (Bernal *et al.*, 2011),(Kahn & Arana, 2008). However, at times unfavorable conditions and lack of oversight may lead to overexploitation, and possibly subsequent degradation of the local culture, the habitat, and the ecosystem. In South America, (Bernal *et al.*, 2011) documented harvesting and management practices for 96 palm species suggest that overexploitation is common without adequate management. Non-destructive management techniques include the harvest of fruits, leaves, fibers and other parts of the plant (in high palms, users climbing the stems, and a tool is used to cut the desired part), and the destructive ones involve cutting down the palms, which is necessary, for instance, for using stems in the manufacture of building materials or for extracting palm hearts (Bernal *et al.*, 2011).

Ecological responses across manifestations of biodiversity

The impacts of leaf harvesting for roofing purposes of houses and other buildings have been studied for the species *Lepidocaryum tenue* (Navarro, Galeano, & Bernal, 2011) and *Sabal mauritiiformis* (Andrade-Erazo & Galeano, 2015). The impacts due to the extraction of buds for the elaboration of handicrafts and other artifacts have been assessed for populations of *Astrocaryum standleyanum* (García, Galeano, Bernal, & Balslev, 2013), *Astrocaryum malybo* (García *et al.*, 2011), *Astrocaryum chambira* (García *et al.*, 2015) and *Copernicia tectorum* (Torres Romero, Galeano Garces, & Bernal, 2016). Studies on the effects of the palm heart crop have been made for *Euterpe oleracea* (Vallejo, Galeano, Bernal, & Zuidema, 2014) (Vallejo *et al.*, 2011). There is some research about harvesting of *Euterpe precatoria* fruits (Isaza, Galeano, & Bernal, 2014) and *Mauritia flexuosa* fruits (Sampaio, Schmidt, & Figueiredo, 2008).

Socioeconomic effects

Trade statistics are only well documented for species that are traded internationally, such as *Euterpe oleraceae* (açaí) of which Brazil is the leading supplier of palmetto and palm oil from this species (Brokamp *et al.*, 2011). However, for local communities, personal use and informal trade of palm products are part of their primary livelihoods, allowing income creation through the commercialization of raw materials or products traded in local and regional markets. The most commercialized palms in northwestern Amazon (Bolivia, Ecuador, Peru, and Colombia) are *Iriartea deltoidea* (timber), *Mauritia flexuosa* and *Oenocarpus bataua* (fruit, oil), *Lepidocaryum tenue* (thatch), *Ceroxylon* spp. (religious ornaments), *Phytelephas* spp. (Vegetable Ivory), *Astrocaryum* spp. (fiber, fruit) and *Euterpe* spp. (Palm hearts, fruit) (Brokamp *et al.*, 2011).

Palm fruits and oils have high nutritional value, and high economic value in international markets, however competitive technologies for the extraction and processing of raw materials must be developed (Brokamp *et al.*, 2011). Additionally, increasing economic sustainability would require strengthening value chains and the implementation of existing international and national legislation. This can

Box 3 14

be quite complicated in countries of South America where there are contradictions between national legislation and the rights of indigenous peoples, as well as the lack of technical and operational capacity of public institutions to control and verify compliance with the rules (de la Torre, Valencia, Altamirano, & Ravnborg, 2011). While implementation of standards that regulate the extraction of forest products is inconsistent, successful examples do exist (*Ceroxylon* spp.) and sustainable harvest is encouraged *(Lepidocaryum tenue)* (Brokamp *et al.*, 2011).

including Amazonia, Andes and Chocó have been documented (Macía *et al.*, 2011). In the Yucatan peninsula, leaves of xa'an palm trees (*Sabal yapa*, and *Sabal mexicana*) have been widely used for family homes. The palm is managed by Maya farmers through indigenous and local knowledge. When they clear a forest patch to grow maize, they spare palm trees, introduce them into home gardens and improve their growth. There are one or two harvest events per year and locals recommend leaving one or two leaves in each event. This traditional practice can stimulate palms to compensate for the effects of defoliation by producing new leaves (Martinez-Balleste, Martorell, & Caballero, 2008).

Although the harvest of *S. yapa* in natural systems has been sustainable for the last 90 years, the availability and quality of mature palm leaves is decreasing as agriculture becomes more intensive (Pulido & Caballero, 2006). In dry forests of northwest Mexico, the recruitment of *Brahea aculeata* may be threatened by the harvesting and livestock grazing. Therefore management, conservation and restoration of palms require careful consideration related to human and environmental factors (Lopez-Toledo, Horn, & Endress, 2011).

3.3.2.4 Emerging issues in gathering

The COVID-19 pandemic has had an enormous economic impact on people in many parts of the world, especially jeopardizing livelihoods among already economically marginalized communities. The Center for People and Forests (RECOFTC, 2020). Restrictions imposed due to COVID-19, such as limiting or prohibiting access to forests and the inability to manage land are having a noticeable (RECOFTC, 2020). In Nepal, commercial gathering of the highly-prized medicinal fungus species *Ophiocordyceps sinensis* **(Box 3.11)** was officially halted due to COVID-19. However, collectors, including many returning from India after losing their jobs, were forced to disobey orders issued by the District Disaster Management Committees and District Forest Offices to overcome humanitarian crises due to the sale of the fungus being the only source of household income (Singh, 2020). In other instances, locals returned to fallow land to cultivate seasonal crops to compensate for the lack of income from fungus harvest (Samiti, 2020). Overall, the pandemic not only reduced the livelihood opportunities for mountainous communities but also substantially affected generation of revenue due to the sale of the fungus for the Nepalese (NRB, 2015) has estimated that Nepal had generated about 4.7 million United States dollars in revenue from the fungus in 2014, presenting a significant source of income for residents. The loss of income from fungus harvesting during the pandemic has therefore most likely had negative financial effects that are as of yet undocumented.

During the pandemic, the biggest flow of "wildlife" in trade has involved wild plants, not animals. The volume of trade in herbal medicines is likely to increase across the world as an impact of the long-term economic crisis due to COVID-19. There have been reports around the use of herbal products as part of the COVID-19 response in Africa, Asia, Europe, South America, and the United States of America (Timoshyna, Ke, Yang, Liang, & Leaman, 2020). In the Asia-Pacific region, there has been an increase in the volume of trade in herbal products, such as those used in traditional Chinese medicine in China and neighboring countries, and Ayurveda in India and neighboring countries. It is anticipated that the number of gatherers of wild species for a variety of uses may increase as the long-term economic impacts due to COVID-19 continue to develop, especially in areas where wild harvesting correlates with high unemployment and poverty rates (Luo *et al.*, 2020; Timoshyna *et al.*, 2020). Communities where indigenous and local knowledge is well-maintained were able to quickly pivot towards gathering wild algae, fungi and plants to cover their food and medical needs as other sources of income fell away (Walters *et al.*, 2021). This underlines the importance of protecting indigenous and local knowledge and wild algae, fungi and plants as a social safety net (Pierce & Emery, 2005).

Increased engagement in gathering to meet subsistence needs and for recreational proposes has also observed in many locations worldwide, for example in Canada, Ukraine and in the United Kingdom (Deutsche Welle, 2020; SickKids, 2020; The New York Times, 2020). Along with increased gathering there have also been reports of increasing incidence of mushroom poisonings, as more people who had not previously engaged in gathering are taking to the forests.

3.3.3 Terrestrial animal harvesting

3.3.3.1 Introduction

Terrestrial animal harvesting is defined in Chapter 1 as the temporary or permanent removal from their habitat of animals (vertebrates and invertebrates) that spend some or all of their life cycle in terrestrial environments. This definition provides a higher-level classification for several practices relating to the direct use of wild animals, most notably hunting, which results in the death of the animal being harvested, but also live capture and removal from the habitat (e.g., for the pet trade), capture and release back into the environment such as can occur with harvest of animal fiber, and practices in which products of animals are removed without intended mortality (e.g., wild honey).

The conceptual framework for this assessment (Chapter 1) recognizes that practices such as terrestrial animal harvesting are influenced by the end use and the associated relational values with wild species. For example, hunting is an ancient practice and continues in many contemporary societies where people hunt to meet a range of nutritional, economic, medicinal, scientific, cultural and recreational needs (A. Fischer *et al.*, 2013; Storaas, Gundersen, Henriksen, & Andreassen, 2001). It is therefore not always possible or meaningful to assess terrestrial animal harvesting according to separate types of use, for example distinguishing the recreational aspect of hunting from other components that may be critical to an assessment of sustainable use. Even the taking of some part of the hunted animal as a memory, or 'trophy', almost never occurs on its own and needs to be considered in the context of all the other uses. Nevertheless, this section presents the evidence according to uses in order to maintain consistency throughout the chapter.

Studies relating to terrestrial animal harvesting often focus on activities for particular species, referring to a wide variety of animal species that are harvested under circumstances that range from abundant to threatened and for populations that are defined as wild, introduced or feral in this assessment (Chapter 1). Ungulates are commonly hunted in many countries and are the subject of the most scientific studies but a wide range of other species are also frequently harvested, such as birds, reptiles and invertebrates (Alves & van Vliet, 2018; Barboza, Lopes, Souto, Fernandes-Ferreira, & Alves, 2016; Coad *et al.*, 2019). For this reason, the assessment provides a summary of evidence for sustainable use of different taxonomic groups.

The assessment of sustainable use for terrestrial animals often needs to occur at a landscape level and should consider the spatial distribution and size of areas with and without use, as well as the population dynamics and dispersal behavior of the hunted species (Novaro, Redford, & Bodmer, 2000; Ohl-Schacherer *et al.*, 2007). For example, if hunting occurs only in some parts of the landscape but not in others (due to protected status, regulation, land-use practices, or placement of human settlements) it can result in heterogeneous hunting pressure across the landscape. In these circumstances, source-sink dynamics can mean that declining population productivity in hunted areas can be compensated by constant dispersal and replenishment from areas where hunting does not occur (Koster, 2008; Novaro *et al.*, 2000; Ohl-Schacherer *et al.*, 2007; Peres & Nascimento, 2006; van Vliet & Nasi, 2008). In addition, economic aspects of terrestrial animal harvest seldom involve the use of just one species but more typically involve the use of a variety of species occurring in the same landscape. This means that the assessment of sustainable use should include a more integrated approach, which considers not just the taxa that are being used or the reasons they are being used, but also the social-ecological systems in which the use of animals occurs (Di Minin *et al.*, 2021). These broader landscape and land-use aspects of use require the inclusion of additional dimensions in the assessment of sustainable use, such as governance systems and issues of land ownership, which have been identified as critical factors affecting sustainable use (Fargeot, Drouet-Hoguet, & Le Bel, 2017; Van Schuylenbergh, 2009). The evidence relating to these broader social-ecological issues is often lacking (Di Minin *et al.*, 2021) making it difficult to assess sustainable use across multiple dimensions.

In the rest of this section different types of terrestrial animal harvesting are explored. The sections are not meant to be mutually exclusive but are designed to consider sustainable use for different aspects of contemporary terrestrial animal harvesting. Scientific literature for this section was obtained through a systematic literature review following the IPBES protocol. Search results are available in the data management report for Chapter 3 systematic literature review at https://doi.org/10.5281/zenodo.6452651. Perish software (https://harzing.com/resources/publish-or-perish) and Google Scholar were used for the literature search with keywords: "sustainability" or "sustainable" + hunting type (for example, "trophy hunting" or "commercial hunting"). Or "trends" or "status" + hunting type. This search returned over 20,000 literature sources. To reduce this number a ranking/citation rate of publications was applied, and the first 50 publications were selected (as recommended in IPBES methodological guide). It should be noted that the obtained results were (1) geographically imbalanced (covering mostly certain regions of Africa or the United States of America); (2) sometimes quite old (an artifact of the methodology because older publications often have higher citation ranking); (3) Neither Perish or Google Scholar search for thematic reports. To overcome these imbalances experts supplemented the literature search from their own collections and those recommended during the external review processes. Invited contributing authors added literature for their sections following literatures searches and relying on their professional experience.

3.3.3.2 Uses

3.3.3.2.1 Ceremonial and cultural expression

Cultural and religious factors influence hunting practices in all regions of the world. Wild species are an important part of cultural life since various animal parts are used as adornments in ceremonies or as ornaments (e.g., feathers and fur) and tools (e.g., bones and teeth) in daily life (Pangau-Adam, Noske, & Muehlenberg, 2012). Hunting as a 'socio-cultural' phenomenon involves non-market values, symbolic and social capital, social status and impacts on good quality of life (see Chapter 4). Hunting supports social interaction and community, especially in terms of creating and maintaining bonds within one's social group, as well as benefits to physiological and psychological welfare for hunters (Bioeconomy.fi, 2017; A. Fischer *et al.*, 2013).

In many cultures across the world, hunting is associated with power, prestige and success, especially when the animals are killed in wild conditions. Europe has a diverse and complex legislative and regulatory hunting environment which includes many traditional elements (Higginbottom, 2004). In Scotland, deer stalking is part of a 150-year-old hunting culture, and continues to be one of the main activities of upland estates. Even where stalking is not commercially viable, it is a culturally important activity and has important bonding functions that help develop and reassure one's social status (MacMillan & Leitch, 2008). Similar functions are also observed in Sweden, where moose hunting teams are organized on a voluntary basis by local hunters' groups and landowners (Gunnarsdotter, 2007).

There are long traditions of bird hunting throughout Europe. However, the only readily available data on numbers of birds legally killed across the European Union are for derogations issued under the Birds Directive. This applies to four countries: France, Italy, Malta and Spain, in which 1.39 million individual birds (11,000 doves, 448,850 finches, 430,000 larks, 3,200 plovers, 200,000 starlings and 297,200 thrushes) are legally hunted each year under these derogations relating to "traditional practices" (Brochet *et al.*, 2016) (these statistics are reported again under the section on recreational hunting). In addition, very restricted derogations are allowed for capture of living finches in some countries, such as Malta and Spain. Whether directly related or not, numbers of migratory birds in the Mediterranean region have declined substantially, with one study estimating that there are 300 million fewer farmland birds in Europe today than in 1980, primarily as a result of agricultural intensification (BirdLife International, 2008).

Consuming meat of wild animals may also demonstrate wealth, prestige, and social standing in some cultures, whereas in others it may be a matter of choice, taste and options. In some urban areas, wild meat is a luxury good which is marketed to and adopted by young men to boost their professional and social status (Gangale, 2016). In Papua-New Guinea, it is a tradition among the Genyem that certain animals could be only hunted by clan leaders, while others could not be killed by hunters at certain times (e.g., when their wives were pregnant) (Pangau-Adam *et al.*, 2012). However, there is some evidence that traditional Genyem beliefs are breaking down as some species that were once considered taboo (e.g., cassowaries, certain birds-of-paradise) are now hunted (Pangau-Adam & Noske, 2010). Wild animals, mainly wild boars, are still occasionally killed for community festivals and religious ceremonies. When a large amount of meat is required for a cultural event, hunting is performed in groups and in more rural areas (Pattiselanno, 2006). Many hunters (91%) also target wild boars because the number of boar jaws they harvested was traditionally a sign of their social status.

Cultural values are considered to be important drivers of biodiversity conservation and sustainable use of wild animals (see Chapter 4). Taboos represent social norms and beliefs that protect species or places because of their cultural values. Taboos have had an important implication, for example, in relation to primate conservation (Infield *et al.*, 2018; Baker *et al.*, 2018). However, the protection of culturally valuable species rarely extends to other species or habitats (Schneider, 2018).

3.3.3.2.2 Decorative and aesthetic uses

The text in this section refers primarily to decorative and aesthetic use of wild animals, documented through formal trade data. Data are not available on informal or subsistence use of wild terrestrial animal species for these purposes. The skin of mammals is used commonly for gloves, shoes, belts, and watchbands. Over 4.6 million mammal skins from wild species were exported for commercial purposes over the period 1996–2010 and vast majority (>99%) were harvested in the wild (CITES, 2012). In Australia, the kangaroo skin industry generates 133 million United States dollars a year. In Peru, total annual value of the peccary-leather trade was estimated at 4,868,500 United States dollars of which only 1.5% was attributed to the rural sector (hunters), 11.1% to the urban sector (the national leather industry), and the majority went to the international leather industry (Bodmer & Lozano, 2001). Since 2007, the skin trade has been decreasing with the decrease in exports of fox (*Lycalopex* spp.) skins by Argentina (CITES, 2012).

Different parts of animals can be legally exported, and legal international trade has contributed to the recovery of some species. For example, skin of peccaries from Peru

(Bodmer & Lozano, 2001), kangaroo meat and animal skin from Australia (Boom *et al.*, 2012), skin of foxes (*Lycalopex* spp.) from South American countries (CITES, 2012), and crocodilian skins (Caldwell, 2017). Legal programs include economic incentives for people to tolerate the recovery of large predators (Fukuda, Webb, Edwards, Saalfeld, & Whitehead, 2020). Conversely, illegal international trade has contributed to the decline of many wild animal species worldwide (Pires & Moreto, 2016; ROUTES, 2020; TRAFFIC, 2008).

In Amazonia, from 1904 to 1969, an average 23.3 million wild mammals and reptiles representing at least 20 species were commercially hunted for their hides; averages of 13.9 million terrestrial mammals, 1.9 million aquatic and semiaquatic mammals, and 7.5 million reptiles (Antunes *et al.*, 2016). Hunted species included the manatee *(Trichechus inunguis)*; capybara *(Hydrochoerus hydrochaeris)*, ocelot *(Leopardus pardalis)*, margay *(Leopardus wiedii)*, jaguar *(Panthera onca)*, neotropical otter *(Lontra longicaudis)*, giant otter *(Pteronura brasiliensis)*, collared peccary *(Pecari tajacu)*, white-lipped peccary *(Tayassu pecari)*, red brocket deer *(Mazama americana)*, black caiman *(Melanosuchus niger)*, common agouti (*Dasyprocta* spp.), Amazonian brocket deer *(Mazama nemorivaga)*, tapir *(Tapirus terrestris)*, iguana *(Iguana iguana)*, tegu lizard *(Tupinambis teguixin)*, caiman lizard *(Dracaena guianensis)*, boa *(Boa constrictor)*, anaconda *(Eunectes murinus)*, and spectacled caiman *(Caiman crocodilus)*. The commercial exploitation of animal hides in the 20th century had led to population collapse for the large-bodied aquatic wild species, signaling the possibility of an "empty river" phenomenon. At the same time, various sustainability indices have shown different results suggesting that drivers other than hunting and complexity of applied models must be taken into account in assessing sustainability (Chapters 2 and 4).

Several species of crocodile are harvested for the leather and fashion industry, with over 5.2 million crocodilian skins reported in trade between 2013–2015 (Caldwell, 2017). The majority of crocodilian skins in trade are from captive bred stock, although many were originally sourced from legal wild egg ranching programs. In many countries, indigenous and local people benefit through the payment of royalties for eggs, and/or employment through the farm supply chain (Fukuda *et al.*, 2020; Joanen, Merchant, Griffith, Linscombe, & Guidry, 2021). As a result, species such as the saltwater crocodile *(Crocodylus porosus)* and American alligator *(Alligator mississippiensis)* have recovered from unregulated hunting in the 1960s and 1970s, back to pre-exploitation levels. The economic value generated through the leather industry has enabled tolerance of this recovery and protection of habitat. The sustainable use of alligators in the United States of America generates more than 100 million United States dollars annually at the raw product level (R Elsey, Woodward, & Balaguera-Reina, 2019).

Feathers are used as ornaments in many cultures. Amazonian indigenous people have a very deep knowledge of birds. They invented the technique of tapirage, which is making the feathers change color on a live bird. They often tame birds that they keep as pets or for their feathers. In some countries of Amazonia there are conflicts with conservation laws that do not allow people to kill birds. In Guiana Amazonian Park (Guyane) a program is being developed to harvest feathers in zoos instead of killing birds to make headdresses.

3.3.3.2.3 Food and beverage

Millions of animals are killed every year in Africa, Asia, and the Amazon for subsistence hunting and the wild meat trade **(Table 3.12)**. The most frequently hunted taxonomic groups in most studies are ungulates, followed by rodents. Large mammals alone comprised 55-75% of total wild meat biomass extracted annually **(Table 3.12)**. Note that given these figures, **Table 3.12** focuses primarily on wild meat from mammals. Wild meat from bird, amphibians and reptiles are discussed in detail elsewhere.

In West and Central Africa, wild meat consumption has increased drastically in recent decades (Wilkie, Bennett, Peres, & Cunningham, 2011). Wild meat comprised 62.2% of the total animal protein consumed by families in Papua New Guinea (Pangau-Adam *et al.*, 2012). Estimates of wild meat consumption differs greatly – with global estimates of more than 5 million tons a year (Kanagavel, Parvathy, Nameer, & Raghavan, 2016) to separate regional estimates of 4.6 million tons in the Congo Basin and 1.3 million tons a year in the Amazon (Rosie Cooney, Roe, Dublin, & Booker, 2018). Wild meat comprised 62.2% of the total animal protein consumed by families in Papua New Guinea (Pangau-Adam *et al.*, 2012). For scale of comparison, it is worth noting that if global wild meat consumption is roughly 5 million tons a year, this is only equivalent to approximately half of the European Union's beef production (Fa *et al.*, 2002; Nasi, Taber, & Van Vliet, 2011).

In semi-arid regions (South America, Africa, Asia), mammal meat is crucial for the nutritional well-being of many human communities especially because the availability of fish or other sources of protein are limited (Barboza *et al.*, 2016; da Silva Santos *et al.*, 2019). In this ecoregion, wild meat can be especially important during the frequent drought periods, a typical phenomenon in these areas, when crops are scarce and domestic animals may die because of starvation and dehydration (Barboza *et al.*, 2016). Within a vast savanna ecosystem, about 50 million people depend to varying extents on wild species for their food security and daily subsistence (Olivero *et al.*, 2016). A significant part of the population, often poor and rural, hunts for their own consumption and as a primary source of income by supplying food to more or less distant consumption

Table 3 12 **Domestic consumption rates of wild meat from subsistence hunting.**

Regions and countries reported based on available literature.

Region/country	Harvest	Main target species or taxonomic group	Share of large animals	Reference
Tropical regions of Africa (Cameroon, Central African Republic, Republic of Congo, Democratic Republic of Congo, Equatorial Guinea, Gabon, Ghana)	340 – 84,093 kg/year per site, 16,000 kg per site, on average	ungulates (47%), rodents (37%)	22% of carcasses to total kills, but 55% of total wild meat biomass extracted per year	(Fa, Peres, & Meeuwig, 2002)
Peru	54-255 inds/100 km^2, 1605 – 4581 kg/ 100 km^2	White-lipped peccary *(Tayassu pecari)*, collared peccary *(Tayassu tajacu)*, lowland tapir *(Tapirus terrestris)*, brown capuchin *(Cebus albifrons)*, howler monkey *(Alouatta seniculus)*, paca *(Agouti paca)*, agouti *(Dasyprocta fuliginosa)*	Large mammals comprised 78% of the estimated biommass of all hunting animals	(Bodmer & Lozano, 2001)
Eastern half of Papua-New Guinea, Indonesia	Between 4 and 8 million individuals	Wild pig, cassowaries, cuscus, and bandicoots	Large mammals comprised 58% of the estimated biomass of all harvested animals	Cuthbert, 2010; Mack & West, 2005; Richards, S. J. & Suryadi, S., 2000)
Papua (the western half of Papua-New Guinea), Indonesia	Wild meat comprised 62.2% of the total animal protein consumed by families	Wild pig, rusa deer, bandicoots	Large mammals comprised 75% of estimated biomass of all harvested animals	(Pangau-Adam *et al.*, 2012)
India	India population ate an average of 0.158 kg of meat per month	Barking deer, Wild pig, Asiatic black bear, Sambar, Serow, Assamese macaque, Goral	Large mammals comprised 70% of the estimated biomass of all harvested animals	(Karanth, Nichols, Karanth, Hines, & Christensen, 2010)
Vietnam	More than 58% of Vietnam population ate 1 kg of meat per month	Wild Pig, soft-shelled turtle, Bear, Snake, Civet	Large mammals comprised 50% of the estimated biomass of all harvested animals	(E.L. Bennett & Rao, 2002)
Amazonian forests	10,691 tons of wild meat might be consumed annually in Amazonia, the equivalent of 6.49 kg per person per year	Mammals, reptiles, and birds	38% of species more than 1 kg	(Bahuchet & de Garine, 1990; H. El Bizri *et al.*, 2020; Fa & Peres, 2001; Noss, 1998)
Tropical forests	177-358.4 kg/km^2/ year on average	The main taxa represented are primates (ungulates, rodents, and carnivores	High harvest rates of largebodied diurnal animals	(Fa *et al.*, 2002)

centers. Even before commercial sale of meat, heads, legs and intestines of harvested animals are typically removed (~1–5 kg per animal) for family consumption prior to transporting the prime meat cuts to the market (Pangau-Adam *et al.*, 2012).

Profound social-economic changes (the introduction of a cash market economy through globalization, combined with rapid urban and infrastructure development) have resulted in marked shifts in hunting practices of many indigenous and local communities. The nature of hunting has changed from local-level subsistence hunting towards more intensive commercial hunting for wild meat trade (Pangau-Adam *et al.*, 2012). For many rural families, wild meat trade is the main source of cash income, providing access to modern services and basic necessities such as medicines, energy and education (Abernethy, Maisels, & White, 2016). However, increases in commercial harvesting of wild species threatens the traditional lifestyles of indigenous populations through the weakening or loss of traditional laws and taboos, which may push hunting activities towards becoming unsustainable (Pangau-Adam *et al.*, 2012). In tropical forests, harvesting of wild meat by forest dwellers has drastically increased recently due to large numbers of urban consumers, advances in hunting technology, scarcity of alternative sources of protein, and individual food preferences (Fa & Brown, 2009; Groom, Meffe, & Carroll, 2006). In competition with these families, professional hunters and/or traders organize illegal networks to transport and sell the products (Van Schuylenbergh, 2009).

Reptiles and amphibians also serve as an important source of protein for human populations (Coad *et al.*, 2019). Of all reptiles, turtles and tortoise species (chelonians) are most heavily harvested for human consumption (Alves, Gonçalves, & Vieira, 2012; Pezzuti, Lima, da Silva, & Begossi, 2010). Live animals (e.g., turtles, tortoises, and lizards) as well as processed, dried, and frozen meat (e.g., pangolin) are commonly traded into food markets for consumption (see routespartnership.org). In South America, the giant Amazon River turtle *(Podocnemis expansa)*, the largest South American river turtle, is one of the most consumed species. Caiman meat (as other crocodilians) is a product that is increasing in acceptance in the world food market. Currently there is a supply of meat from many managed areas in Argentina, Bolivia, Brazil and the United States of America (Piña, Lucero, Simoncini, Peterson, & Tavella, 2017). Crocodile and alligator meat is considered a delicacy (Huchzermeyer, 2003a, 2003b), and it is particularly consumed in Australia, South Africa, Thailand, Ethiopia, Cuba, and in some regions of the United States of America (Hoffman & Cawthorn, 2012). The consumption of snakes is generally opportunistic, but in Asian countries and West Africa, these animals are important sources of meat (S. E. Brooks, Allison, Gill, & Reynolds, 2010; Hoffman & Cawthorn, 2012). Although amphibians are consumed on a smaller scale than vertebrates, Mohneke *et al.* (2009) highlight that at least 32 amphibians (3 *Urodela* spp., 29 *Anura* spp.) are used as food.

Many investigations showed that the crucial factor explaining target species preferences is the anticipated benefits from hunting the largest-bodied animals. Usually opportunistically hunted small and medium-sized game are consumed by the hunters themselves, especially in low-income countries throughout Africa, Asia, South America and Eastern Europe (Fischer *et al.*, 2013). In contrast, big game provides a greater return for the energy invested in hunting, more meat for consumption, and significant revenue for hunters' households (Coad *et al.*, 2013; Constantino, 2016; de Albuquerque *et al.*, 2012; D. J. Ingram *et al.*, 2015; P. Lindsey, Balme, Booth, & Midlane, 2012; Maisels, Keming, Kemei, & Toh, 2001; Nasi *et al.*, 2008; Pangau-Adam *et al.*, 2012; Redmond, 2006) with a few exceptions due to underdeveloped markets (MacMillan & Leitch, 2008). It should be noted that while it may seem that hunting larger animals is energetically more efficient, large game are infrequently acquired (there are higher number of unsuccessful days) and storage is often a problem. Furthermore, it is typically a riskier activity. In some traditional small band societies (e.g., the San, the Hadza, the Ache, various Native American and First Nation peoples) small game and plant resources are more regularly gathered as primary sources of protein and daily nutrition (Hawkes, O'Connell, & Blurton Jones, 2001).

Over-harvesting may take place due to the lack of knowledge or monitoring, lack of sufficient regulation, or lack of political will and prioritization of conservation. In these cases varying degrees of hunting pressure often result in faunal biomass collapses, mainly through declines of large-bodied species with low intrinsic rates of population increase, as was the case in Oceania (Pangau-Adam *et al.*, 2012), Africa (Coad *et al.*, 2019; Gill, Fa, Rowcliffe, & Kümpel, 2012; Groom *et al.*, 2006; Ingram *et al.*, 2015; Milner-Gulland & Bennett, 2003; Van Vliet, Milner-Gulland, Bousquet, Saqalli, & Nasi, 2010; van Vliet, Muhindo, Kambale Nyumu, Mushagalusa, & Nasi, 2018; van Vliet & Nasi, 2008; Weinbaum, Brashares, Golden, & Getz, 2013), and Asia (Bennett & Rao, 2002; Karanth, Jain, & Mariyam, 2017). As populations of larger animal decline, the time and effort required to catch these large species will eventually outweigh the potential gain, leading hunters to shift to target mid-size and small species (Jerozolimski & Peres, 2003). Throughout this process, the largest species of a multispecies hunt will continue to be opportunistically captured whenever possible, preventing large species recovery even when the primary target is now a smaller species (Robinson & Bennett, 2004).

Unsustainable hunting for human consumption is only one factor affecting declines in mammalian species **(Figure 3.46)**, (Alves, 2012; Chapman & Peres, 2001; Dirzo *et al.*, 2014; IUCN, 2016; Ripple *et al.*, 2014; Ripple *et al.*, 2016), but is especially prevalent in tropical environments (Coad *et al.*, 2019; Fa *et al.*, 2002; Fa, Ryan, & Bell, 2005; Weinbaum *et al.*, 2013). Overall extraction rates in the Congo Basin, for example, were calculated on the basis of extraction–production models to be as much as six times greater than the maximum sustainable rate (Fa *et al.*, 2002). Fossil evidence suggests that hunting has contributed to the local extinction of several species of larger mammals in New Guinea in the past (Flannery, 2000). Ripple *et al.* (2016) found that 301 mammals are threatened by hunting globally: 113 species in Southeast Asia (13% of all threatened mammals are east of India and south of China) and 61 in the rest of Asia (7%); 91 in Africa (8%); 38 in Latin America (3%); and 32 in Oceania (7%). Unsustainable hunting has been identified as a threat for 1,341 wild mammal species assessed by the International Union for Conservation of Nature, including 669 species that were assessed as threatened (IUCN Red list 2021). Nearly 20% of the International Union for Conservation of Nature Red List's threatened and near threatened species are directly linked to hunting (Maxwell, Fuller, Brooks, & Watson, 2016). Eleven of the 14 species of tree-kangaroos (*Dendrolagus* spp.), most of them endemic to New Guinea, and two of the three cassowaries (Casuariidae; 25–60 kg), are now considered threatened by, or vulnerable to extinction, principally due to

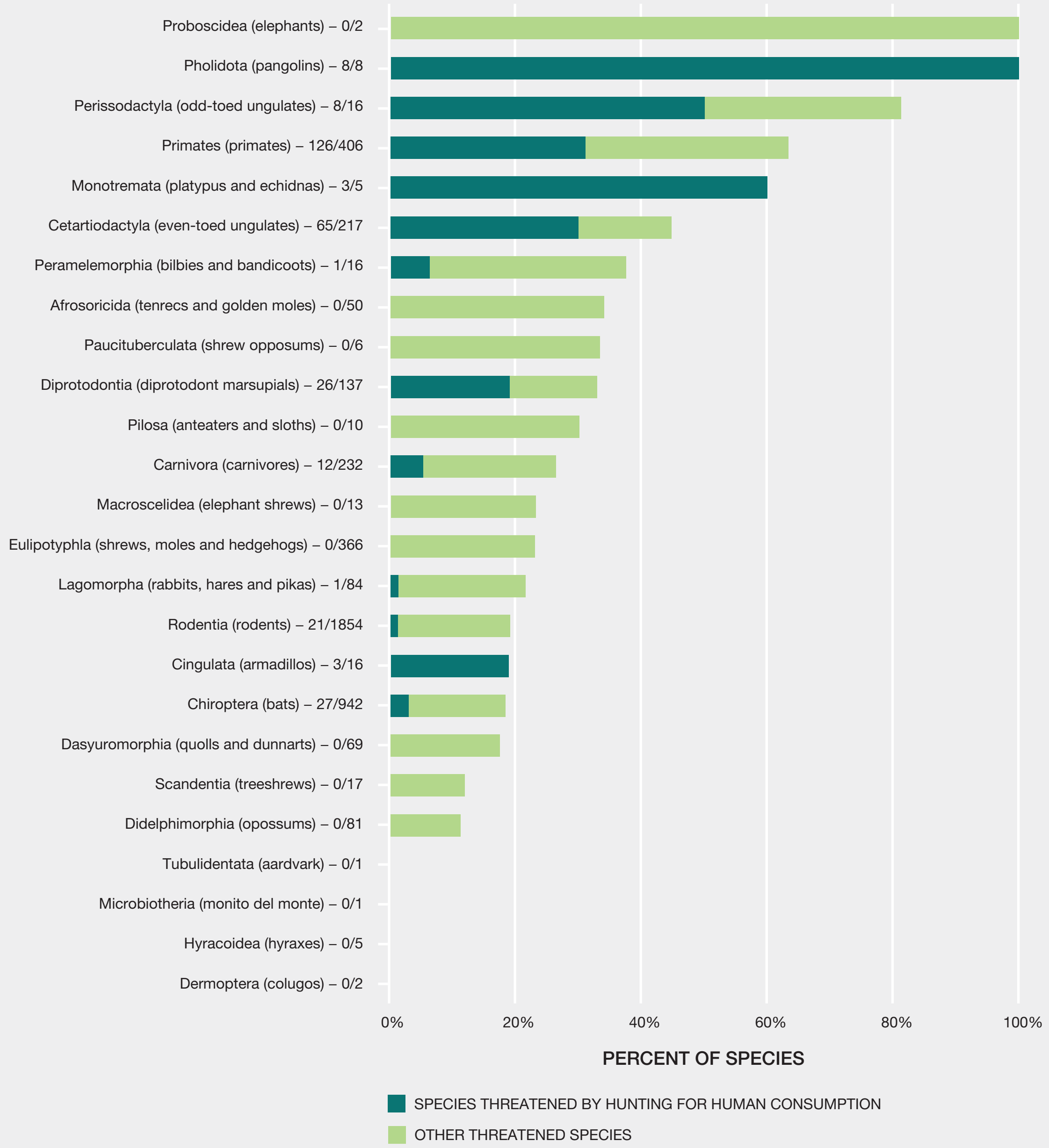


Figure 3 46 **The percentage of species threatened by hunting for human consumption and other threatened species in each mammalian order.**

Values on the x-axis refer to the percentage of species out of all mammal species in each order. The category "Other threatened species" consists of the other threatened mammal species where hunting for consumption is not a primary or major threat. Horizontal bars are sorted from highest to lowest total percentage of threatened species in each order. Numbers on the y-axis after the order names are the number of species threatened by hunting followed by the total number of species in the order. The order Notoryctemorphia (marsupial moles) was omitted as it contains only data-deficient species. Source: (Ripple *et al.*, 2016) under license CC BY 4.0.

hunting (IUCN, 2016; Stattersfield, Crosby, Long, Wege, & Rayner, 1998). Increasing commercial demand, availability of sales markets (Pangau-Adam *et al.*, 2012), lack of adequate monitoring and enforcement by the government (Pangau-Adam *et al.*, 2012), poaching and illegal trade (Pangau-Adam & Noske, 2010) further complicate this process.

The sustainability of wild meat hunting is increasingly driven by social-economic changes, recreation, entertainment, trade, and trafficking, rather than take-off for subsistence. These drivers are discussed in detail in Chapter 4. In the absence of effective governance, many experts continue to focus primarily on total offtake from an area, suggesting that as long as hunting is profitable, the largest animals will be driven to local extinction by hunters (Branch *et al.*, 2013; Harrison *et al.*, 2016; Lindsey, Alexander, Balme, Midlane, & Craig, 2012).

Wild meat consumption **(Table 3.12)** and trade carry health risks related to the transmission of zoonotic diseases to humans through handling (e.g., hunters, middle market distributors, and sellers) or consumption of wild meat. This is especially of concern at traditional food markets when wild animals are caged, and then slaughtered and dressed in close proximity to the public (OIE, WHO, & UNEP, 2021). The emergence of new infectious diseases, particularly zoonoses (derived from animals), is increasing.

With regards to commercial demand for wild meat, there is growing demand in cities stimulated by migration of rural peoples to urban landscapes (Bennett *et al.*, 2007). There is evidence that the commercial trade of wild meat has heavily increased offtakes in West and Central Africa because of the higher prices likely to be paid by urban dwellers, with the situation anticipated to worsen as populations continue to rise and become more urbanized. A similar trend is apparent in Eastern and Southern Africa, where increasing urbanization is associated with a growing consumption of wild meat resources (Barnett, 2000; Cowlishaw, Mendelson, & Rowcliffe, 2004; Peter Lindsey & Bento, 2012). The demand of game meat in many European Union countries is also growing due to beliefs that it is a more ecological and ethical choice consistent with ideas of the green transition. The demand for wild meat in many developed countries among the diaspora communities from developing countries has also created new demand for international trade in wild meat (Chaber, Allebone-Webb, Lignereux, Cunningham, & Rowcliffe, 2010).

Economic incentives and unclear rules and regulations may be leading to additional commercial hunting on indigenous lands (Fischer *et al.*, 2013; Pangau-Adam *et al.*, 2012). In Papua, Indonesia, the anticipated financial gain for a hunter from the sale of three individual wild animals (35–50 United States dollars each) is approximately equivalent to the monthly salary of a locally employed permanent worker (Pangau-Adam *et al.*, 2012). In Central Amazon, Brazil, wild species hunting and consumption are driven by many factors such as source of income, taste preference, culture, lack of alternative meat, meat price, and wealth. The relative importance of these factors varies from place to place (Chaves, Valle, Tavares, Morcatty, & Wilcove, 2021).

Amphibians and Reptiles

Amphibians and reptiles were historically harvested and traded for different reasons. For example, tortoises, large freshwater turtles, sea turtles, and crocodilians were used as an important source of protein for human populations around the world (Klemens & Thorbjarnarson, 1995; Pritchard, P.C.H., 1996; Schlaepfer, Hoover, & Dodd, 2005). Exploitation of these species for food is heaviest in the tropical and sub-tropical regions, but also occurs in temperate areas also. Amazonian markets, for example, include the domestic consumption of wild meat and turtle eggs and the use of crocodile parts and products in the international leather industry. In examples such as these the mixed-use nature of terrestrial animal harvesting is apparent: where meat consumption is a by-product of the commercial skin harvest of crocodilians, snakes, and lizards (Gorzula, 1996; Schlaepfer, Hoover, & Dodd, 2005).

The United States of America plays a major role in the international trade of wild amphibians and reptiles. During 1998–2002 in the United States of America alone, 14.7 million wild-caught whole amphibians, 5.2 million kg of wild-caught amphibians and 18.4 million wild-caught reptile parts and products were imported, and 26 million wild-caught whole reptiles were exported (Schlaepfer, Hoover, & Dodd, 2005). The crocodilian harvest programs in the United States of America (alligator) and Australia (saltwater crocodile) are highly regulated and monitored, with a coordinated system of permits, licenses, and rigorous tagging and export requirements (Elsey *et al.*, 2019; Fukuda *et al.*, 2020; Joanen *et al.*, 2021). More than 50% of all traded individuals of reptiles had no species-specific identification, making species-based regulation especially difficult without extensive genetic testing, which is temporally and financially unrealistic. Crocodilian meat is particularly favoured in Southeast Asia. The top species traded for meat are *C. niloticus* and *C. siamensis*, with trade peaking annually in 2006 at 1000 tonnes (Caldwell, 2017).

The most commonly traded species of amphibians and reptiles are abundant, widely distributed, and have long histories of sustaining use and trade, with varying degrees of regulation matched to their life history parameters. A species with a large range, high density, and high reproduction rate, for example, may be able to sustain a relatively large harvest. In contrast, species with restricted ranges, high levels of endemism (e.g., small island species), or life-history strategies that depend on high adult survivorship like many turtle and tortoise species (e.g. Heppell, 1998), could be detrimentally affected by relatively low harvest rates. Many amphibian and reptile species aggregate in small areas during breeding or hibernation, making them particularly vulnerable to intensive harvest efforts during that period (Klemens & Thorbjarnarson, 1995; Schlaepfer, Hoover, & Dodd, 2005).

Frog meat is considered a delicacy in many countries. The FAO has estimated the worldwide production of frog legs at 80,000 metric tons annually (FAO, 2012a). In Europe, there are 4600 tons of frog meat imported per year, corresponding to c.a. 46 million frogs, mainly coming from Indonesia and Vietnam, where they are predominantly harvested from the wild (Warkentin, Bickford, Sodhi, & Bradshaw, 2009). Human populations from Southeast Asia are estimated to be the largest producers and consumers of amphibians worldwide, even if there is a lack of proper evaluation for comparative purposes (Warkentin *et al.*, 2009).

In his book "The culinary herpetologist", Liner (2005) cites cooking recipes based on 26 salamander and 193 frog species; only a few of these edible species are consumed in large quantities. At the same time, edible amphibian populations are declining worldwide and humans have already faced the risk of losing this food source due to the overexploitation of animals harvested from nature (Carpenter *et al.*, 2007; Carpenter, Andreone, Moore, & Griffiths, 2014). India, followed by Pakistan and Bangladesh, banned the export of frogs in the early 1980s (Fugler, 1985). More recently, Turkish authorities have banned frog hunting in some provinces and advocated the promotion of frog farming (Şereflişan & Alkaya, 2016). Frog farming is already quite extensive in Indonesia, where many exotic and invasive species are harvested for international trade. Indonesian exports of frogs were 28 tons per year in 1969 and increased to 5600 tons in 1992 before decreasing to about 3800 tons in the early 2000s (Kusrini & Alford, 2006). Sustainable frog farming is lagging behind in major consumer countries, the first frog farm in France opened only in 2009.

Unlike in Indonesia, in Africa frogs are mainly used for local consumption and local trading. A long-standing tradition of frog hunting exists in the Lake Chad basin that relies on large populations of grassland frogs *(Ptychadena trinodis)*, edible bullfrogs *(Pyxicephalus adspersus)*, African tiger frogs *(Hoplobatrachus occipitalis)* and the marbled shovelnose frog *(Hemisus marmoratus)* (Seignobos, 2014). In West African countries, six species of frogs are among the most consumed and sold frog species (Mohneke, 2011; Mohneke, Onadeko, Petersen, & Rödel, 2010). Studies carried out in Benin and Nigeria showed that between both countries and over 2.7 million frogs are harvested annually for cross-border trade (Mohneke, 2011). In Central Africa, goliath frog *(Conraua goliath)* and slippery frog *(Conraua robusta)* are heavily harvested from the wild and sold in local wild meat markets in Cameroon (Gonwouo & Rödel, 2008; Herrmann, Babbitt, Baber, & Congalton, 2005). Similarly, frog species harvested from the wild contribute to the local supply chain including markets and restaurants in the Democratic Republic of Congo (Sandra Altherr, Goyenechea, & Schubert, 2011). Large tadpoles of endemic species such as *Conraua* sp., *Trichobatrachus* sp. and *Astylosternus* sp. are also harvested and traded for consumption (Gonwouo & Rödel, 2008; Mohneke, 2011). Despite the importance of these small wild animals, assessments of the value chains of which they are a part are scant, especially in many Central Africa regions.

Edible Insects

There are a considerable number of reports on the need for forest conservation using edible insects. They are important sources of food in arid and semi-arid areas of Africa and in the great sandy deserts of Australia (Yen, 2009). Traditional consumption of edible insects and small terrestrial invertebrates is common in one third of the world's population, mainly in Asian, African, Central American and South American cultures. Globally, more than two thousand identified arthropods are eaten (Arnold van Huis, 2018). Over 500 species of edible insects are reported for Mexico (Ramos-Elorduy, Pino-Moreno, & Martínez-Camacho, 2012) and 324 species of insects from 11 orders are documented as either edible or associated with entomophagy in China. People also feed insects to livestock and indirectly consume them (Feng *et al.*, 2018). People throughout higher income countries in Europe and North America are contemplating using of edible insects as an alternative, more sustainable source of protein than animals (Mlcek, Rop, Borkovcova, & Bednarova, 2014) **(Figure 3.47)**.

Globally 92% of insect species used by people are harvested from the wild (Alan L. Yen, 2015). Edible insects are often harvested by women and minority groups (A. van Huis & Oonincx, 2017; Arnold van Huis, 2018; Arnold van Huis *et al.*, 2013), but not exclusively. Insects are often harvested by hand. Some examples of harvest techniques are included here. The most common technique to harvest swarming termites and edible grasshoppers *(Ruspolia differens)* is using light sources at night. In Central African countries, women listen to trunks of the palm trees to check whether larvae of the palm weevil (*Rhynchophorus* sp.) are ready to be harvested (van Huis & Oonincx, 2017; van Huis, 2018). This also happens in Colombia, but it is usually the men who search and find the larvae in palm trees (*Oenocarpus bataua, Oenocarpus bacaba* in most cases). In many cases they cut down the palm (Mesa & Galeano, 2013) to harvest the insects. In the Venezuelan Amazon, the Jöti people manipulate *Oenocarpus bacaba* palms in order to increase abundance of their favorite palm weevil, *Rhynchophorus palmarum* (Choo, Zent, & Simpson, 2009). In the Asia Pacific, tarantulas (*Haplopelma* sp.) are harvested out of tropical forests. Yen and Ro (2013) observed that skilled spider hunters are able to harvest several hundred spiders a day, although how this is done is undocumented. Only female spiders are cooked and eaten. This may be because the females are larger. The income from selling spiders for food and medicine is substantial enough that this can be considered an important subsistence practice. Although reports suggest a decline in the population around Skun and other provinces is observed, direct causality from human harvesting has not been proven. There is little additional information available on the biology or population status of this species.

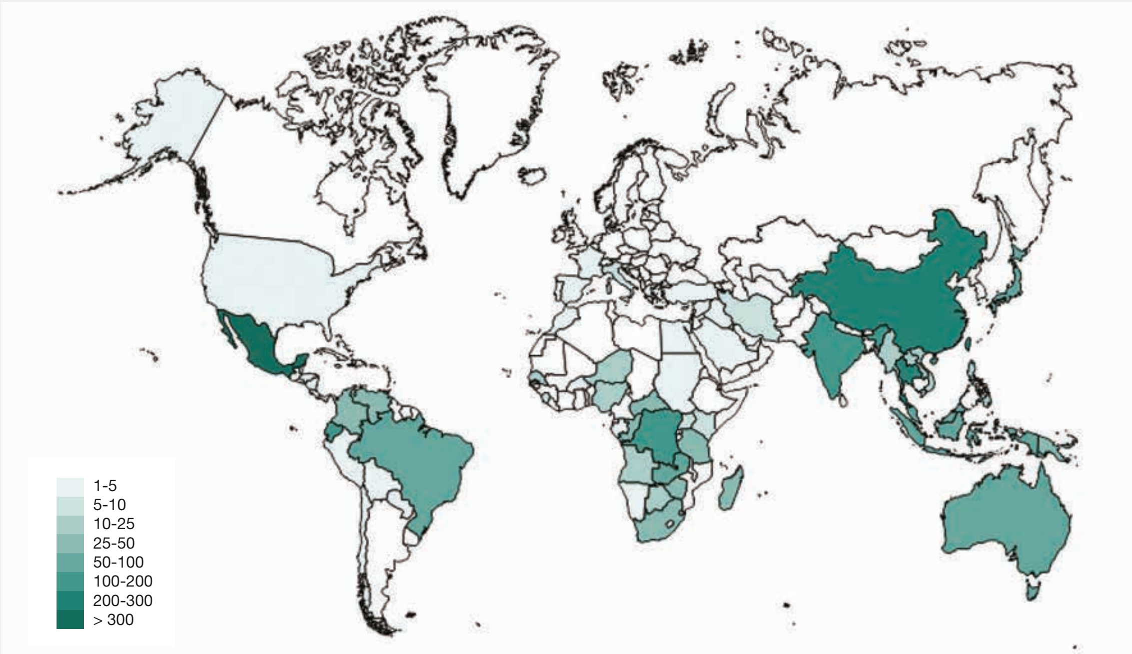


Figure 3 47 **Recorded number of edible insects, by country.**

This map is directly copied from its original source (Tiencheu & Womeni, 2017) and was not modified by the assessment authors. The map is copyrighted under license CC BY 3.0. The designations employed and the presentation of material on the maps used in the assessment do not imply the expression of any opinion whatsoever on the part of IPBES concerning the legal status of any country, territory, city or area or of its authorities, or concerning the delimitation of its frontiers or boundaries. These maps have been prepared or used for the sole purpose of facilitating the assessment of the broad biogeographical areas represented therein and for purposes of representing scientific data spatially.

The life cycle and host plants of edible caterpillars are well understood by local communities and this knowledge is communicated orally over generations. A survey of 39 ethnic groups, covering 21.4% of the all-ethnic groups in the Amazon basin, identified 115 edible insects with 131 local names. An additional 384 local names of edible small invertebrates could not be identified, indicating that local traditional knowledge was richer than the scientific understanding at the time (Paoletti, Buscardo, & Dufour, 2000).

Traditional land owners have, in most cases, developed harvesting protocols and habitat management practices that ensure sustainability (Yen, 2009). Traditional regulation of caterpillar harvesting in northern Zambia involves several aspects. Local people monitor development and abundance of edible caterpillars, changes in caterpillar habitats, protection of host plants and moth eggs against late bush fires and temporary restrictions on harvest of edible caterpillars (Mbata, Chidumayo, & Lwatula, 2002). Local knowledge also involves an understanding of processing to remove toxins that make inedible insects edible.

In agricultural systems, chitoumou *(Cirina butyrospermi)* are harvested. The time of harvesting, eating and selling of these caterpillars (so called 'chitoumou wakati') varied greatly in different areas and from year to year. Women consider caterpillars and shea nuts to be their primary income sources (Payne, Badolo, Cox, *et al.*, 2020; Payne, Badolo, Sagnon, *et al.*, 2020). Harvesting caterpillars has increased food security, although this is often from increased income from sale of fresh or dried caterpillars, rather than direct consumption.

Insects on the whole are vulnerable to overharvesting, habitat destruction, pesticides and other pollution and to climate change (Arnold van Huis *et al.*, 2013). For instance, habitat destruction has an impact on the availability of edible caterpillars *(Eucheira socialis)* in mountainous regions of Mexico. Problems can also arise when there are market demands that encourage non-specialist harvesters to harvest. In Australia, the use of edible insects by traditional indigenous owners has decreased significantly since European settlement. This is due in part to the displacement of indigenous people, the loss of traditional knowledge and language, and the adoption of a European diet. The harvest of edible insects, particularly in relation to nature-based tourism, now has implications for overharvesting in Australia (Yen, 2009). This is also the case of escamoles *(Liometopum* spp.) in Mexico where edible ant larvae with a high market value were affected when non-local people harvested them for profit (Ramos-Elorduy, 2006).

During the last five years the scientific interest and knowledge on insects as food has grown exponentially (van Huis, 2020). The industrial sector is increasingly engaged in rearing, processing and marketing of edible insects. The use of insects as human food (or as food supplements) or for feeding poultry and fish can contribute to more energy-efficient food production and promote environmental protection. An assessment conducted by the FAO concluded that insects represent a potential sustainable food source to address global food security concerns (van Huis *et al.*, 2013). However, insects could pose several microbiological and chemical health risks, which must also be considered (Imathiu, 2020).

3.3.3.2.4 Recreational hunting

Recreational hunting refers to practices where the purpose of the hunt is for the hunter's own personal use and enjoyment as opposed to harvesting for commercial or subsistence use (which are dealt with in section 3.3.3.2.3). Hunting is broadly considered as one way in which nature contributes to human wellbeing in a variety of context specific ways (Díaz *et al.*, 2018) and recreational hunting may be associated with a range of values and motivations, including food, social and cultural motivations, sport and exercise. As in all forms of hunting, there is a high degree of multi-functionality (*sensu* Fischer *et al.*, 2013). For example, a Scandinavian moose hunter may hunt in order to secure a year's supply of wild meat, in order to enjoy time exercising outdoors in the forest during autumn, to enjoy time spent socializing with family members or friends that make up his hunting team, to maintain the cultural tradition of harvesting natural resources by hunting in a forest, to help regulate the size of the moose population so that damage to commercially harvested tree species and traffic collisions is kept to acceptable levels, and for the possible chance to bring home a "trophy" set of antlers. Depending on if the hunter is a landowner, he/she may also have commercial interests via the sale of meat or hunting licenses (Fischer *et al.*, 2013; Storaas *et al.*, 2001).

The range of values associated with recreational hunting is reflected in the many terms linked to recreational hunting in the literature, including *inter alia* sport hunting, hunting tourism, safari hunting, trophy hunting, and big game hunting, or the use of terms associated with the hunting of particular species like deer hunting or duck hunting. Although these terms are sometimes used as synonyms, there is no agreed typology and the same terms can have different meanings in the literature, which can confound any attempt at synthesizing the evidence on sustainable use. The International Union for Conservation of Nature Red List uses the term 'sport hunting' for hunting where the end use is for the "collection and preservation of dead specimens for personal pleasure" (IUCN, 2020a). This definition is close to the definition of trophy hunting (see following paragraph) but differs from other interpretations where the term sport hunting/shooting is meant to differentiate it from market or commercial hunting and therefore covers a broader range of end uses. For example, grouse shooting in Scotland and England is regarded as sport shooting (Tharme, Green, Baines, Bainbridge, & O'Brien, 2001). This is an important distinction when trying to identify and interpret data sources for this assessment. The definition of recreational hunting used here encompasses all forms of hunting where the primary purpose is not subsistence or the commercial harvest of animals.

The term "trophy" hunting is a non-technical label that has been used for hunting practices where one of the end products is a photograph and/or the preservation of the whole or part of the hunted animal (i.e., a "trophy'). Within the context of recreational hunting, trophy hunting is often used for hunting practices where client hunters pay high prices to shoot particular species or individuals with particular attributes, e.g., large horns. There are therefore certain ecological, social and economic considerations that differ from other forms of recreational hunting.

There is a large amount of academic literature on the sustainability of recreational hunting and active management strategies for maintaining this practice. However, only a limited number of these studies contain well-argued, data-driven evidence. A recent assessment of recreational hunting (Di Minin *et al.*, 2021), using a similar protocol to IPBES assessments, identified 1342 relevant references but still concluded that "despite the extensive literature on recreational hunting, the evidence to address some of the most pressing academic and societal questions is still limited". Crucially, this included a paucity of evidence on critical policy relevant questions about when recreational hunting is sustainable and who benefits from it. One consequence of the limited information is that conclusions often reflect the value system, community status ("outsiders" *versus* "locals"), and professional background of the authors (Houdt *et al.*, 2021; Mkono, 2019; Nordbø, Turdumambetov, & Gulcan, 2018). Despite the limitations of the available literature and the challenges with assessing recreational hunting as a form of sustainable use, some of the key points raised in the literature are discussed further in this section.

The section first outlines differences in approaches across IPBES geographic regions and then examines evidence for various aspects relating to the sustainability of recreational hunting.

An overview of recreational hunting across IPBES regions

There is no global database of countries where recreational hunting occurs but several sources indicate that it is widespread. Species assessed for the International Union for Conservation of Nature Red List, and where sport hunting has been identified as a use, come from all major IPBES regions. Academic studies of recreational hunting have been conducted in 147 countries (Di Minin *et al.*, 2021) indicating that the practice takes place in a large number of countries spread across all IPBES regions.

There is considerable variation in the way that recreational hunting is governed and administered in different regions, especially relating to whether recreational hunting is allowed, whether it is regulated, who owns the wild species (government or private), who owns the land where the hunt takes place (private, public or communal lands), who can hunt (residents vs foreigners), how the hunt is managed (with an outfitter or community involvement), whether the use is purely personal or the hunted animal can be sold, who issues the licenses, whether there are bags or quotas for target species, what monitoring systems are in place, and whether the revenue from hunting is retained by landowners. These factors all have important implications for assessing sustainable use. It is not possible to provide a detailed analysis for all countries but some of the major aspects relating to each IPBES region are presented below.

AMERICAS

There are important policy differences regarding recreational hunting across the Americas. The practice is mostly not encouraged in Central and South American countries and legislation to prohibit recreational hunting exists in at least Colombia, Costa Rica and Brazil. Some South American countries allow recreational hunting of introduced animals (Argentina) and recreational hunting has been recorded as a use for at least 39 species of birds and mammals endemic to the region (IUCN Red List 2021). Despite prohibitions on recreational and other forms of hunting in South America, it is regarded as widespread and under-researched (Petriello & Stronza, 2020). An analysis of online videos showed that recreational hunting occurs frequently in Brazil (El Bizri, Morcatty, Lima, & Valsecchi, 2015) and is regarded as a part of local culture (Bragagnolo *et al.*, 2019).

In contrast, recreational hunting of wild animals is allowed in Canada, United States of America and Mexico where there also are active communities of hunters. In Canada there are an estimated 1.3 million hunters (Conference Board of Canada, 2018) whereas in the United States of America there were 11.5 million hunters in 2016, down from 37.8 million in 2001 (U.S. Department of the Interior, 2017). Recreational hunting occurs across large parts of Canada and the United States of America where the practice is allowed on private and public lands. The United States of America Department of Interior noted that hunting was permitted in "76 areas managed by the National Park Service, 336 national wild species refuges and 36 wetland management districts managed by the United States of America Fish and Wildlife Service, and over 220 million acres (890 000 km^2) of managed public lands" (U.S. Department of the Interior, 2017).

In the United States of America, regulated recreational hunting has been an integral part of the North American model of wild species conservation, providing social and political support as well as financing for wild species management activities (Arnett & Southwick, 2015; P. Mahoney & Geist, 2019). The early phase of North American wild species management concerned halting and reversing wild species decline, but more latterly the focus has been to manage populations within a 'social carrying capacity' (Heffelfinger, Geist, & Wishart, 2013). Wild species in the United States of America "owned" by state governments and hunting, are administered by State Fish and Wildlife Departments on both public and private land. However, it is estimated that >60% of hunting days occur on private land, which can present challenges in prescribing the legal relationships between publicly owned wild species and privately owned land (Freyfogle & Goble, 2019). The financing, management and governance of this land is under-studied (Poudyal, Bowker, Green, & Tarrant, 2012). Hunting is generally open to residents, with low priced hunting tags providing access to most prospective hunters. The sale of wild species meat and other products is illegal, and exchange is usually personal. Hunting revenues form part of a publicly managed and funded system where part of the budgets for all fifty State Fish and Wildlife Agencies is derived from user fees, including hunting and fishing licenses and federal excise taxes on hunting/fishing equipment (Arnett & Southwick, 2015). Through the Pittman-Robertson Act (Federal Aid in Wildlife Restoration Act of 1937), there is an 11 percent excise tax on the sale of firearms and ammunition products. The funding paid into the Wildlife Restoration Trust Fund provided an average of 751 million United States dollars annually from financial year 2012 to 2018 (P. Mahoney & Geist, 2019). Total expenditures on hunting decreased from 36.1 billion United States dollars in 2011 to 26.2 billion United States dollars in 2016, in line with declines in the number of hunters (U.S. Fish and Wildlife Service, 2016).

AFRICA

Africa is the only continent that retains its full spectrum of Pleistocene wild species (Ripple *et al.*, 2015) but there are substantial differences across Africa in the abundance of wild species, the way that wild species is managed, whether hunting is allowed, and the conditions regulating recreational hunting. Recreational hunting is not permitted in Kenya whereas it is allowed in many other African countries. Recreational hunting is recorded as a use for at least 90 species of mammals and birds across Africa (IUCN Red List 2021), and recreational hunting opportunities are advertised on the internet for at least nine countries in Southern, East and West Africa.

In African countries where recreational hunting is allowed, large areas of land may be managed partially or exclusively for hunting, especially for high paying clients. The area managed for recreational hunting, or where recreational hunting occurs, comprises as much as 26% in some countries, e.g., Tanzania (Di Minin, Leader-Williams, & Bradshaw, 2016) and has been estimated to be 1,394,000 km^2 for all of Africa (Lindsey, Roulet, & Romañach, 2007) and separately as 140 000-170 000 km^2 in South Africa (Taylor, Lindsey, Nicholson, Relton, & Davies-Mostert, 2020) and 288 000 km^2 in Namibia (Lindsey, 2011). The revenues from hunting have been estimated at 217 million United States dollars per year for seven Southern African countries (Di Minin *et al.*, 2016) and these revenues have been credited with funding 'rewilding' of commercial and communal farmlands in some Southern African countries where land conversion has been reversed or avoided by allowing regulated hunting and other uses of wild species (Child, 2019; P. Lindsey, 2011; W. A. Taylor *et al.*, 2020). In the case of South Africa, hunting is estimated to contribute 64% of income on lands that are managed for wild species compared to live sales (28%) and nature-basted tourism (8%) (DEA, 2015).

Several park agencies in Africa are at least partially funded by hunting revenues, although the percentage of revenues used to fund conservation agencies may be considerably less than what accrues to private companies managing recreational hunting (Di Minin *et al.*, 2016).

EUROPE AND CENTRAL ASIA

Hunting is an integral part of the cultures and traditions of European rural society and there are estimated to be over 7 million hunters across the continent (Brainerd, 2007). The governance of hunting is often situated within the broader context of biodiversity conservation and recognizes that Europe is a biocultural system with blurred boundaries between nature and culture and wild and domestic systems (John D. C. Linnell, 2015).

Some form of recreational hunting has been recorded as a use for at least 88 species of mammals and birds from across Europe (IUCN Red List 2021). In terms of European Union legislation, 82 species of birds are allowed to be hunted in the European Union (Hirschfeld, Attard, & Scott, 2019) and 13 species of mammals and seven birds have been regularly recorded in hunting bags from Central Europe (Reimoser & Reimoser, 2016). Some species such as Red deer have been valued game species for millennia (John D. C. Linnell, 2015). The recorded volumes of animals hunted every year varies from a few individuals to several million: in the bags from nine countries in Central Europe, six species comprised >100 000 individuals and wild boar exceeded one million per annum (Reimoser & Reimoser, 2016); the estimated bags for birds across Europe was 52 million (Hirschfeld *et al.*, 2019) per year. The trend in some countries is for hunting of fewer species but for an overall increase in the biomass of hunted animals **(Figure 3.48)**.

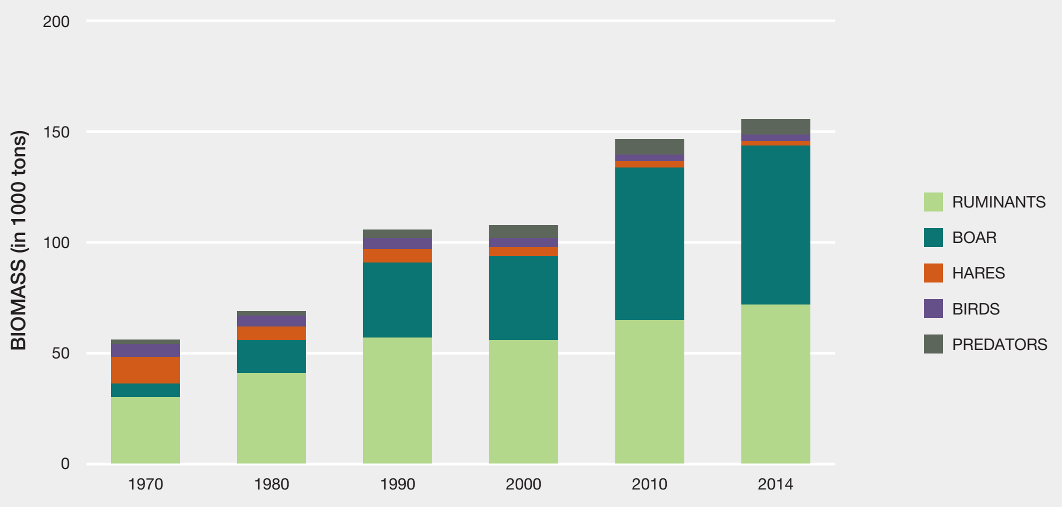


Figure 3 48 **Composition of harvested biomass (for nine European countries) in 1000 tons.**

Source: (Reimoser & Reimoser, 2016) under license CC BY 4.0.

The increase in biomass could be explained by the increase in the number of harvested ungulates which amount to approximatively 7 million every year (Linnell *et al.*, 2020). Hunting of large carnivores may also be allowed with the aim of reducing human-wildlife conflict, maintaining stable populations, and building public support for carnivores.

Despite the apparent increase in hunting bags of some species over the past 120 years (Reimoser 2014) populations of wild ungulates have increased across Europe (Linnell *et al.*, 2020) and this has also facilitated the recovery of large carnivores (Linnell *et al.*, 2020; Popescu, Artelle, Pop, Manolache, & Rozylowicz, 2016). Wildlife populations have tended to increase in Eastern Europe since 1990, especially in countries with reforms on the management of land and wild species (Bragina *et al.*, 2018). Hunters in Europe have been credited with providing monitoring data that supplements other forms of citizen science for wild species monitoring in 32 European countries (Cretois, Linnell, Grainger, Nilsen, & Rød, 2020).

There is also a long history of hunting and use of wild animals by people in Central Asia. The professional hunting economy that existed up to the 1950s gradually disappeared and was replaced by a growing number of amateur hunters. During the Soviet period, strict protected areas were imposed and some species were recovered through hunting bans (e.g., the nearly extinct saiga population). Hunting was controlled by central authorities. However, dramatic habitat loss and over-exploitation of wild species outside protected areas increased the threat to ungulates and other wild species, especially when trade liberalization after the Soviet era coincided with economic hardships and the weakening of state controls and capacities (Damm G.R., 2008). Unregulated hunting of species like markhor, combined with widespread and unregulated use of wild species for multiple purposes, has resulted in unsustainable use where poaching and sale of game meat became normal, and ungulates were reduced by poaching and rapidly increasing livestock populations (Blank & Li, 2021). The hunting sector is generally managed by government organizations through a permit system and wild species ownership remains centralized. This has replaced ancient kin-related ownership of hunting grounds, and some of the challenges associated with sustainable use of wild species have been ascribed to the lack of enforcement by state agencies and the loss of local systems of control (Blank & Li, 2021).

ASIA-PACIFIC

There is limited information on recreational hunting in Asia and the Pacific although it is recorded as a use for at least 100 resident or migratory mammal and bird species across all subregions (IUCN Red List 2021). The number of recorded scientific studies of recreational hunting is very low across the region, particularly for South Asia and Southeast Asia (with fewer than 10 publications) and to a slightly lesser extent for Northeast Asia and Oceania (Di Minin *et al.*, 2021). Recreational hunting in New Zealand and Australia focuses primarily on introduced or feral populations. New Zealand has hunting zones specifically set aside for recreational hunting.

Recreational hunting and sustainable use

Several studies have pointed out that an assessment of sustainable use needs to consider the social (including institutional and economic) and ecological factors affecting sustainable use (Fischer *et al.*, 2013), and be aware that sustainable use relating to recreational hunting is highly context specific (Di Minin *et al.*, 2021) **(Figure 3.49)**. This section examines evidence relating to the ecological, social and economic dimensions of sustainable use as it relates to recreational hunting.

Ecological aspects of sustainable use

The ecological and biological metrics used to assess sustainable use vary considerably but typically include the impact on population numbers. For bird and mammal species assessed for the International Union for Conservation of Nature Red List, and where sport hunting is identified as a use, 51% (*n*=620) have a declining population trend (IUCN Red List 2021). This implies that recreational hunting may not be biologically sustainable for these species. However, there are several limitations to the use of Red List data at the species level which would affect this conclusion. First, almost all the assessed species are subjected to multiple threats across different sites of which recreational hunting may only be a minor threat or a threat in only some areas of its range, so it is important to understand the context in which recreational hunting occurs. Second, the same species can be subjected to subsistence, commercial and recreational hunting and it is often not possible to disaggregate the effects of these different types of hunting.

An analysis of >1000 publications specifically focusing on recreational hunting (Di Minin *et al.*, 2021), identified 35 species that had been studied across multiple sites and these data provide a better understanding of population trends across sites. The results showed that only one species declined consistently across all sites, 11 (33%) species showed population declines in some sites but not others, and 23 (66%) species showed no decline or the results were inconclusive. These results highlight the extent of variation between sites. The study noted the geographical and taxonomic bias in published results with most of the studies focusing on a small number of mammal species mostly in Africa and North America. The paucity of evidence for many species and across other IPBES regions is important because the International Union for Conservation

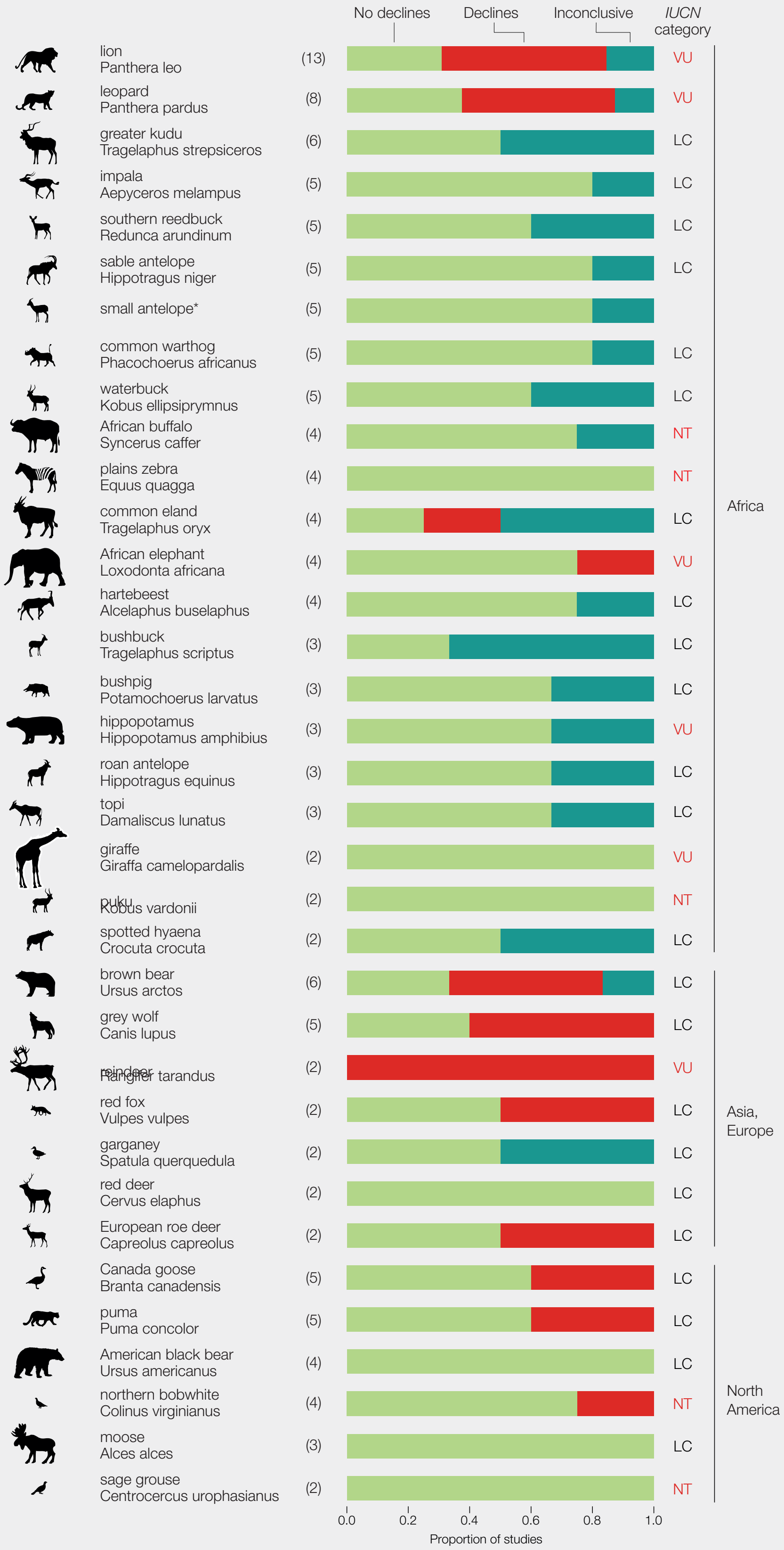


Figure 3 49 **Impact of recreational hunting on the population abundance of targeted species.**

Depicted is the proportion of studies that found inconclusive evidence, evidence of population declines, or evidence of no population declines. Number of studies is indicated in parentheses next to species name; only species included in more than one study are included. Abbreviations: LC: Least Concern; VU: Vulnerable; NT: Near Threatened. *Small antelope refers to steenbok Raphicerus campestris, oribi Ourebia ourebi, grysbok Raphicerus sharpei, duiker Cephalophus sp. or Sylvicapra grimmia, and dik-dik Madoqua kirkii. Source: (Di Minin *et al.*, 2021) under license CC BY-NC-ND 4.0.

Table 3 13 **Examples of populations of wild mammals that have recovered in areas where hunting management is in place even though global trends may be decreasing.**

(this does not mean there is an absence of continued threats)

Species name or taxonomic group	International Union for Conservation of Nature species status & global trends	Region or country	References
Black rhino *(Diceros bicornis)*	CR -Critically Endangered, increasing	Africa	(Challender & Cooney, 2016; CITES, 2019; Rosie Cooney *et al.*, 2017; NACSO, 2019)
White rhino *(Ceratotherium simum)*	NT – Near Threatened, decreasing	Africa	(Challender & Cooney, 2016; CITES, 2019; Rosie Cooney *et al.*, 2017; NACSO, 2019)
African lion *(Panthera leo)*	VU – Vulnerable, decreasing	Africa	(P. Lindsey, Balme, *et al.*, 2012; NACSO, 2019; Whitman, Starfield, Quadling, & Packer, 2004)
Different deer species (*Cervus* **spp.)**	LC – least concern, increasing	Europe (e.g., Germany), United States of America, Canada, Russia	(J D C Linnell *et al.*, 2020; Mustin, Newey, Irvine, Arroyo, & Redpath, 2012; Reimoser & Reimoser, 2016)
Bighorn sheep *(Ovis canadensis)*	LC – least concern, increasing	North America and Mexico	(Challender & Cooney, 2016)
Markhor *(Capra falconeri)*	NT – near threatened, increasing	Asia	(Rosie Cooney *et al.*, 2017)
Argali *(Ovis ammon)*	NT – near threatened, decreasing	Asia	(Rosie Cooney *et al.*, 2017)
Urial *(Ovis orientalis)*	VU – vulnerable, decreasing	Asia	(Rosie Cooney *et al.*, 2017)
Grey wolf *(Canis lupus)*	LC – least concern, stable	Some European countries	(Epstein, 2017)
Waterfowl	LC – least concern	North America	(M. G. Anderson & Padding, 2015; Hirschfeld *et al.*, 2019; P. Mahoney & Geist, 2019; Mustin *et al.*, 2012; Reimoser & Reimoser, 2016)
Mallard *(Anas platyrhynchos)*	LC – least concern, increasing	Europe, North America	(P. Mahoney & Geist, 2019; J. D. Nichols, Runge, Johnson, & Williams, 2007)
Greater white-fronted goose *(Anser albifrons)*	LC – least concern, unknown	Europe	(Hirschfeld *et al.*, 2019)
Phasianids e.g., black grouse *(Lyrurus tetrix)*	LC – least concern, decreasing	Europe	(Hirschfeld *et al.*, 2019)
Red-legged partridge *(Alectoris rufa)*	LC – least concern, decreasing	Europe	(Hirschfeld *et al.*, 2019)
Wild turkey *(Meleagris gallopavo)*	LC – least concern, increasing	North America	(Hirschfeld *et al.*, 2019)
European bison *(Bison bonasus)*	VU – vulnerable, increasing	Belarus	((Артеага В., 2019)
White-lipped peccary *(Tayassu pecari)*	VU – vulnerable, decreasing	South America	(Bodmer & Lozano, 2001)
Collared peccary *(Pecari tajacu)*	LC – least concern, stable	South America	(Bodmer & Lozano, 2001)
Paca *(Agouti paca)*	LC – least concern, stable	South America	(Bodmer & Lozano, 2001)
Agouti *(Dasyprocta fuliginosa)*	LC – least concern, stable	South America	(Bodmer & Lozano, 2001)
Polar bear *(Ursus maritimus)*	VU – vulnerable, decreasing	Canada	(Foote & Wenzel, 2009)
American alligator *(Alligator mississippiensis)*	LC – least concern, increasing/stable	United States of America	(Ruth Elsey, Woodward, & Sergio Balaguera-Reina, 2018)

of Nature Red List indicates that many more species across other IPBES regions are used for recreational hunting but there is no additional information to assess sustainable use of these species.

For those species that have been more intensively studied, there is evidence that mammalian game species with high reproduction rates, such as roe deer and wild boar, can tolerate more intensive exploitation and still maintain population numbers and structure as well as genetic diversity (Baldus, Damm, & Wollscheid, 2008; Challender & Cooney, 2016; J D C Linnell *et al.*, 2020; Loveridge, Reynolds, & Milner-Gulland, 2006; Tapper & Reynolds, 1996). As an example, populations of roe deer (Europe) and white-tailed deer (North America) have increased their range and density despite the intended use of hunting to reduce density-related human conflicts (Morellet *et al.*, 2007).

The evidence also shows that some populations of threatened species and those with low regeneration capacity have increased in numbers in systems where hunting is well managed **(Table 3.13)**. Attempts to combine hunting with the effective management and conservation of such species, has taken place in several IPBES regions. Note that the examples provided in **Table 3.13** apply only to the particular populations that were assessed, and not for the species generally.

In contrast, there is also evidence for populations where poorly regulated hunting is not sustainable and has contributed to local population declines that have reduced the number of animals that can be harvested sustainably, for example, some populations of lions and elephants in Africa (Fischer *et al.*, 2013; IUCN, 2016; Loveridge *et al.*, 2016; Mweetwa *et al.*, 2018; Packer *et al.*, 2009), brown bears in Northern Europe (Frank *et al.*, 2017), ungulates and Snow leopards in Asia (Rashid, Shi, Rahim, Dong, & Sultan, 2020).

Operationally, sound biological management is contingent on appropriate institutional, social and economic conditions. Scientists argue that biological sustainability of recreational hunting is highly connected with the proper regulation of the hunting system, including regular monitoring and adaptive management responses that adjust offtake to changes in population size (M. G. Anderson & Padding, 2015; Damm G.R., 2008; P. Mahoney & Geist, 2019; Souchay, Besnard, Perrot, Jakob, & Ponce, 2018). While these factors are important, they can also be achieved through local control and knowledge, and simple adaptive management systems (Goredema, Taylor, Bond, & Vermeulen, 2005). Some of the instances of unsustainable use have been associated with weak tenure, the centralization of revenues derived from hunting (Child, 2019) and breakdown of community governance without any effective replacement by state officials (Blank & Li, 2021).

Beyond population numbers, scholars have identified other biological and ecological issues that should be considered in the assessment of the sustainable use of recreational hunting. These include the indirect effects of hunting, which are often poorly known and therefore make it difficult or impossible to fully assess biological sustainability (for example Artelle *et al.*, 2018; Frank *et al.*, 2017; Macdonald *et al.*, 2017; M. N. Peterson & Nelson, 2017; Popescu *et al.*, 2016; Swenson *et al.*, 2017). In addition, all forms of hunting can have evolutionary and behavioral consequences for the target species, affect food chains, or alter herbivory, predation and other ecological processes (Fukushima *et al.*, 2020; Leclerc, Frank, Zedrosser, Swenson, & Pelletier, 2017). Selective harvesting of animals with particularly desirable phenotypes can also alter the distribution of traits in a population (Allen, Brent, Motsentwa, Weiss, & Croft, 2020; Coltman *et al.*, 2003; Crosmary *et al.*, 2013; Knell & Martínez-Ruiz, 2017; Milner, Nilsen, & Andreassen, 2007; Russo *et al.*, 2019; Wielgus, Morrison, Cooley, & Maletzke, 2013). Features such as body size or horn shape and size, may be linked to other fitness-related attributes, including physiological tolerances or disease resistance (Crosmary *et al.*, 2013; Knell & Martínez-Ruiz, 2017; Russo *et al.*, 2019).

Although these are important issues, the nature of available studies means that is not possible to make any firm conclusions regarding sustainable use based on these parameters (Di Minin *et al.*, 2021).

Social sustainability

Humans control their use of resources through formal or informal rules or institutions. The literature suggests that the primary variable affecting the sustainability or otherwise of recreational hunting is the governance of hunting systems (Cooney, 2017) and the quality and social legitimacy of relevant institutions (Fischer *et al.*, 2013). Analysis of a global dataset of utilized populations (not just for hunting) showed that utilized species declined more rapidly than unutilized species, but that where management systems were in place there was a positive impact on trends (McRae *et al.*, 2022). This broad analysis did not include institutional quality as an independent variable, and it is not possible to disaggregate the data for utilization under controlled *versus* open-access conditions, so it is not possible to assess the impact of management in more detail.

In an analysis of sustainable use, (AFischer *et al.*, 2013) fix citation format identified two aspects of institutional misfit that affect sustainability of recreational hunting: (i) conflicts between the functions of hunting as defined by the government and functions identified by local communities; and (ii) ecological functions embedded in formal institutions generated by non-local actors that are developed separately from, and in conflict with, the local institutions guiding the social and economic functions of hunting and land use more generally. One of the hypotheses is that hunting and the management of wild species become unsustainable when they are under-policed as open access resources and where wild species-based livelihoods are deinstitutionalized by land-use policies that favor agriculture farming (Bowles & Choi, 2013). These ideas have not been widely tested in hunting systems.

Legal, well-regulated recreational hunting has been shown in specific instances to play an important role in delivering benefits for both wild species conservation and for the livelihoods and well-being of indigenous and local communities living with wild species (Baldus *et al.*, 2008; Eklund T., 2017; C. Fischer, 2010). Investments from revenues generated through hunting on community conservancies have been used to improve local services such as water infrastructure, schools and health clinics, as well as providing meat for community members (IUCN, 2016; Naidoo, Weaver, *et al.*, 2016). Nevertheless, the evidence regarding these benefits across all areas where recreational hunting occurs is lacking.

Economic aspects of sustainable use

The economic literature on recreational hunting, specifically the generation and allocation of financial flows, tends to concentrate on two separate policy relevant questions. At a national level, total economic value is important because national policy makers are interested in economic growth, jobs and taxes and the revenue from hunting can therefore influence broader policy decisions affecting the sustainable use of wild species. At a local level, the policy question is whether the proportion of the value chain captured by the manager of land on which wild species occur is sufficient to enable reinvestment in the supply and management of wild species.

Recreational hunting has been considered an important economic activity by various scholars and stakeholders where it is credited with generating revenues and creating jobs in the land management and hospitality sector, as well as providing income and other important economic and social benefits to indigenous and local people in rural, remote and/or otherwise marginal areas (Conference Board of Canada, 2018; R. Cooney, 2017; Di Minin *et al.*, 2016; Sánchez-García *et al.*, 2021). Economic impacts of recreational hunting can be measured in terms of gross output (revenue), sales, income, employment or value-added benefits, and a summary of the data is provided in **Table 3.14**. While these measures are not always comparable, the table provides an indication of economic values.

Prices paid for hunts vary from hundreds to hundreds of thousands of United States dollars **(Table 3.15)**, and globally create a substantial revenue flow from developed

Table 3 14 **Hunting economic output.**

Abbreviations: OECD: Organization for Economic Cooperation and Development, EUR: euros, USD: United States Dollars.

Country	Gross revenues	People employed	Number of hunters	Reference
Finland	EUR 0,23 billion	100	304,245	Bioeconomy.fi, 2017)
Sweden	The annual gross hunting value is estimated to be in the neighborhood of USD 460 million	More than 12,000	300,000	(Mattsson, 2008; Mensah & Elofsson, 2017)
Austria	EUR 0,475 billion	More than 157 thousand	123,283	OECD.stat 2019
France	EUR 3,6 billion	25,800	150-200	(P. A. Lindsey et al., 2007)
Germany	EUR 1,6 billion	More than 617 thousand	368,664	OECD.stat 2019
United Kingdom	EUR 3,2-5,5 billions	12,000-74,000	600,000	(Mensah & Elofsson, 2017)
United States of America	USD 7,978,472 million	57,251,937	11,500,000	(U.S. Fish and Wildlife Service, 2016)
Russia	USD 518 million	More than 5587 thousand	2.8 million	(Braden, 2014)
South Africa	EUR 0,341 billion	17,000	76000	(Saayman, van der Merwe, & Saayman, 2018)
South Africa ("trophy" hunting alone)	USD 181 million			(Snyman *et al.*, 2021)
Canada	USD 13,2 billion	107,000	More than 50,000	(The Economic Footprint of Angling, Hunting, Trapping and Sport Shooting in Canada, 2019)
Sub-Saharan Africa	at least USD 201 million per year	More than 150,000	More than 100,000	(Di Minin *et al.*, 2016)

Table 3 15 **Indicative information on the species hunted, the number of individuals and the costs of trophy hunts in different countries.**

Abbreviations: DKK: Danish Krone, EUR: euros, USD: United Staes Dollars.

Country	Main hunted species	Individuals hunted/year	Cost of the trophy	Reference
United States of America	Deer, wild turkey, elk	More than 6 million	USD 2,659	(Bergstrom, 2008; Munn, Hussain, Spurlock, & Henderson, 2010)
Finland	Moose	49,667		(Bioeconomy.fi, 2017)
Kyrgyzstan	Snow leopard	Average of 25	EUR 7,000-10,000	(Eklund T., 2017)
Middle Europe	Red deer	-	EUR 10,000-15,000	(Bioeconomy.fi, 2017)
United Kingdom of Denmark	Red deer	Approx. 25,000 red-deer	DKK 7,000-25,000	(Bioeconomy.fi, 2017)
Germany	Red deer	9-12	EUR 1,000 without antlers, up to 5,000 with antlers	(Bioeconomy.fi, 2017)
Zambia	Lechwe, Hippopotamus, Leopard	300		Science Direct/Statista Charts, (P. Lindsey, Balme, *et al.*, 2012)
Tanzania	Leopard, Hippopotamus, Elephant	7034		Science Direct/Statista Charts, (P. A. Lindsey *et al.*, 2007)
Botswana	Elephant, Leopard, Lechwe	2500		Science Direct/Statista Charts, (P. A. Lindsey *et al.*, 2007)
South Africa	Impala, Warthog Kudu	53,885		Science Direct/Statista Charts, (P. A. Lindsey *et al.*, 2007)
Zimbabwe	Elephant, Leopard, Chacma Baboon	11,318		Science Direct/Statista Charts, Lindsey *et al.*, 2012(P. A. Lindsey *et al.*, 2007)
Mozambique	Crocodile Elephant	900		Science Direct/Statista Charts International (Sheikh & Bermejo, 2019)
Namibia	Zebra, Chacma Baboon, Leopard	22,462		(P. Lindsey, Balme, *et al.*, 2012; Sheikh, Bermejo, & Procita, 2019)

to developing countries, as well as from urban to rural areas within countries (Booth, 2010; Di Minin *et al.*, 2016; IUCN, 2016; Sánchez-García *et al.*, 2021). Besides spending money on hunting equipment, guns, ammunition, transportation, clothing, and meat processing, hunters typically also spend large amounts of money on permits, guide and outfitting services and travel (Lindsey *et al.*, 2007; U.S. Fish and Wildlife Service, 2016), contributing to the economies of the areas where this practice occurs, for example, on communal conservancies in Namibia (NACSO, 2015; NACSO & MET, 2018; Schmitt & Rempel, 2019).

From a production perspective, recreational hunting is regarded by scholars in this area as different from other forms of harvesting. First, the commodity value is only one of many values which collectively exceed the value of the raw commodity. Second, the process of hunting is considered a benefit to the production system, whereas with harvesting it is a production cost (Child, 2019). This is expected to give the multi-value approach to managing lands for wild species use, including recreational hunting, an economic comparative advantage over simple commodity production, and therefore provides an avenue to keep natural lands intact (Child, 2019). However, these economic advantages may not be realized due to the tendency to associate property rights with domestic species but not wild ones (Bowles and Choi 2013). It should be noted that this perspective also focuses on recreational hunting in comparison with other more commercial activities such as production forestry. It is not meant to be a direct comparison with the wide range of practices reported on in other parts of section 3.3.

Large areas of land are managed for the production of recreational hunting. For all of Africa, this was calculated as 1 394 000 km^2 (Lindsey *et al.*, 2007), which includes 288000 km^2 of freehold land in Namibia (Lindsey, 2011) and between 170 000 and 205 000 km^2 (14-17% of total land area) in South Africa (Taylor *et al.*, 2020). Figures for other areas comprise at least 890 300 km^2 in the United States of America (Bureau of land management areas, US Dept of Interior 2017) and 1 780 km^2 managed as Recreational Hunting Areas in New Zealand (Fraser & Department of Conservation, 2000), noting that in New Zealand these areas are designated for hunting introduced land mammals and that larger areas were designated for commercial hunting.

The key species that generate the largest proportion of income through recreational hunting tourism in Africa are: elephants in Mozambique, Namibia and Zimbabwe, African buffalo in Tanzania, and sable antelopes *(Hippotragus niger)* in Zambia (P. A. Lindsey *et al.*, 2007). With the exception of rhinoceroses (*Ceratotherium simum* and *Diceros bicornis*) in Namibia and South Africa, and exceptionally large elephant trophies, lions generate the highest revenue per hunt (24,000–71,000 United States dollars) of any species in Africa **(Figure 3.50)**. Prices for lion hunts have been particularly high in Tanzania, and were also high in Botswana prior a hunting moratorium placed in that country (up to 140,000 United States dollars per hunt) (Lindsey *et al.*, 2012).

Legal, well-regulated recreational hunting can therefore support conservation by contributing to the preservation of the target species and the habitat in which it lives (Baldus *et al.*, 2008; Eklund T., 2017; Fischer *et al.*, 2013) (for discussion of this form of management as a driver of sustainable use, please refer to Chapter 4).

For emotional and ideological reasons hunting is often excluded as an option for income generation by international conservation non-governmental organizations and certain international funders. Some species are indeed so rare, endangered or sensitive that they are not suitable for even strictly managed and regulated hunting use. However, if wild species and protected areas are not successful resources and options for alleviating poverty, conservation efforts could be undermined. Total protection and trade bans can lead to a major devaluing of wild species because there are no longer economic incentives to protect them (Baldus *et al.*, 2008; Rosie Cooney *et al.*, 2017; NACSO, 2019). For example, it was estimated that if lion hunting were banned, areas across Southern Africa and outside of national parks (approximately 59,500 km^2) currently set aside for lion

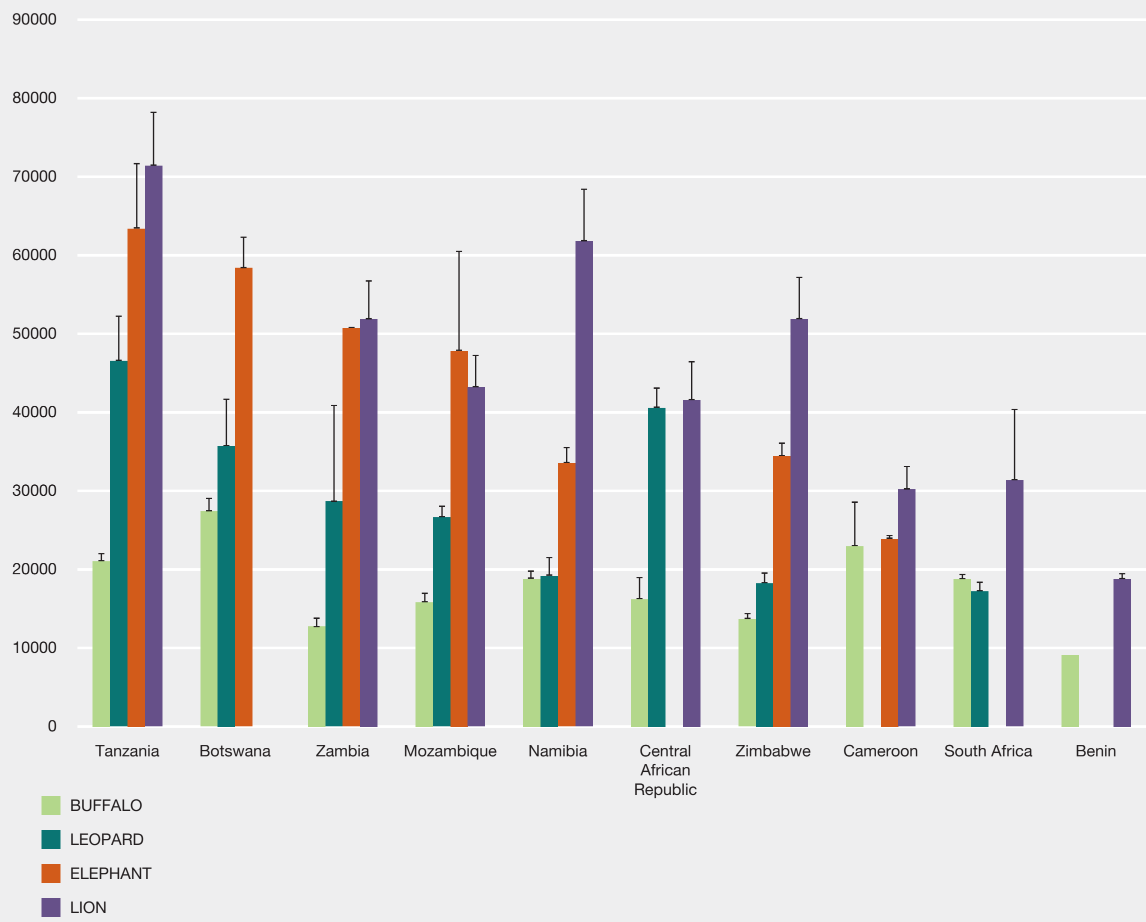


Figure 3 50 **Mean price for the cheapest trophy hunting packages (daily rates and trophy fees) for each of four key species.**

Source: (Lindsey, Balme, *et al.*, 2012) under license CC BY 4.0.

habitat could be converted to other uses such as agriculture (P. Lindsey, Balme, *et al.*, 2012). It is unlikely these areas would be incorporated into existing protected parks due to lack of funding (P. Lindsey, Balme, *et al.*, 2012).

Canned hunting

"Canned hunting" is a non-technical label that refers to the practice of placing captive-bred, semi-domesticated, and exotic animals within relatively restricted outdoor enclosures for the sole purpose of having the animals "hunted" and killed by paying clients (Graves, Mosman, & Rogers, 2012). "Canned hunting" represents a very small proportion of world hunting (IUCN, 2016) and is not a conservation strategy (Bilchitz, 2016; Williams, Loveridge, Newton, & Macdonald, 2017; G. C. Young, 2007). The practice has resulted in negative environmental and political consequences relating to recreational hunting and is regarded as a potential source of zoonotic diseases (HSI/HSUS, 2016; P. Lindsey, Alexander, *et al.*, 2012; D. W. Macdonald & Willis, 2013; Organ, Decker, & Lama, 2016; Somers & Hayward, 2012; B. K. Williams, Johnson, & Wilkins, 1996).

This is a highly contentious practice but mainly involves animals that are bred in captivity and therefore does not fall under the definition of wild species used in this assessment. In scientific and policy analyses, canned hunting needs to be separated from "ranched" wild species production, which involves the management of wild populations across extensive areas. In 2004 the World Conservation Congress, noting strong opposition to all forms of "canned hunting", accepted that well-managed recreational hunting has a role in the managed sustainable extractive use of wild species, and condemned the killing of animals in small enclosures where they have little or no chance of escape or where they do not exist as free ranging (IUCN, 2004). The International Union for Conservation of Nature encouraged the media and decision-makers to distinguish between canned hunting of confined animals and trophy hunting of free-ranging animals (IUCN, 2016).

3.3.3.2.5 Science and education

Scientific gathering is a tightly regulated and highly controlled activity. It brings benefits to conservation, management and science, can help diagnose or monitor the health of a population, species, or ecosystem and, as a result, protect certain species of animals from other causes of decline (Remsen, 1995; Sikes & Paul, 2013; Winker *et al.*, 2010). It can also have detrimental effects. Documented cases of decline due to removal of animals for scientific purposes usually involve large vertebrates (Gibbons *et al.*, 2000). For small mammals, responsible specimen gathering and removal have little impact on populations and have several benefits for science (Hope, Sandercock, & Malaney, 2018). However, recent studies indicate that harvest of voucher specimens for bats research is harming fragile populations (Russo *et al.*, 2017). For many invertebrate species, collection for scientific research is fundamentally important for species identification. Currently extraction of wild animals for scientific and educational purposes faces a series of economic and social pressures, including budget cuts and shortfalls (Suarez & Tsutsui, 2004), high harvesting cost (Enrique *et al.*, 2020), ethical considerations and significant and costly compliance procedures like those from the Convention on International Trade in Endangered Species of Wild Fauna and Flora for the cross-border exchange of specimens (Roberts & Solow, 2008).

Systematized repositories of life in all of its forms are cornerstones of quality research and education in many areas of science and innovation (National Academies of Sciences, Engineering, and Medicine; Division on Earth and Life Studies; Institute for Laboratory Animal Research; Roundtable on Science and Welfare in Laboratory Animal Use, 2019; Winker *et al.*, 2010). Schools, universities, and research laboratories use biological collections to teach concepts of evolution, ecology, taxonomy, physiology, biogeography, conservation, and more. Museum collections, while historically significant, have been greatly reduced by limiting numbers, even if species are common, as financial costs and ethics of maintaining and building these collections have changed. Larger series collected historically have been profoundly important in establishing both presence of absence, and providing evidence on historical population levels. This has been especially important with amphibians (Mahoney & Rueschemeyer, 2003). Biological collections also connect the public to nature and science, bolstering lifelong learning (Graham, Ferrier, Huettman, Moritz, & Peterson, 2004; Hill *et al.*, 2012; MacFadden, 2019; National Academies of Sciences, 2020; Suarez & Tsutsui, 2004). In some cases, digital technologies are able to successfully replace extractive practices for scientific and educational purposes.

In many countries, legislation improves animal welfare by setting minimum standards and currently covers all taxonomic groups of vertebrates and cephalopods. In European Union countries the use of wild animals is largely prohibited (Hartung, 2010). In the United States Institutional Animal Care and Use Committees (IACUC) are based at colleges and universities and follow national standards to conduct evaluations of animal care and use, including ethical and properly implemented care of wild animals by researchers. Permit-granting agencies are also in a position to place severe restrictions on the number of specimens that may be taken by scientists (Remsen, 1995; Russow & Theran, 2003; Silverman, Suckow, & Murthy, 2000). Currently, there is a tendency for reducing animals in experimentation and replacing animals for artificial models or digital simulators (Robinson *et al.*, 2019; Soulsbury *et* *al.*, 2020; Volker D., 2006). The potential harm to animal populations should be balanced with anticipated benefits (Brønstad *et al.*, 2016; Russow & Theran, 2003). Stricter controls can be detrimental to building conservation knowledge (Hochkirch *et al.*, 2021).

The impact on wild populations of scientific extraction of specimens is usually, but not always, small relative to other causes of mortality including natural mortality, hunting, collisions (e.g., road kill, bird death due to glass windows and communication towers, etc.), and habitat loss or alteration (Erickson, Johnson, & Young, 2005; Remsen, 1995; Rocha *et al.*, 2014; Winker *et al.*, 2010). For example, the entire vertebrate specimen collection of Museum Victoria (Australia), houses less than 200,000 specimens harvested within Victoria over the past 150 years (Clemann *et al.*, 2014). In comparison, duck and quail hunters in Victoria are estimated to have killed 638,000 native birds as of 2012 (Moloney & Turnbull, 2012). It should be noted that museum collections, unlike game hunting, aim at covering a much broader biodiversity, and hence may also exploit small, rare, or endangered populations unlikely to be targeted by hunters (Donegan, 2009).

The use of animals in human biomedical research has been of particular focus, more so for ethical considerations than for whether or not extraction of individuals for medical research is a sustainable form of use. Members of the Callitrichidae primate family (marmosets and tamarins) have been used since 1960s as biomedical research subjects because of their small size, wide availability, and relatively inexpensive costs. They are extracted mostly from wild populations in South American countries such as Brazil, Colombia, Peru, and Venezuela. In the 1960s and 1970s the Oak Ridge Associated Universities had the largest tamarin/marmoset population in the United States of America, housing about 550–650 animals (National Academies of Sciences, Engineering, and Medicine; Division on Earth and Life Studies; Institute for Laboratory Animal Research; Roundtable on Science and Welfare in Laboratory Animal Use, 2019).

Due to the Convention on International Trade in Endangered Species of Wild Fauna and Flora, the export of wild primates for biomedical research from Central and South America decreased significantly from 200,000 in the 1950s and 1960s to 5,000 specimens after 1975 when the Convention on International Trade in Endangered Species of Wild Fauna and Flora was enacted (Fialho, Ludwig, & Valença-Montenegro, 2016). High export levels have led to declines of some species, such as the cotton-top tamarin *(Saguinus oedipus)* (National Academies of Sciences, Engineering, and Medicine; Division on Earth and Life Studies; Institute for Laboratory Animal Research; Roundtable on Science and Welfare in Laboratory Animal Use, 2019). The cotton-top tamarin, as well as most marmosets and tamarins, are listed under Appendix I of the Convention. Thus, current import controls will favor wild populations, even though it does make it harder for researchers to acquire marmosets (National Academies of Sciences, Engineering, and Medicine; Division on Earth and Life Studies; Institute for Laboratory Animal Research; Roundtable on Science and Welfare in Laboratory Animal Use, 2019).

Today, marmosets as model organisms are attracting so much research interest that their demand far outstrips the already limited supply. Currently, 10–15 institutions are developing small marmoset colonies (of 20–60 animals each) for neuroscience studies. The growing focus on transgenic work has led to the development of some larger colonies (250–350 animals). If the field continues to grow, some facilities may establish much larger colonies (up to 1,000 animals) for line maintenance and characterization (National Academies of Sciences, Engineering, and Medicine; Division on Earth and Life Studies; Institute for Laboratory Animal Research; Roundtable on Science and Welfare in Laboratory Animal Use, 2019).

According to the United Nations World Conservation Monitoring Centre and the Convention on International Trade in Endangered Species of Wild Fauna and Flora trade database, reported exports of live macaques (for example, long-tailed macaques for research purposes) from six Southeastern Asian countries were more than 25,000 in 2019. While many animals are bred in captivity for scientific research, there are still significant extractions from wild populations to provide breeding stock. When the illegal trade is factored in (which often relies on legal trade to launder animals into the trade) coupled with unreliable or absent data on wild population numbers, this overall trade may be unsustainable.

The high demand for research animals has resulted in the manipulation of the Convention on International Trade in Endangered Species of Wild Fauna and Flora to ban imports of wild primates and birds into the United States of America and the European Union in order to bolster profit generated through commercial captive breeding programs. This raises all sorts of ethical issues (especially with the Convention on Biological Diversity) about the ability of indigenous and other peoples in range states to use their natural resources for economic development. The *ex situ* commercial captive breeding industry economically favors extinction of wild populations in range states that can potentially compete with them (Kasso & Balakrishnan, 2013).

The second-largest use of amphibians, after food, is for teaching and medical research. Frogs and salamanders are used as model organisms in medical research and are one of the classics for teaching animal biology at universities all over the world (Smith, Wassersug, & Tyler, 2007). The use of amphibians to aid advancing science has led to a number of significant scientific breakthroughs, and several Nobel prizes in physiology or medicine have benefitted from studies involving frogs, the last of which pertain to stem cells in regenerative medicine (Rossant & Mummery, 2012). According to studies on small lizards in Central America, many reptile populations are resilient to standard herpetological gathering (intensive gathering in short-term) (Poe & Armijo, 2014). Current sustainability efforts can potentially focus on reducing and replacing the use of animals in research and teaching with scientific alternatives emerging from innovative education and medical technologies. Using common and widespread species or animals raised in facilities for such activities would promote sustainability (Coleman, Carpenter, & Dunphy, 1996).

The killing of critically endangered birds and reptiles for scientific reference has caused debate and ethical dispute in the last two decades (Collar, 2000; Donegan, 2009). It was argued that the scientific gathering of voucher specimens is linked to the decline or loss of Mexico's elf owl *(Micrathene whitneyi soccorroensis)*, but others ascribe the extinction to invasive species (Minteer, Collins, Love, & Puschendorf, 2014; Rocha *et al.*, 2014).

3.3.3.2.6. Medicine and hygiene

The 2019 version of the International Union for Conservation of Nature Red List reports 1,660 species of animals have medicinal uses. Most known species (~77%) are chordates in terrestrial habitats (~72%). Globally, animals used for medicine comprise a relatively narrow subset of all animals, but they do occur across diverse taxa, habitats, and geographies. At least about 62% (n = 1025) of species have multiple uses. The most common additional use is food for human consumption, which approaches half (~46%, n = 769). Geographic hotspots of medicinal species occur in South America, Southeast Asia, India, and the tropical regions of Africa. Although not previously examined, geographic areas of prominent medicinal use (and threats to their use) likely occur where so-called human development is low (Short & Darimont, 2021).

Across varied geographies, threats to medicinal animals are more closely related to overall ecosystem degradation than human use. Among species with known population trends (n = 839), the highest proportion have a decreasing trend (~63%, n = 525), whereas about 30% (n = 254) are stable, and only about 7% (n = 60) have increasing populations. Primary threats are related to agriculture and aquaculture (~45% of species, n = 143) and biological resource use (~44%, n = 142), which includes exploitation for medicine, food, clothing, and other uses (Short & Darimont, 2021).

There are many examples of surveys that have documented the diversity of animals used in traditional medicine, some are highlighted in the section below.

Species of global interest

Amphibians and reptiles are used in traditional medicine or as part of cultural beliefs all over the world, resulting in harvest of these animals from the wild (Gorzula, 1996; Hocking & Babbitt, 2014; Schlaepfer, Hoover, & Dodd, 2005; UNODC, 2016). Alves *et al.* (2013) found that 331 species (284 reptiles and 47 amphibians) are used as part of traditional folk medicines around the world. The use of secretions, especially those of Bufonids that contains numerous active molecules, is one of the reasons they are desirable (Rodríguez, Rollins-Smith, Ibáñez, Durant-Archibold, & Gutiérrez, 2017). Insects are also used as medicinal resources all over the world (Costa-Neto, 2005).

Pangolins (four species in Asia and four species in Africa) are the most heavily traded wild mammal in the world (UNODC, 2016). Their various body parts, especially their scales, fetuses, blood, bones, and claws are largely used in traditional medicines (Boakye, Pietersen, Kotzé, Dalton, & Jansen, 2014; Mohapatra, Panda, Nair, Acharjyo, & Challender, 2015; Soewu A Durojaye & Sodeinde A Olufemi, 2015). Harvesting of two Asian species of Pangolins is largely driven by demand from China. This, in combination with additional threats related to habitat decline, are affecting the sustainability of use. These species are listed as critically endangered in the International Union for Conservation of Nature Red List (Heinrich *et al.*, 2016). With declining Asian pangolin populations, a shift in trade from Asian to African pangolin species has been suggested. As a result, the total number of incidents involving Asian species declined since 2000, yet they were still being traded in large volumes (more than 17,500 estimated whole Asian pangolins were traded between 2001 to 2014) despite a zero-export quota for commercially traded wild sourced Asian species (Heinrich *et al.*, 2016). The United States of America is also a significant largest importer of pangolins and their products (UNODC, 2016).

EUROPE

The traditional use of animals as a source of medicine is relatively low in Europe. However, Benitez (2011) reported 26 different animals provided 61 distinct medicinal uses in Western Granada Province, Andalusia (Spain). The high number of uses is due to the fact that some animal species are involved in more than one preparation method, sometimes with different parts used.

AFRICA

In Benin, 87 mammal species have been reported as traded for medicinal purposes including some vulnerable, endangered and threatened species (Djagoun, Akpona, Mensah, Nuttman, & Sinsin, 2013). El-Kamali (2000) identified 23 animal species whose products were commercialized for traditional medicine purposes in Central

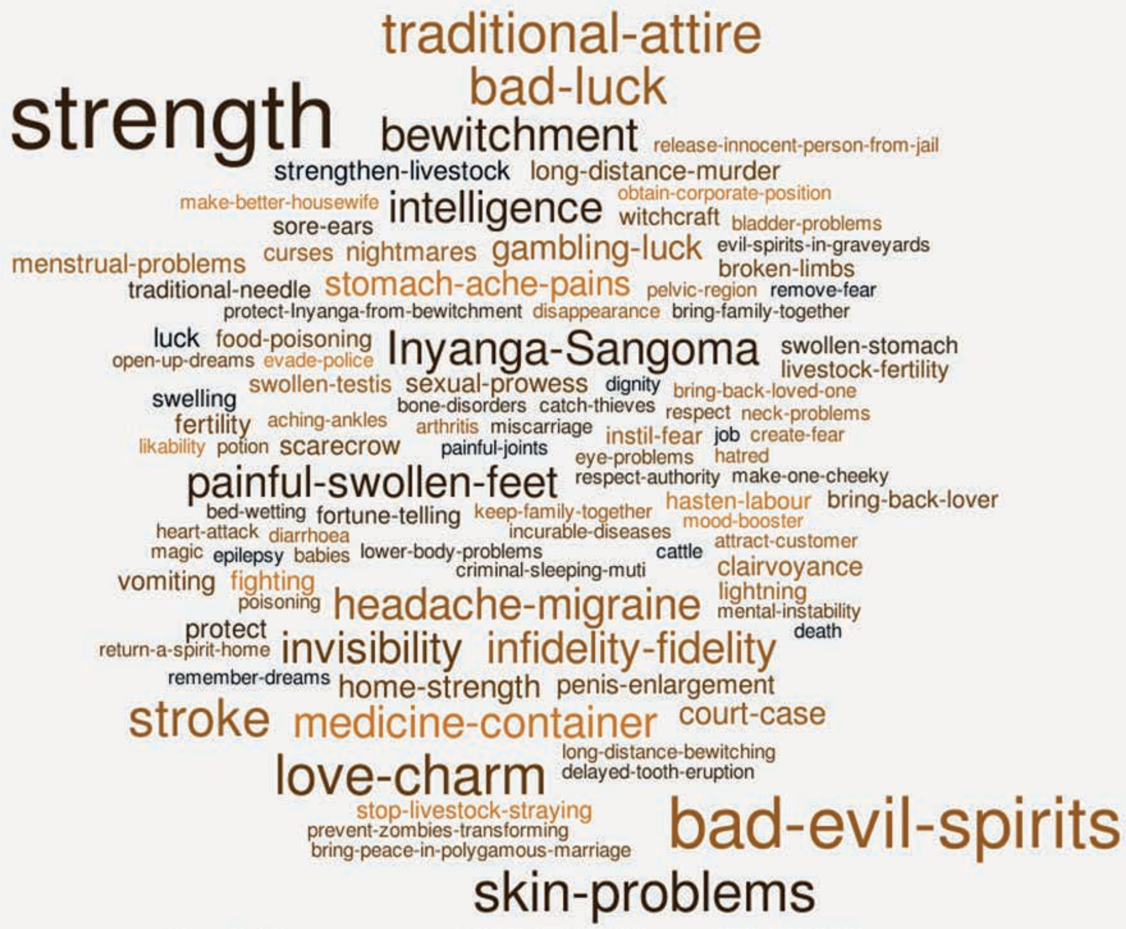


Figure 3 51 **Word cloud of the use categories derived from species used in animal-based medicine in South Africa.**

The size of the words in the figure is proportional to the number of times the uses were mentioned. Source: (Vivienne Linda Williams & Whiting, 2016) © 2015 Elsevier Ireland Ltd., license number 5153130663118. CC-BY NC.

Sudan. Sodeinde and Soewu (1999) recorded the use of 45 medicinal species in Nigerian markets. Simelane and Kerley (1998) showed that 44 species (eight reptiles, six birds, 30 mammals) were sold in 19 herbalist shops in the Eastern Cape Province of South Africa. Cunningham and Zondi (1991) examined the trade in animals for medicinal uses in KwaZulu-Nata Province and reviewed the literature reports for South Africa, recording at least 79 species of vertebrate (18 reptiles, 16 birds, 45 mammals), excluding domestic mammals and various marine invertebrates and fishes. A total 132 species of vertebrates (21 reptiles, 32 birds, 79 mammals) were reported by Ngwenya (2001) to be traded across KwaZulu-Natal Province, with 50 species highly demanded by the costumers. These were vultures, chacma baboon, green mamba, Southern African python, Nile crocodile, puff adder, striped weasel, and black mamba. Whiting *et al.* (2013) identified 147 vertebrate species that were traded in all South African traditional medicine markets. This represented around 63% of the total number of documented wild species. Recently William *et al.* (2014) reported 354 bird species (from 205 genera, 70 families, and 25 orders) used for traditional medicine in 25 African countries.

In numerous societies of Western and Central Africa, body parts of great apes (chimpanzee, gorilla, bonobo) are used for medicinal and/or ritual purposes. These practices usually operate according to the principle of analogy: the quality and value of the foodstuff is incorporated by the person who ingests it (Epelboin, 2012; Leblan, 2017). For instance, in Guinea, consuming the right arm of a chimpanzee will protect children from disease and make them good hunters, because monkeys are considered as violent and powerful beings (Leblan, personal observation). Scientists point out the unsustainability of such practices and the need for conservation strategies (Sá, da Silva, Sousa, & Minhós, 2012 on Guinea-Bissau). However, given the widespread interest in these species, it is also a matter of global debate.

Williams and Whiting (2016) reported 301 uses of animal parts for 122 broad-use categories **(Figure 3.51)** across South Africa. They used a word cloud to report their findings for visual impact. Although the study was conducted in South Africa, the categories of uses reported paint a picture of the health needs of the consumers of animal-based medicine elsewhere. 'Strength' (e.g., home strength, imbuing physical strength and overcoming fear) stands out as a dominant use, followed by protection to ward off evil spirits from within a person or from their residence.

LATIN AMERICA

A recent literature review on animal-based medicine recorded at least 584 wild species (13 taxonomic categories) as being used in the entire continent of Latin America (R. R. N. Alves, Rosa, Albuquerque, & Cunningham, 2013). The authors even speculated that this number might be underestimated given the limited number of studies on the theme, highlighting the conservation implications of the wild species use in medicine. Surveys carried out in 15 Brazilian cities reported that at least 180 animal species are traded for medicinal purposes (R. R. N. Alves & Rosa, 2010). In the State of Bahia, in Northeast Brazil, 50 insect species were reportedly used for medicinal purposes (Costa-Neto, 2005).

Didelphis. marsupialis has an undeniable cultural significance for local communities in the Amazon, both in terms of food and medicine. it is also designated as the best wild meat in the region. It is hunted by men, but the preparation of meat and medicinal oil are tasks mainly performed by women. The current study focused on riverine communities, who reportedly hunt the "common opossum" in morning or at night. They have a variety of techniques including handmade traps called "mundé", made from locally gathered wood and vines. However, this technique is declining because riverine people themselves believe that "mundé" does not select animals and it is harmful. Based on structured and semi-structured interviews with the local community, Barros and Azevedo (2014) found that this activity has not negatively affected the local populations of *D. marsupialis*. Some respondents stated that there is a decreased number of animals, other respondents argued that there is an increased number of opossums in the region. A third group said that the common opossum is a species that has a good reproductive capacity (it is a "mineral animal"), therefore, they think the population remains stable. Scientific studies suggest consumption of this species should be the subject of further studies, as this marsupial species has been described as a reservoir for parasites that cause severe disease.

ASIA

Use of wild terrestrial animals for medicinal purposes is widespread throughout Asia. Ashwell and Waltson (2008) recorded at least 47 animal species being traded for medicinal purposes in Cambodian markets, while Van and Tap (2008) recorded 100 different medicinal products from 68 animal species traded in Ho Chi Minh City, mainly sold as dried products (either the whole animal or parts) soaked in rice wine, or as a gel product which remains after boiling animal remains slowly in water.

The rhinoceros horn cut from live individuals are used in traditional Chinese medicine to dispel heat, detoxify blood, but were split over other purported medicinal properties, including its ability to treat cancer (Cheung, Mazerolle, Possingham, & Biggs, 2018). In 2018, the import and export of rhinoceros and their products will continue to be strictly prohibited; the sale, purchase, transportation, carrying and mailing of rhinoceros and their products are strictly prohibited; rhino horn and tiger bone are strictly forbidden to be used as medicine (http://www.china.com.cn/news/2018-12/13/content_74271446.htm).

3.3.3.3 "Non-lethal" terrestrial animal harvesting

Non-lethal uses of wild animals include all use forms that do not result in the death of animal through killing, contrary to lethal uses which take the life of animals. Non-lethal uses include ornamental use, scientific research, pets, green hunting, and religious and cultural practices and can benefit food security, economy, industry, and result in conservation. Traditional non-lethal uses of wild animals at local scales occur among indigenous communities, although biodiversity conservation and poverty alleviation remain a challenge in tropical biodiversity hotspots (Tranquilli, 2014).

3.3.3.3.1 Decorative and aesthetic

Natural fibers have important properties and are used as luxury goods and handicrafts that sell for better prices and generate higher profits for the community. Vicuñas *(Vicugna vicugna)* are a species which has received considerable attention regarding its sustainable use. Its hair produces one of the finest natural fibers in the world and is highly valued to make luxury fabric and clothing. The vicuña is the most representative wild ungulate of the high Andes of South America. In 1965, at its low point, the population of vicuña was estimated at only 6000, having collapsed from 1 million animals 25 years prior. Current population size is about 460,000-520,000 individuals, but they went through a serious and long-term overexploitation for 500 years. The recovery has benefited from a series of conservation actions, including the early prohibition of hunting and trade, establishment of the National Council of South American Camelids (Consejo Nacional de Camélidos Sudamericanos, CONACS), corral programs on community land and the practices of capturing and live shearing wild animals to earn high profits from selling the fiber. The benefit to society and the natural world of these efforts was the survival of a charismatic animal in its historical landscape (Sahley, Vargas, & Valdivia, 2007; Wakild, 2020). The restoration of depleted wild populations of vicuñas has reinstated the species in the ecosystem, and has allowed the development of sustainable use programs that directly assist the livelihoods and well-being of local people, and provides options for further economic development linked directly to successful conservation.

3.3.3.3.2 Food and beverage: honey

Wild honey is an important source of nutrition and medicine, and contributes to the income of local communities in many parts of the world. Wild honey harvesting is practiced by men and women belonging to many indigenous peoples and local communities. The harvest, filtration and preservation of wild honey relies in many parts of the world on rich traditional knowledge and its continued transmission across generations.

Wild honeybee local knowledge and traditional skills are key to sustainable use. In Lizongole, Mozambique wild honey gathering is typically carried out by groups of five to seven men sometimes called honey hunters (Ribeiro, Snook, Vaz, & Alves, 2019). Honey hunters have a mutualistic interaction with the honey guide bird *(Indicator indicator)* that directs men to trees containing honey. There they burn dried sticks to initiate fire and the felling of trees. Honey hunters apparently fell up to 560 trees per year. Impacts on tree populations vary among the 12 species killed for honey and are considered as a diminishing resource. Non-destructive traditional practices based on tree climbing are recommended (N. S. Ribeiro *et al.*, 2019). In Asian countries including India, the harvesting of wild honey from tall forest trees is done using bamboo baskets and bamboo ladders, and climbing trees with a smoke torch (Deori, Deb, Singha, & Choudhury, 2017). In the Southeastern United States of America tupelo honey production has been part of rural livelihood practices for several generations, and is carried out according to local ecological knowledge of both the trees *(Nyssa ogeche)* and the bees (Watson, 2017).

This non-lethal use of wild bees is widely proposed by local conservation stakeholders and is generally integrated into most of management plans of worldwide protected areas. It constitutes a sustainable alternative which provides a long-term income source to local people (Syampungani *et al.*, 2020). However, in many areas the required traditional knowledge is threatened due to changes in the related socio-economy, as more young people choose to work in the cash economy. For example, only 24% of the 251 local community members surveyed in Palawan Philippines could correctly identify the giant honeybee (Matias, Borgemeister, & von Wehrden, 2018).

3.3.3.3.3 Recreation: green hunting

Green hunting occurs with tranquilizer dart guns and the animals are released alive. This is typically performed for veterinary procedures or translocation, and has been suggested as an alternative to lethal forms of hunting (Greyling, McCay, & Douglas-Hamilton, 2004). Green hunting is cheaper and less harmful compared to traditional hunting and while immobilized, the animal can be micro-chipped or have tissue sampled (Greyling *et al.*, 2004). However, as green hunting is as of yet not a significant recreational activity, there is insufficient information on the status, trends and/or impact of the activity with regards to its potential impact on sustainable use of wild terrestrial species.

3.3.3.3.4 Pet and zoo trade

There are two distinct, but related, aspects to the live animal trade: the pet trade and the zoo trade. An array of live animals, eggs, and taxidermy are targeted by buyers worldwide for private collections and zoos or as exotic pets. Examples include reptiles, such as chameleons and tortoises; birds such as parrots and falcons; and mammals, such as tiger cubs and apes (ROUTES, 2022).

Zoological gardens and zoos represent *ex situ* conservation of wild animals for research and educational activities, tourism and recreation. The zoological parks basically exchange or buy/sell animals from each other, rarely do they buy specimens coming from the wild.

Wild animals are maintained in captivity for visual observations by the public. In an increasingly urban world, the ability of people to have contact with animals through zoos and pets adds significantly to the positive values people attribute to wild species – a prerequisite for their active engagement in conservation (Fukuda *et al.*, 2011). They constitute the ideal sites for environmental education, and therefore may have secondary level benefits for sustainable use due to a better educated public. In these cases, animals may progressively lose their wild instinct through a long-term habituation process. Given relatively few captive-bred animals do get released back into the wild, where they can make a significant contribution to *in situ* conservation, the role of encouraging people to value wild species positively is arguably the biggest conservation benefit that flows from zoos and exotic pet ownership.

The global pet trade is a large and complex industry. Pets are widely kept in many countries with 46% of United Kingdom of Great Britain and Northern Ireland and 62% of United States of America households estimated to have pets, supporting a multi-billion-dollar industry dedicated to their care and feeding (Human Society of United States, 2014; Pet Food Manufactures Association, 2014). Pet owners not only display more positive attitudes toward animals (Daly & Morton, 2009; N. Taylor & Signal, 2009), but also engage in more animal-related activities such as bird watching and viewing nature documentaries (Bjerke, Østdahl, & Kleiven, 2003). Pet owners are also more inclined to join and support animal welfare and environmental organizations (R. Bennett, 2003). As specified in the 2016 World Wildlife crime report: "In a range of countries, the capture and sale of wild-caught pets can be a way for rural communities to make money and for urban communities to express a link to the natural heritage of their countries. Display of these wild species can also draw tourists – exotic birds or even primates may be strategically positioned in front of restaurants for example, or wild species may be shown for a fee as a roadside attraction. International trade in exotic species has also become big business. Most of this involves relatively common species, but dedicated collectors may pay thousands of dollars for protected specimens, captive bred or supplied from the wild. Much of this trade involves birds, reptiles, and fish – populations that may prove difficult to monitor. The trade of tropical fish for aquaria and freshwater turtles and tortoises for terraria involves millions of individuals annually, and the share of this trade that comes from the wild is not always clear. About one quarter of all commercial live animal exports permitted under the Convention on International Trade in Endangered Species of Wild Fauna and Flora in 2013 were declared as wild sourced, with most involving species of birds, amphibians, or reptiles prized in the pet trade. In terms of total live animals, the most commonly exported were map turtles" (Vereinte Nationen, 2016).

From the 1980s to the present, approximately 12 million live internationally protected parrots were reported in international trade, according to the Convention on International Trade in Endangered Species of Wild Fauna and Flora export data. Most were either wild-sourced or of unknown origin (62%). Trade trends have been strongly influenced by national controls in key destination markets (UNODC, 2016). In 1992, the United States of America passed the Wild Bird Conservation Act, which sharply reduced the number of parrots and other wild birds imported to the United States of America. In 2005, the European Union banned the import of wild birds due to concerns about bird flu transmission (Vereinte Nationen, 2016). Both acts radically changed the international live bird market.

The pet-trade in wild frogs and amphibians concerns a minority of the recorded importations or exportations compared to other taxonomic groups of vertebrates. An analysis of trade data reported in the Convention on International Trade in Endangered Species of Wild Fauna and Flora database showed that trade in amphibians has increased in the last years, with ~ 40,000 animals exported per year globally as part of the trade of captive-sourced live animals (Harfoot *et al.*, 2018). This has resulted in a decrease in wild sourced exports since 2000. At the same time, these numbers may be underestimates due to mislabeling of specimens as captive-bred which may in fact be wild-caught (Auliya *et al.*, 2016). For example, between 2013 and 2018, the United States of America alone imported 3,655,620 live amphibians for the pet trade, belonging to 283 species (Mohanty & Measey, 2019).

The Asian houbara bustard *(Chlamydotis macqueenii)* is listed as Vulnerable by Birdlife International (2004) due to global population decline of 35 per cent over the last 20 years. The principal cause of declines has been hunting by Arab falconers (Collar *et al.*, 2017; Seddon & Launay, 2008; Tourenq *et al.*, 2004), and associated poaching of live birds, especially from Pakistan, for training of falcons in the Arabian Peninsula. However, Saudi Arabia has taken necessary steps to conserve dwindling populations of Houbara Bustards. The goal of houbara conservation in Saudi Arabia is to restore self-sustaining populations of resident breeding birds protected within a network of protected areas, but which may one day support sustainable falconry in hunting areas outside reserves (Gelinaud, Combreau, & Seddon, 1997; Seddon, Knight, & Budd, 2009; van Heezik & Ostrowski, 2001).

Unfortunately, not all species can be bred in captivity and some consumer countries do not have access to captive bred animals, so demand for wild animals persists. The harvest of live specimens, in many cases, involves significant mortality during capture, transport and holding. Many wild animal species controlled under current policies remain unsustainably traded to supply the international pet markets, with rare and endemic species most threatened (Auliya *et al.*, 2016; E. G. Frank & Wilcove, 2019; R. O. Martin, 2018; R. O. Martin *et al.*, 2014; Ngo, Nguyen, Phan, van Schingen, & Ziegler, 2019). Even with existing international regulations, the majority of species in exotic pet trade are not protected under the Convention on International Trade in Endangered Species of Wild Fauna and Flora, leaving international trade mostly unregulated and unmonitored (Janssen & Shepherd, 2018). In particular, species with small wild populations and/or small areas of occupancy, including island populations, are highly prone to overexploitation and decline due to the exotic pet trade (S. Altherr & Lameter, 2020; Flecks *et al.*, 2012; Lyons & Natusch, 2013). The high demand by specialized collectors for a "new" (i.e., only recently scientifically described) or rare species has caused intense collections in the wild, shortly after type localities were published – which is why an increasing number of scientists warn against publishing type localities (Lindenmayer & Scheele, 2017a; Maron, D.F., 2019). The sustainability of this form of consumer-driven use is unclear.

Additional issues related to the pet trade are: (i) some common methods of animal harvest for commercial trade result in destruction of habitats and shelters (see for example, Goode, Horrace, Sredl, & Howland, 2005) and (ii) the exotic pet trade has been identified as a pathway for the spread of invasive alien species (Shivambu, Shivambu, & Downs, 2020; Soule, 1990; Warwick & Steedman, 2021).

3.3.3.4 Emerging issues: terrestrial animals harvesting for integrated species and habitat management

Hunting is not only done for food and other products, but can also be an important component of wild species management (Linnell *et al.*, 2020; Winker *et al.*, 2010), and can be an important part of sustainable management practices for the wild species and their habitats. Wild species management is defined as the application of science-based and local knowledge in the stewardship of wild animal populations (including game) and their habitats in a manner that is beneficial to the environment and society (IUFRO, 2017).

Wild species are managed for several reasons, such as: (i) to reduce a range of human-wildlife conflicts; (ii) to prevent over-population and thus reduce related socio-economic and ecological threats; (iii) to maintain desired structure of game species populations (e.g., sex, age, morphology, etc.); (iv) and to support ecosystem functioning and resilience, including control of invasive and alien species.

There are many ongoing debates within conservation and management science concerning the best models for human-nature interactions (Cornicelli, Fulton, Grund, & Fieberg, 2011; Linnell *et al.*, 2020). Wild species management institutions designed to regulate hunter impacts on wild species and wild species impacts on human interests go back centuries in various forms, although the modern tradition appeared in North America and Europe in the early 20th century (e.g., Leopold, 1933). Management for hunting may involve the introduction of alien species, habitat modification, artificial feeding and the intensive control of predators, all of which can have widespread ecosystem effects.

Wild species management institutions motivated and funded by hunting activities have led to the dramatic recovery of many species of game (roe deer, red deer, white-tailed deer, moose, wild boar, brown bears, black bears, mountain lions, wild turkeys, the American Alligator) across North America and Europe to the extent that their populations are today higher than they may have been for centuries (Gross, 2008; Joanen *et al.*, 2021; Linnell *et al.*, 2020; P. Mahoney & Geist, 2019; Ripple *et al.*, 2014). The population levels of game species that are optimal for commercial hunting can at the same time be detrimental for forest regeneration and biodiversity conservation and lead to conflicts between different groups of actors and management goals. These high populations have secondary effects including changes in animal and plant community structure and function and spread of diseases (Gortázar, Acevedo, Ruiz-Fons, & Vicente, 2006; Mustin *et al.*, 2018). Human-wildlife conflicts often include damage to agriculture, disease transmission, traffic collisions, etc.

Trends in increasing populations for several popular game species has had detrimental effects on non-game species, both because of competition for resources and they are pursued by hunters as 'vermin' that threaten game populations (Denny, Latham-Green T., & Hazenberg R., 2021; Gross, 2008; Linnell *et al.*, 2020; Ripple *et al.*, 2014; Teichman, Cristescu, & Darimont, 2016). In the 20th century, large predators' populations were almost exterminated in North America and Western Europe (Ripple *et al.*, 2014), which caused multiple cascade effects to ecosystem functioning, such as "mesopredator release" effects (Brashares, Prugh, Stoner, & Epps, 2013; Prugh *et al.*, 2009; Soule *et al.*, 1988). Growing numbers of ungulates and other desirable game species (Grant, Mallard J., Leigh, S., & Thompson, P. S., 2012; Kuijper *et al.*, 2013) resulted in habitat alterations and degradation (Grant *et al.*, 2012; Kuijper *et al.*, 2013; Theuerkauf & Rouys, 2008), increased levels of infanticide among certain species (Swenson *et al.*, 2017) and hybridization (Salvatori *et al.*, 2020). Presently, populations of most large predators are maintained at a socially acceptable maximum and are even decreasing in certain parts of Europe (Fernández-Gil *et al.*, 2016; J D C Linnell & Cretois, 2018; Niedziałkowski, Sidorovich, Kireyeu, & Shkaruba, 2021; Virgós & Travaini, 2005). In the United Kingdom, one of the most criticized management actions of grouse hunting is population control of raptors (Denny *et al.*, 2021).

Killing of people and domestic stock by predators is a serious human-wildlife conflict. Most predator populations were historically subject to severe depletion and sometimes eradication to the point of extinction. However, societal perspectives on wild species have changed over time, and now conservation actions are focused on rebuilding populations. If successful this often results in a need for additional management of the conflicts which then arise from larger predator populations. The result is that many wild species require ongoing, active management to negotiate the human-wildlife interface (Arroyo-Quiroz, García-Barrios, Argueta-Villamar, Smith, & Salcido, 2017; Lute, Carter, López-Bao, & Linnell, 2018). Saltwater crocodile populations in Australia have followed this pattern (Saalfeld, Fukuda Y., Duldig T., & Fisher A., 2016; G.J.W. Webb, 2014; Grahame J W Webb, 2021). Recovery of their populations has largely been tolerated due to the economic benefit derived from commercial skin and meat production, egg collection and tourism (Fukuda *et al.*, 2020; Joanen *et al.*, 2021). Wolves in Southeastern Norway and the French Alps in Europe, and the Midwestern and Western

United States of America have similarly rebounded after strict protection in recent decades, leading to conflicts between pro-wolf and anti-wolf "camps" that highlight different aspects of the wolf recovery history in attempts to influence management (Ruid *et al.*, 2009; Skogen, Mauz, & Krange, 2008; Smith & Peterson, 2021). Although economic valuation through nature's contributions to people is a popular approach to address such issues, it is likely to fail with regard to wild species management (Linnell *et al.*, 2020) because so many of the costs and benefits of wild species conservation are of an intangible nature, and not conducive to economic valuation. In addition, the distribution of costs and benefits vary widely by spatial scales (Linnell, 2015) and within different value domains (Arias-arévalo, Gómez-baggethun, Martín-lópez, & Pérez-rincón, 2018).

These complex trade-offs challenge governance structures. When decisions are likely to be controversial, it is essential that decision making processes maintain broad societal legitimacy by balancing inputs of diverse experts, key stakeholders and the public before making transparent decisions. It is important to consider not only the direct practical and economic impacts of human-wildlife conflicts but the wider social, cultural and political context within which these impacts occur and which co-constitute sustainable use (e.g., Linnell & Cretois, 2018; Linnell *et al.*, 2020; Lüchtrath & Schraml, 2015; Skogen, Krange, & Figari, 2017). In order to attend to the increasing diversity of conflicting interests and objectives, existing management structures would require greater transparency, scientific robustness and social legitimacy. The integration of all these elements is more likely ensure successful co-habitation among humans and wild species can continue (Carter & Linnell, 2016).

Finally, eradication of invasive alien species, including invasive wild animals, is globally acknowledged as a key management option for mitigating the impacts they cause to biological diversity, economy and human well-being (Courchamp *et al.*, 2011, p. 2011; Genovesi & Carnevali, 2011; Simberloff, Parker, & Windle, 2005). Most of these eradications have been done on islands and involved vertebrates (Genovesi, 2005), but there are also examples of successful eradications of invertebrates, including fruit flies from Nauru (Allwood, Vueti, Leblanc, & Bull, 2002), mosquito *Anopheles gambiae* from Brazil (Davis JR & Garcia R, 1989), and the Asian Gypsy Moth in North America (Elkinton & Liebhold, 1990). In Europe, rats (*Rattus* spp., 67% of all eradications) and rabbits were the most common target species (Genovesi, 2005). Although effective, possible cascading ecological effects of eradications must be taken into account (Courchamp *et al.*, 2011, p. 2011).

3.3.4 Logging

3.3.4.1 Introduction

Logging practices differ widely around the world. These include felling of individual wild trees, selective timber-harvesting, clearcutting and variable retention harvesting. The broader the group of forest users, the more likely logging needs to be reconciled with other uses and services, which support very diversified and complex livelihood strategies (Zenteno, Zuidema, de Jong, & Boot, 2013). This section assesses the status and trends of logging in the relation to the sustainable use of wild tree species. Due to the relative complexity of grouping all logging practices together, in this introduction several topics relevant to logging are briefly introduced. This includes the formal definition from Chapter 1, the issue of plantation vs. natural forests, and how forests are classified and forest management defined. A more detailed overview of the global status and trends of forests and forest management is presented in the following section (3.3.4.2). Section 3.3.4.3. is a review of timber products and uses structured similarly to the other uses sections in section 3.3. Finally, emerging issues are discussed in section 3.3.4.4. Direct and indirect drivers of use and sustainable use are discussed in detail in Chapter 4.

The review on key aspects of sustainable use focusing on logging practices relies heavily on meta-analyses carried out by either independent academic scholars or in affiliation to forest-based research departments or institutions including the FAO, the Center for International Forestry Research and the International Tropical Timber Organization (see the data management report for Chapter 3 systematic literature review at https://doi.org/10.5281/zenodo.6452651). Due to the peculiarities of the forest management and logging practices across and within the same biomes and regions, the analyses are further supplemented with a limited number of country specific case-studies. A review of the available relevant scientific literature is also included. Reports from national forest management departments, case study reports and academic theses at both masters and doctoral levels were also used as appropriate (see the data management report for Chapter 3 systematic literature review at https://doi.org/10.5281/zenodo.6452651).

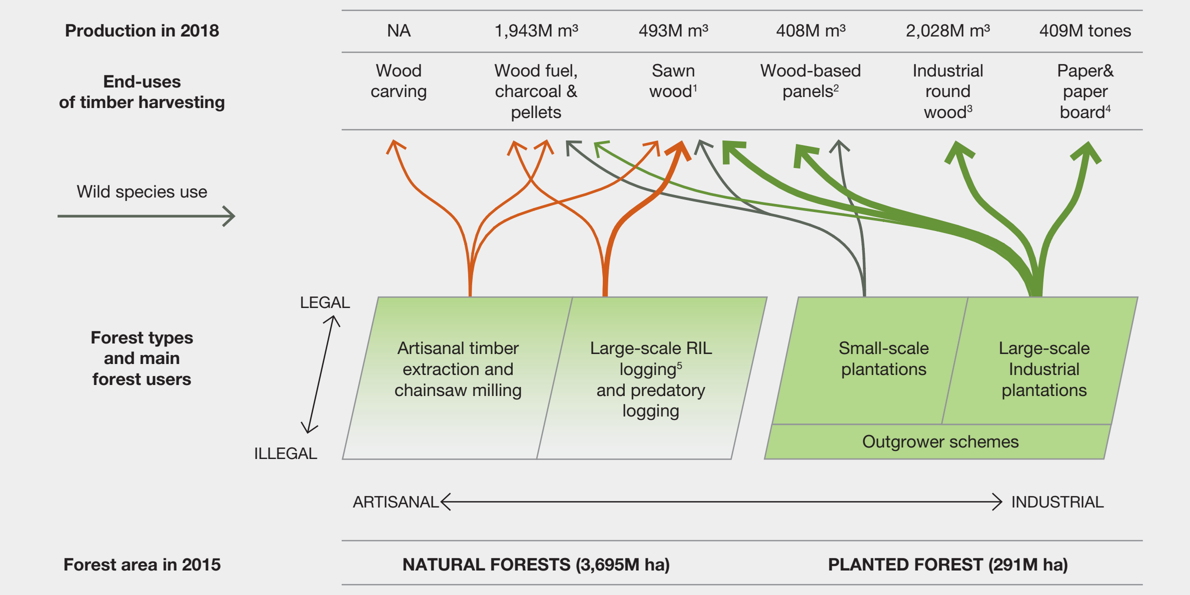
Logging is defined in this assessment as the removal of whole trees or woody parts of trees from their habitat. Logging generally results in the death of the tree, but also includes cases in which it may not, such as coppicing. Harvesting of non-woody parts of trees, such as fruits, bark or leaves, is considered under gathering (See Chapter 1 for definition, 3.3.2 for gathering). Logging is a key aspect of forest management, guided by site-specific requirements and prescriptions set out in forest management (and harvest) plans or through long-standing practices. It occurs in varying land tenure conditions including private, communal, and public ownership, and in forests ranging from simple (few dominant species) to complex (multiple species). The practice can be carried out formally or informally at small to large scales, for different uses, and for subsistence and commercial benefits **(Figure 3.52)**.

Timber is obtained from both natural and planted forests **(Figure 3.52)**. Estimates suggest plantations provide one third to one half (500-800 million m³) of global industrial round wood (Jürgensen, Kollert, & Lebedys, 2014; Siry, Cubbage, & Ahmed, 2005), meaning that natural forests are still the major sources of timber globally. Widespread adoption of tree planting for industrial purposes began in the 1960s (Bull *et al.*, 2006; Evans, 2009; McEwan, Marchi, Spinelli, & Brink, 2020; Szulecka, Pretzsch, & Secco, 2014) to generate mainly industrial roundwood and reduce deforestation (FAO, 1967). However, while there are projected increases in the extent and volume of wood that will be produced from plantations (Armesto, Smith-Ramirez, & Rozzi, 1999; C. Brown, 2000; FSC, 2012), their relative contribution is projected to decrease as demand increases (Carle & Homgren, 2008). Thus, the pressure on existing natural forests is expected to greatly increase in the coming decades, starting with the areas with easiest access.

Forests are classified under four climatic domains. The largest domain is tropical, constituting 45% (1834 million ha) of the world's forests, followed by boreal (27%) (1110 million ha), then temperate with 16% (666 million ha) and lastly the subtropical domain that constitutes 11% (449 million ha) of the world's forests (FAO, 2020a) **(Figure 3.53)**. For the purposes of this assessment, tropical and subtropical are at times referenced together and temperate and boreal are at times referenced together.

Widespread changes in forest types are more evident in tropical forests (Fearnside, 2004; Malhi & Phillips, 2004; Root *et al.*, 2003), which are more sensitive to climate changes (Hughen, Eglinton, Xu, & Makou, 2004) and have been greatly affected by loss of forest cover and forest degradation. These changes affect the ability of species to migrate and can lead to extinction of some species (Pounds *et al.*, 2006; Pounds, Fogden, & Campbell, 1999). The subtropics contain some of the most prominent biodiversity hotspots in Latin America, Australia, and South Africa, however many forest tree species exist in highly fragmented environments and are at particular risk of extinction (Locatelli, Brockhaus, Buck, & Thompson, 2010).

Temperate forests are the most extensively altered forest biome due to global change factors, with a smaller fraction of original vegetation remaining compared to boreal and tropical forests (Reich & Frelich, 2002). More changes in vegetation type are anticipated over the next 70-100 years



(1) Encompasses planks, sleepers (cross-ties), beams, joists, boards, rafters, scantlings, laths, and boxboards
(2) Comprises plywood, particleboard
(3) Is all roundwood used for any purpose other than energy. It comprises pulpwood, sawlogs and veneer logs
(4) Comprises of graphic papers (newsprint, printing and writing paper) and other paper and paperboard.
(5) RIL = Reduced Impact Logging

Figure 3 52 **Flow diagram of timber products from natural and plantation forests.**

Based on Global Forest Products: Facts and Figures 2018 (FAO, 2019a) under license CC BY-NC-SA 3.0 IGO.

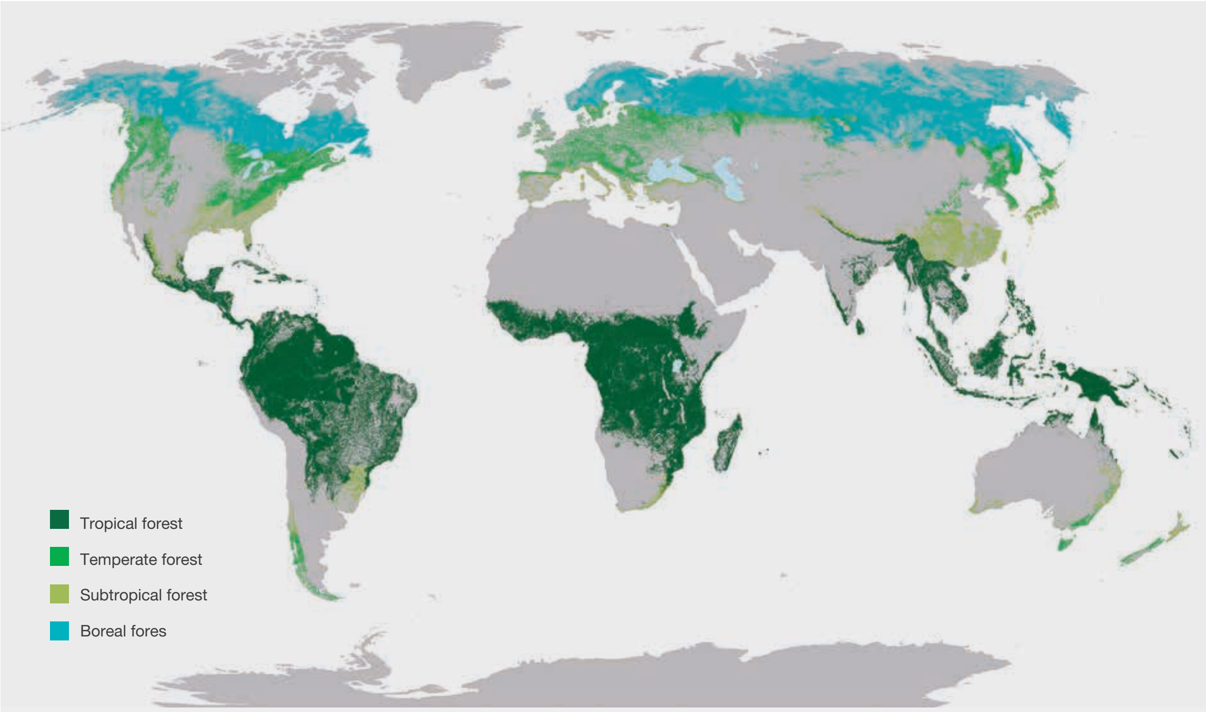


Figure 3 53 **Global distribution of forests sub-divided by climatic domains.**

Red: tropical, purple: subtropical, green: temperate, and blue: boreal.
This map is adapted from its original source (FAO, 2020a) and is copyrighted under license CC BY-NC-SA 3.0 IGO. The designations employed and the presentation of material on the maps used in the assessment do not imply the expression of any opinion whatsoever on the part of IPBES concerning the legal status of any country, territory, city or area or of its authorities, or concerning the delimitation of its frontiers or boundaries. These maps have been prepared or used for the sole purpose of facilitating the assessment of the broad biogeographical areas represented therein and for purposes of representing scientific data spatially.

(Locatelli *et al.*, 2010), though with a high degree of uncertainty due to interactions among increased fire, invasive species, pathogens, and storms (Virginia H. Dale *et al.*, 2001). It is reported that temperate and boreal forests are expanding northwards, a trend expected to continue due to climate change (Chamberlain, Emery, & Patel-Weynand, 2018; Locatelli *et al.*, 2010). Models suggest boreal forests will also undergo increased fires, increased insect and disease infestations, altered stand composition and structure. Declines in old-growth forests and conversion of southern-central dry forests to grasslands are also predicted due to climate change over the next several decades (Locatelli *et al.*, 2010).

3.3.4.2 Global trends and overview

Logging and trade in timber products has increased over the last several decades due to land use change including conversion to agricultural lands, transition to timber plantations and urban development, leading to deforestation and forest degradation (Estrada, Garber, & Chaudhary, 2019; Hosonuma *et al.*, 2012; Kissinger, Herold, & De Sy, 2012; Miller, Mansourian, & Wildburger, 2020; Ngansop, Biye, Fongnzossie, Forbi, & Chimi, 2019). According to the Food and Agriculture Organization of the United Nations (2020a), forests decreased from 32.5 percent to 30.8 percent of the global area between 1990 and 2020, representing a net loss of 178 million hectares (FAO, 2020a) **(Figure 3.54)**. Africa had the highest net loss of forest area between 2010–2020, with a loss of 3.94 million hectares per year, followed by South America with 2.60 million hectares per year. Asia showed the highest net gain in forest area in the period 2010–2020 (FAO & UNEP, 2020), however this is attributed to expanding already extensive plantation forests (Paradis, 2020; Sloan, Meyfroidt, Rudel, Bongers, & Chazdon, 2019; Szulecka *et al.*, 2014) **(Figure 3.54)**.

Primary forests, which are defined as naturally regenerated forests of native species (FAO, 2018c) have reduced by 81 million ha. since 1990, though the rate of loss decreased by over 50% between 2010–2020. Forests with high ecosystem integrity remain in Canada, Russia, the Amazon, Central Africa, and New Guinea (Grantham *et al.*, 2020). Ecosystem integrity here refers to the degree to which a system is free from anthropogenic modification of its structure, composition, and function (Parrish, Braun, & Unnasch, 2003). The majority of remaining forest areas have moderate to low forest ecosystem integrity as a result

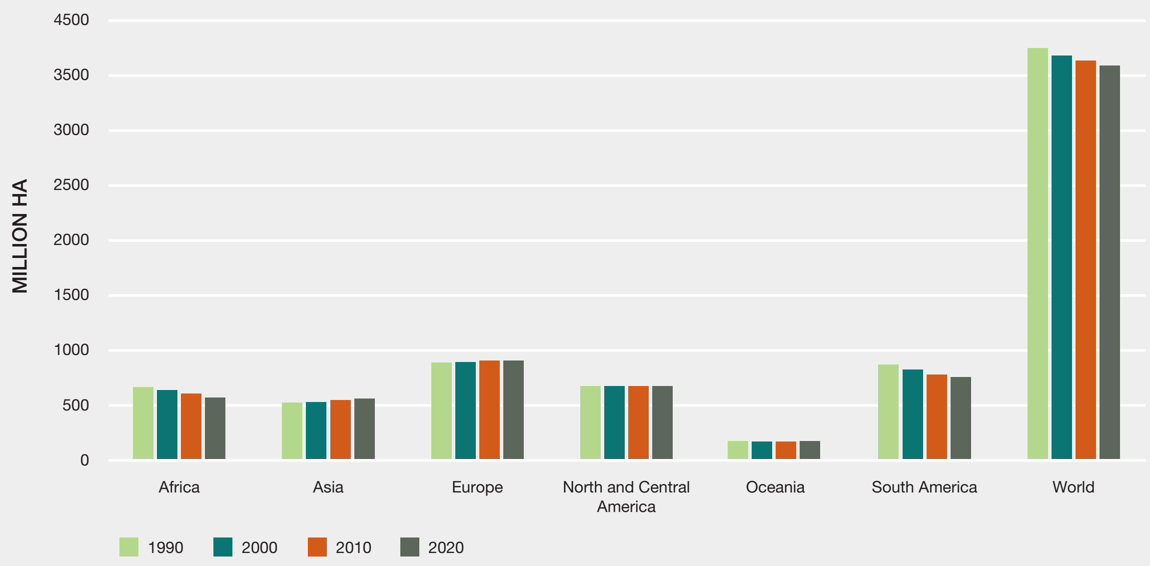


Figure 3 54 **Forest area by region, from 1990 to 2020.**

Data from the Global Forest resource assessment (FAO, 2020a) under license CC BY-NC-SA 3.0 IGO. See data management report for the figure at https://doi.org/10.5281/zenodo.6453095.

of human use of forest systems, affecting the capacity of forests to provide benefits. This degradation can be a precursor to outright deforestation (Grantham *et al.*, 2020; McNicol, Ryan, & Mitchard, 2018).

Planted forest cover increased by 123 million ha between 1990 and 2020, although rates of increase have slowed since 2010 (**Figure 3.55**, also see Supplementary material Table S3.2). In 2020, plantations and other planted forests equaled 294 million ha (7%) of the world's forest cover. Asia has the largest proportion of planted forests, 135 million ha, which constitute 22% of the region's total forest cover. Approximately 44% of plantation forests feature introduced species. Native species are mainly planted in North and Central America (96%) and Asia (68%) while the percentage of plantation forests comprised of native species are 30%, 23%, 22% and 3% in Africa, Europe, Oceania and South America respectively (FAO, 2020a).

Forest management objectives can be categorized under production, protection of soil and water, conservation of biodiversity, social services, multiple use, and other uses **(Table 3.16)** (FAO, 2020a). Approximately 1.15 billion ha., accounting for 31 percent of the world's total forest area is managed for production purposes, that is for timber, fiber, bioenergy and/or wild plants and fungal products. The area has however slightly decreased by 1.22 million ha between 1990 and 2020 with some fluctuations. Decreases in production forests occurred in Europe, Asia and most significantly in Africa (from 109 to 91.4 million ha) with a corresponding decrease in forest area. North America and Oceania had slight increases in forest area under production during the same time period. Concurrently, approximately 749 million ha (22% of total forest area) of forest globally are designated primarily for multiple use. This total decreased by 70.7 million ha between 1990 and 2020 in all regions except Asia and Europe (FAO, 2020a).

Selective harvesting is one of the dominant logging practices that contributes nearly 15 percent of global timber needs (P. A. Martin, Newton, Pfeifer, Khoo, & Bullock, 2015; Poudyal, Maraseni, & Cockfield, 2018). Selective harvesting can be low impact timber-harvesting when it involves harvesting 1-2 species and 1-2 individuals per hectare. It can be moderate when 5-15 species are harvested and 1-3 individuals per hectare (Uhl *et al.*, 1997). The practice is done on a large scale through mechanized tree extraction or on a small scale through manual extraction (Rendón-Carmona, Martínez-Yrízar, Balvanera, & Pérez-Salicrup, 2009). The practice can also be carried out by local people who hand harvest wood in exchange for staples, while large distant companies do the wood processing.

Globally selective logging is practiced on about 20.3% (3.9 million km^2) of humid tropical forests (Asner, Rudel, Aide, Defries, & Emerson, 2009). Selective logging is considered unsustainable when it is carried out the conventional way without measures to reduce damage to the residual forest stand. It is considered sustainable when specific planning and techniques are used to minimize damage to the residual stand. Several of these techniques are included in guidelines which are referred to as Reduced Impact Logging (RIL) (Arets *et al.*, 2011; Dykstra & Heinrich, 1996; Pinard, Putz, Tay, & Sullivan, 1995; F. E. Putz, Sist,

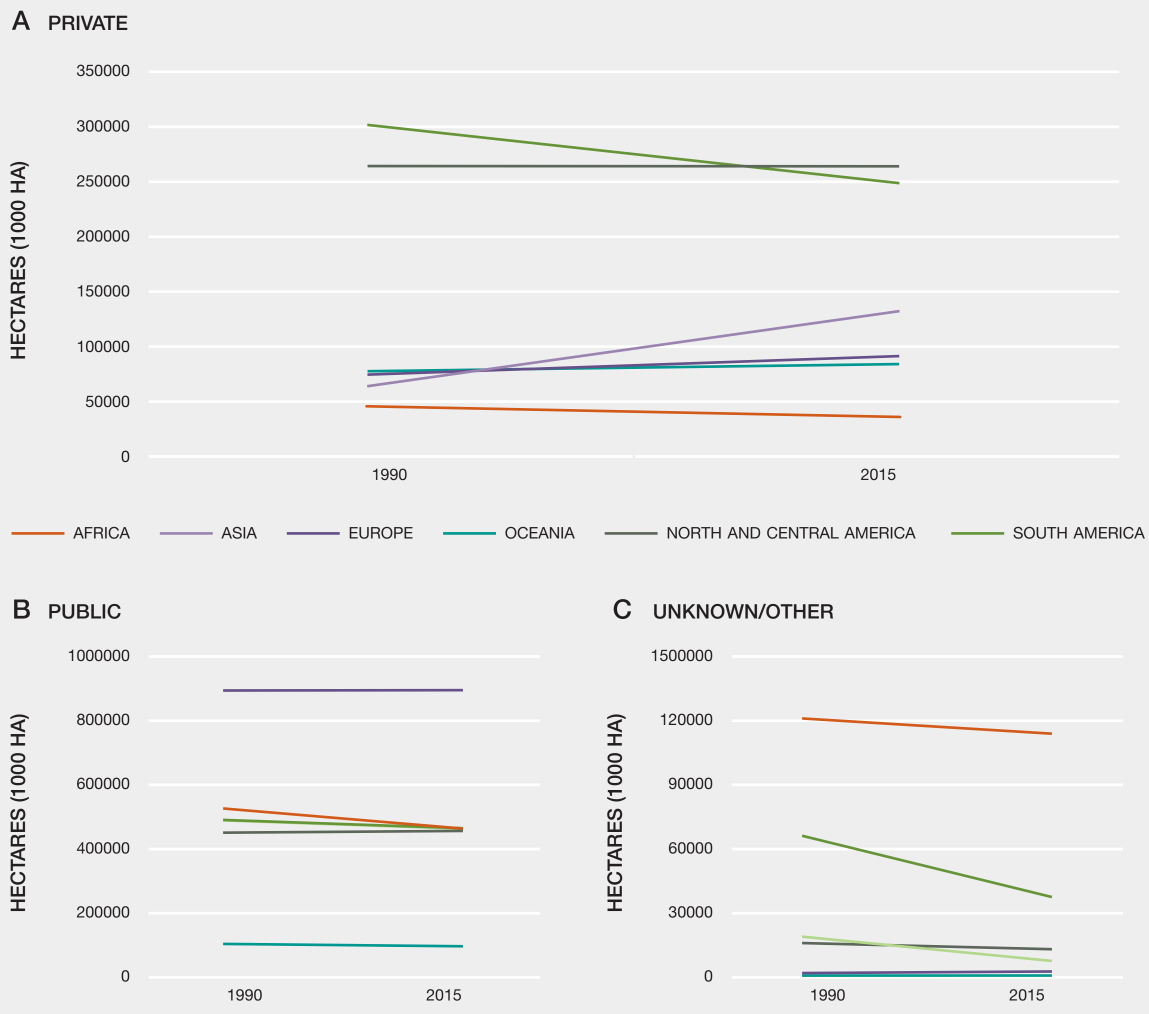


Figure 3 55 **Changes in global planted forest cover between 1990–2015.**

(A) Changes in privately owned forest land. The figure shows an overall four percent (4%) increase in privately owned forest lands, with decreases in Africa and South America, no change in North and Central America, and increases in Asia, Europe and Oceania. **(B) Changes in publicly owned forest land.** Figure shows an overall decrease of four percent (4%), with decreases in Africa, Asia, Oceania and South America, no change in Europe and a slight increase (1%) in North America. **(C) Changes in forest land with unknown ownership.** Figure shows an overall decrease of 22% in forest cover on lands with unclear ownership. Large decreases are observed in Asia and South America with lesser decreases in Africa, North and Central America. There was a slight increase in Oceania and a larger increase in Europe. Source: Global Forest resource assessment (FAO, 2020a) under license CC BY-NC-SA 3.0 IGO. See data management report for the figure at https://doi.org/10.5281/zenodo.6453095.

Fredericksen, & Dykstra, 2008). Reduced impact logging is implemented at the operational level by planning skid trails, practicing carefully controlled felling and skidding, and reducing damage to soils and residual trees (Sist and Ferreira, 2007). Implementation of reduced impact logging is still limited (Arets *et al.*, 2011; P. A. Martin *et al.*, 2015) and conventional logging practices continue to dominate (F. E. Putz, Dykstra, & Heinrich, 2000). Among the major factors hindering adaptation of reduced impact logging is the expense in comparison with conventional timber-harvesting (F. E. Putz *et al.*, 2000, 2008). In addition, evidence that reduced impact logging achieves the desired objectives is also contradictory. Some studies do suggest a reduction in the negative impacts of logging activities when reduced impact logging guidelines are followed (Bicknell, Struebig, Edwards, & Davies, 2014; R. Pereira, Zweede, Asner, & Keller, 2002; Putz *et al.*, 2012; T. A. P. West, Vidal, & Putz, 2014). However, other studies suggest that positive effects of reduced impact logging are in fact more closely related to differences in harvesting intensity (Griscom, Ellis, & Putz,

Table 3 16 **Forest area (1000 ha) designated primarily for production, and annual change, 1990–2020.**

Source: Global Forest resource assessment (FAO, 2020a) under license CC BY-NC-SA 3.0 IGO.

	Year				Annual change		
	1990	2000	2010	2020	1990–2000	2000–2010	2010–2020
Forest area	4236433	4158050	4106317	4058931	-7838	-5173	-4739
Planted Forests	170061	210662	261958	292587	4060	5130	3063
Forest area with long-management plans		1757831	1855538	1990865		9771	13533
Forests in protected areas	437 821	499 853	600 845	629 139	6 203	10 099	2 829
Forest designated for Production	1 135 826	1 112 657	1 097 126	1 134493	-2 317	-1 553	3 737
Forest designated for multiple use	809 181	780 458	750 728	738 464	-2 872	-2 973	-1 226

2014; Johns, 1992; Picard, Gourlet-Fleury, & Forni, 2012; Sist, 2000; Sist, Fimbel, Sheil, Nasi, & Chevallier, 2003; Sist, Nolan, Bertault, & Dykstra, 1998). Reduced impact logging provides guidelines to reduce environmental impacts of logging, however its lack of specificity regarding intensity can sometimes result in perverse effects. Therefore, more research is recommended to clarify whether reduced impact logging should be practiced in a way which incorporates harvesting intensity (Martin *et al.*, 2015).

Low intensity harvesting that is recommended in reduced impact logging may encourage expansion into previously unlogged areas in order to distribute the impact more widely (Martin *et al.*, 2015). In addition, the recovery rate of commercial trees after reduced impact logging is very low. One study in tropical rain forests, revealed that only 50% of the commercial stand was predicted to recover after a period of 30 years, creating a major reduction in stock for the next harvesting cycle in that area. This is not compatible with sustainable yield production on a long-term basis (Sist & Ferreira, 2007). Sist and Ferreira (2007) suggest that more sophisticated silvicultural systems are required to ensure sustainable management of the forests on a long-term basis. There is evidence suggesting that reduced-impact logging practices, if actually employed, could increase future timber yields (Griscom *et al.*, 2014; Johns, 1992; Picard *et al.*, 2012; Sist, 2000; Sist *et al.*, 2003, 1998). There is not much change on the ground in spite of these recommended practices (Putz, 2018).

The status of illegal logging and associated timber trade as well as its trends in harvesting practice, constitute complex and serious challenges in the sustainable use of wild species (J. Liu, Yong, Choi, & Gibson, 2020). Although illegal timber trade and unsustainable logging that threaten sustainable use are rampant and not well documented, it is promoting large-scale forest destruction, especially in the tropics (Laurance, 2004). The Illegal timber trade is highly international, which may result in substantial loss of large old trees. Owing to the higher prices of timber in India and China, smugglers are motivated to export timber from Nepal to the Tibetan Autonomous Region of China (Chaudhary, Uprety, & Rimal, 2016); and Nepal-India border (Chaudhary *et al.*, 2016).

Regarding data on logging, a common understanding is that it is hard to obtain accurate data on the scope of illegal logging. Scientific studies as well as reports present conflicting views on whether illegal logging is declining or not (Kleinschmit, Mansourian, Wildburger, & Purret, 2016). According to Hoare (2015), there has been important progress made in reducing illegality in the forest sector over the last decades. However, another report published three years earlier claims that illegal logging has remained high in many regions, even increased in some areas, and become more advanced with better organized activities also comprising criminal activities (Nellemann, International Criminal Police Organization, & GRID--Arendal, 2012). China (importing more than 50%, of total illegal export value), Vietnam, India, the European Union, Thailand and the United States of America are among the major importers of illegal timber accounting for 84% of the total value of imports. Southeast Asia (mainly Cambodia, Laos, Myanmar Indonesia and Malaysia), the Russian Federation, Papua New Guinea and the Congo Basin (Democratic Republic of Congo, the Republic of Congo and Cameroon are among the main exporters with Southeast Asia accounting for 55% of the exports (Chaudhary *et al.*, 2007).

There has also been an observed geographic shift in illegal logging and related timber trade. Illegal logging in Brazil, Indonesia and Malaysia has declined in recent years (Hoare, 2015). After decades of conservation efforts, forests along the China-Russia border have been recolonized (Wang *et al.*, 2016). In recent years the smuggling of timber as well as other forest resources has declined along Nepal-China and Nepal-India borders due to improved monitoring and collaborative transboundary conservation (Chaudhary *et al.*, 2016; personal communication with Bishnu Lama, indigenous people and local community member and chairman of the Namkha rural municipality – Humla District, Nepal, December 2020). However, Russia, other Southeast Asian countries (e.g., Cambodia, Laos and Myanmar), Papua New Guinea and some African countries have also witnessed increases in illegal forest activities (Guan *et al.*, 2016). Russia (primarily in its Far East region) is one among rising timber producer countries and exports timber mainly to China (Guan *et al.*, 2016). China has become the world's largest importer of tropical timber since a ban on domestic logging was implemented in 1998. It is also a key processing country, for example, it is the leading manufacturer of furniture worldwide, occupying 40 percent of the global market share (Richer, 2016); much is exported to the United States of America and Europe (Tacconi *et al.*, 2016).

Commercial logging is illegal in Afghanistan which leaves a massive smuggling industry to satisfy international demand. Local communities have lost control over the resources on which they depend for their survival, and forest resources are now largely used for immediate profit by organized crime syndicates and traders (Milbrandt & Overend, 2011). Additionally, poor forest management, lack of incentives for reforestation, lack of community involvement and awareness, and agricultural and urban encroachments on forest land also contributed to the severe decline of forest cover in Afghanistan (Milbrandt & Overend, 2011). The results have been that rangelands have deteriorated, forests have been felled, and wild species populations have greatly diminished from uncontrolled hunting and habitat degradation (UNDP, 2014).

In 2006, an executive order that was issued by then President Hamid Karzai banned illegal timber-harvesting and felling of trees and shrubs in natural forests in Afghanistan. After that, Afghanistan's Forest Management Law, passed in 2012, declared natural forests and woodlands as public property owned by the national government. The law also has a provision to support community-based forest management, allowing indigenous communities to utilize and manage the forest in collaboration with the Department of Natural Resources. However, the deterioration of overall law and order situation in Afghanistan means that the 2012 forest law has only been partially implemented.

Curbing illegal timber extraction and trade poses special challenges because of the need for cooperation among sovereign states. In order to support producer countries, bilateral arrangements have emerged, either between neighboring countries or between primary export and import countries (Kleinschmit *et al.*, 2016). Imports of illegal tropical hardwood timber in China with the republic of Congo, Ghana, Papua New Guinea, Laos, Brazil and Malaysia; India with Brazil, and Paua New Guinea; Japan with Republic of Congo, Cameroon, Malaysia and Paua New Guinea; and South Korea with Malaysia are considerable (Z. Guan, Chen, Xu, & Liu, 2020). Bilateral actions that also include transboundary cooperation have been initiated at the national level (Tacconi *et al.*, 2016). Besides scientists, transboundary conservation deserves more attention from policymakers too (Liu *et al.*, 2020). Policy in one country can easily have a major impact in other countries. For example, some research suggests that logging bans in Thailand and China have led to increased logging and forest loss in the neighboring countries including Lao People's Democratic Republic, Cambodia, Indonesia, the Russian Far East and Mongolia (Fisher, Maginnis, Jackson, Barrow, & Jeanrenaud, 2008). Hence, there is need to further strengthen international cooperation and domestic legislation in order to control the imports of illegal timber, enhance the protection and cultivation of forest resources and reduce dependence on imported timber (Guan *et al.*, 2020).

The spread of illegal logging and other forest crimes into protected areas occurs because valuable timber is still available in commercial volumes (Wardojo, Suhariyanto, & Purnama, 2001). Timber felling in protected areas in Indonesia involve multiple stakeholders, including local people, logging companies, military personal and forestry officials (Barber & Talbott, 2003; Hiller *et al.*, 2004; Laurance, 2004; McCarthy, 2002; Ravenel, 2004; Robertson & van Schaik, 2001). Illegal logging provides immediate income for local communities and may aid in day-to-day survival (Schroeder-Wildberg & Carius, 2005). In some places illegal forestry activity is a function of local livelihood context such as reduced income from farming (Yonariza & Webb, 2007).

3.3.4.3 A stratified typology on sustainable use of wild species in logging

Forests are owned either publicly by the state for the benefit of the citizens or privately by individuals, local, tribal and indigenous communities, or business entities and institutions. The proportion of forests under public ownership has declined, while those under private ownership increased between 1990 and 2015. In all regions, public administration holds management rights to most of the publicly owned forests. Globally, individuals own most privately owned forests, followed by local, tribal and indigenous communities and the least are owned by business entities and institutions (FAO, 2020a) **(Table 3.17)**.

Table 3 17 **Management of forest area under private and public ownership.**

Source: Global Forest resource assessment (FAO, 2020a) under license CC BY-NC-SA 3.0 IGO.

Region/ subregion	Area of forest in three types of private ownership, by region, 2015 (1000 ha)			Holders of management rights to public forests, by region, 2015 (1000 ha)				
	Individuals	Local, tribal and indigenous communities	Business entities and institutions	Public administration	Individuals	Local, tribal and indigenous communities	Business entities and institutions	Unknown/ other
Africa	824	15599	1978	378849	0	7104	41485	844
Asia	7196	3900	1742	323232	45	30245	1275	40052
Europe	50946	2535	11691	641273	1	1324	244003	809
North and Central America	129468	45579	59723	389302	202	5570	54882	2956
Oceania	160	37551	0	6728	0	0	278	0
South America	0	3491	144	435192	2014	7173	5925	3
World	**188592**	**108655**	**75279**	**2174576**	**2263**	**51416**	**347848**	**44664**

Features of logging activities vary depending on the specific contexts in which they develop. Land tenure, the level of access to public infrastructure (e.g., roads, energy, health, education) and proximity to markets are all important structural conditions. Ecological conditions including stand composition, seasonality, and soil types also affect harvesting conditions. In all these cases, logging may apply technologies that range from artisanal, often manual and carried out with or without permits by individual small-scale millers, to industrial operations with highly mechanized large scale tree removal. To try to account for these variables, the status and trends of logging operations have been analyzed using a three-element typology which generally corresponds to the scale of volume harvested and size of harvest area **(Table 3.18)**. Specific actors have also been associated with these categories. In increasing volume and area, these are identified as: (1) smallholder, (2) community and (3) industrial logging operations:

1. **Smallholder forestry**, where logging is undertaken by individuals or family groups in lands on which they hold individual private access to forests and timber

2. **Community logging or community forestry**, where logging is organized and carried out collectively in forests stocking on community lands, using either artisanal techniques or externally supported reduced impact logging with heavy machinery

3. **Industrial Logging**, where individual companies holding individual or long-term concession rights conduct either conventional or reduced impact logging.

These logging operations are further differentiated by key aspects of use, identified here as harvest regime, governance, and economy:

1. **Harvesting regime** refers to the species harvested, species characteristics which affect volume of harvest such as growth and regeneration rates, and the techniques and equipment used.

2. **Governance** refers to different forms of access to forests and timber. It also refers to individual and collective rights, which range from diffuse and well-defined customary rights to full formal ownership of private lands and long-term usufruct rights in public lands. Legality is also considered a governance issue.

3. **Economy** refers to ways in which benefits are accumulated by actors. These include subsistence needs, and produce goods for the formal and informal economies. There are differential distributions of benefits depending on the form of capital accumulation and capital distribution.

3.3.4.3.1 Smallholder Logging practice

Estimates suggest that 1.3 billion people live in or around the world's remaining forests (Chao, 2012). These include right holders with individual and collective access to forests, either formal or informal. Rights holders may include individual landowners, indigenous traditional communities, local communities with established land tenure and historical access, and naturalized immigrant communities (for example from settler colonial expansion). In many cases

Table 3 18 **Typology of logging systems.**

Actors = Social entities organizing logging operations. **Harvest regimes** = equipment used, volume harvested, species, age class, size, return interval, regeneration, etc. **Governance** = customary and formal norms (including cultural knowledge and principles), rules, and regulations, management plans. **Economy** = subsistence, informal trade, formal trade; harvest to consumption value chains, distribution of benefits, and capital accumulation.

	Actors	Harvest regime	Governance	Economy
Aggregate	All	3.3.4.2	3.3.4.2	3.3.4.2
Smallholder	Individual or collective	3.3.4.3.1.	3.3.4.3.1.	3.3.4.3.1.
Community	Collective	3.3.4.3.2.	3.3.4.3.2.	3.3.4.3.2.
Industrial	Individual or corporation	3.3.4.3.3.	3.3.4.3.3.	3.3.4.3.3.

different individuals and communities may co-exist in the same locations. For example, in the Brazilian Amazon, traditional dwellers are comprised by *caboclos* or local people who descended from immigrants who followed the several waves of resource exploitation into the region and mixed with indigenous residents (Adams, Murrieta, Neves, & Harris, 2009). In the north central United States of America, the indigenous Menominee and Ojibwe peoples manage their tribal forests independently of the surrounding state and federally owned lands (Mausel, Waupochick, & Pecore, 2017; Ronald L. Trosper, 2012; Waller & Reo, 2018). Naturalized communities and migrants are more recent arrivals into forest zones who settled spontaneously or followed government-sponsored programs (B. M. Fernandes, 2004). In Southeast Asia, immigrants followed state-driven immigration programs but also followed the development of plantations that attracted rural labor to forest landscapes (Budidarsono, Susanti, & Zoomers, 2013). All these local groups undertake some type of small-scale logging, along with landless people trying to make a living, a portion of which may carry out logging operations on smallholder lands through different arrangements.

Smallholder plot sizes range widely across world regions. Plots in more remote areas tend to have more independent logging activities. For example, 60% of family forest owners in the United States of America have an area ranging between 0.4-4.0 ha (Snyder, Butler, & Markowski-Lindsay, 2019). Many smallholder farmers in the Amazon have access to larger pieces of land of up to roughly 100 ha (Siegmund-Schultze, Rischkowsky, da Veiga, & King, 2007) (Budiman, Fujiwara, Sato, & Pamungkas, 2020). In the Amazon, the more remote farms are, the higher the probability that they still have some primary forest remnants stocking their property. These remote farmers often operate more independently regardless of the status of their tenure (Serra, 2020). In the Amazon, most of the forests on the land occupied by immigrant smallholders are already degraded from fires or former harvesting by commercial loggers. Smallholders may harvest trees from their plots, but, due to the immense logistical and legal challenges, this is rarely carried out for commercial purposes. Accordingly, for most farmers, forests are a reserve for agricultural land, or provide materials for subsistence needs (e.g., fences, fuel wood) (Pacheco, 2009). If marketable timber is still available, they may also extract trees for commercial purposes when quick cash is needed (Pokorny, 2013). While smallholder farmers selectively extract high-value timber from remnant forests, they may also sell timber that originated from secondary forests emerging in agricultural fallows, often to local markets. At the same time, growing trees is an essential component of most smallholders' production systems (Hoch, Pokorny, & de Jong, 2012). Accordingly, over time, landscapes occupied by smallholders develop complex land-use mosaics that include swidden fields, fallows, agroforestry plots and forest patches (Denevan & Padoch, 1987; Padoch & Pinedo-Vásquez, 2006).

The small-scale logging industry is characterized by stakeholders that may or may not have a felling permit, often use chainsaws (sometimes mobile saw) for felling and processing in the forest, have smaller numbers of trees per operation, often produce lower quality sawnwood for national market and neighboring countries and is largely informal (Cerutti & Lescuyer, 2011). In addition to chainsaws, winches and canoes with outboard motors are often used when water transport is involved. Chain saw milling requires a relatively small investment as the equipment is readily available and inexpensive to buy or rent, is portable and efficient (Pinard *et al.*, 2006). Among the products of chainsaw milling are boards and planks for personal use, those that are sold directly to the market, and blocks or scantlings that are further processed in sawmills (Wit, van Dam, Omar Cerutti, Lescuyer, & Mckeown, 2010). The logging team consists of a few individuals who could be part of an entitled community or recruited from elsewhere, and they may own their own equipment or operate equipment owned by others (Salo, Sirén, & Kalliola, 2013). They harvest significantly smaller volumes of timber. The practice is usually very selective, concentrating only on the most valuable commercial species such as, in the tropics, mahogany, cedar, teak and *tornillo (Cedrelinga catanaeformis).* Chainsaw systems persist especially in areas with more rugged terrain. Across the United States of America, one-third to over three-quarters of loggers used chainsaw felling (Conrad, Greene, & Hiesl, 2018). Wherever possible, small-scale chainsaw millers target large trees to maximize their output (Cerutti & Lescuyer, 2011).

Wood production by small-scale chain saw operators can be for personal use (Snyder *et al.*, 2019) and to supply domestic markets (Rozemeijer & Aggrey, 2011). In some instances, the wood is for social or community purposes and not sold or exchanged (Lesniewska & McDermott, 2014). When entering into formal markets, the timber is usually purchased by middlemen at cheaper prices who then sell it to the timber industries (Salo *et al.*, 2013). The industry is rapidly growing in tropical countries (Hoare, 2015), representing approximately 30-40% (in Guyana, Republic of Congo, Democratic Republic of Congo and Uganda), more than 50% (in Ghana, Cameroon and Peru), and almost 100% (in Liberia) of total timber trade (Wit *et al.*, 2010). However, wood for timber is only a small part of the total domestic market, with most locally traded wood in the tropics being used for fuel or made into charcoal (Wit *et al.*, 2010).

In many Amazonian countries (e.g., Bolivia, Peru, Ecuador), smallholders are allowed to extract timber from their properties for commercial purpose, yet they have to obtain permits, often through simplified processes including simpler management plans (Cerutti & Lescuyer, 2011) **(Box 3.15)**. That said, few smallholders, such as those in the Peruvian Amazon, have secured formal property rights (Cronkleton & Larson, 2015). For those with formal usufruct rights to households occupying forest lands, they may be able to register a formal forest management plan to carry out selective timber-harvesting (Robiglio, Acevedo, & Simauchi, 2015). In spite of the options allowing for the use of simplified plans, only a small portion of smallholders formally apply for forest permits (Pacheco, Mejía, Cano, & de Jong, 2016).

Informal logging by smallholders provides thousands of jobs in Central African countries. In the Congo Basin, countries have embraced forest policies that mainly targeted the sustainable management of timber in large-scale timber-harvesting concessions targeting export markets and overlooked small-scale production. Yet small-scale chainsaw milling, which is chiefly informal, has undergone rapid development to meet the domestic demand for cheap timber in Central African countries and other nearby countries, as well as the interests of stakeholders all along the chain of custody (Eba'a Atyi *et al.*, 2016). Over the last decade, in Central Africa, the annual volume of timber from informal chainsaw milling consumed domestically or unofficially exported to nearby countries is greater than that of timber from the industrial sector (Guillaume Lescuyer & Cerutti, 2013). In Cameroon, around 45,000 people find their main employment in this sector (Cerutti & Lescuyer, 2011). In the cities of Congo, the Central African Republic and Gabon, more than 1,000 people have jobs directly linked to the sale of small-scale timber production (Guillaume Lescuyer, Cerutti, & Robiglio, 2013).

Small-scale chainsaw milling is an important source of income for rural stakeholders, and accepted by urban consumers (Guillaume Lescuyer *et al.*, 2017), who gain access to materials at prices three to four times lower than those from industrial timber (Guillaume Lescuyer *et al.*, 2013). In remote areas, smallholders, when in need to harvest and sell timber often face distorted market conditions, mainly for two reasons. They may suffer from elevated transport costs, due to long distances, bad roads, and small quantities, or, to avoid logistical challenges depend on intermediaries or sawmill operators that tend to underprice the timber (Pacheco, 2012). In locations closer to the markets, smallholders who still dispose on forests, are better engaged to extend market networks managed by

Box 3 15 Smallholder logging in Ucayali, Peruvian Amazon.

There are approximately 440,000 smallholder producers (i.e., plots <115 hectares) in Ucayali region in the Peruvian Amazon, with approximately 80% holding less than 20 hectares of land (Robiglio *et al.*, 2015). It is estimated that mosaic production systems of these smallholders cover more the 4.5 million hectares in the Peruvian Amazon, with approximately 90,000 ha under fallow-forestry (Sears, Pinedo-Vasquez, & Padoch, 2014). Farmers in the Ucayali region of Peru produce 950,000 m^3 of sawn wood annually (Sears, Cronkleton, Polo Villanueva, Miranda Ruiz, & Pérez-Ojeda del Arco, 2018). This is based on production of 38 m^3 ha^{-1} of sawnwood on a stand of 7 years. Smallholder farmers extract timber from remnant standing forests but also from secondary forests growing from fallows. The main product from the fallow-forestry system is small dimension lumber from *Guazuma crinita* (Sterculiaceae) and *Calycophyllum spruceanum* and Rubiaceae (Sears *et al.*, 2014).

In tropical regions, forest concessions occupy more than 20% of public forests in west and central Africa and Southeast Asia and about 4% in Latin America. In the tropics 15% of forests are managed by communities (Arts & de Koning, 2017). All these communities manage forests through different socio-ecological systems with very peculiar characteristics that are associated with traditional knowledge and community identity.

intermediaries who organize the extraction in response to orders from end-buyers in the cities (Mejia, Pacheco, Muzo, & Torres, 2015). Main markets are for construction such as in Central Africa (Eba'a Atyi *et al.*, 2016), and the furniture industry such as in Jepara district, Indonesia **(Box 3.16)**. In some places in the Amazon, the broader value chain for small-dimension lumber supports hundreds of other actors involved in the harvest, transport, transformation, and wholesale activities within marketing networks stretching from remote areas of the Amazon to major urban centers in Peru's coast and highlands (Pokorny, 2013; Sears *et al.*, 2018).

In the Amazon, for most smallholders, primary forests play only a little role for income generation, whenever wood products such as firewood, poles, or for construction are regularly used by the families (Porro *et al.*, 2014). In addition, forest fallows have a potential to generate income if the production areas are located near to roads and markets. In such conditions, farmers benefit from additional incomes from selling wood ranging from \$35 to \$1870 United States dollars per hectare (Hoch *et al.*, 2012; Sears *et al.*, 2014), with multiplier income effects associated with the processing. In Central Africa, chainsaw milling also constitutes an important source of employment for rural population, which translates into a relatively regular income stream on the lack of other job opportunities (Eba'a Atyi *et al.*, 2016).

In the Amazon, much timber harvest takes place at the scale of individual households, even within communal properties granted to indigenous communities (Cronkleton & Larson, 2015). The smallholders may harvest the timber themselves with chainsaws and then process the log to produce planks, which are easier to transport, they hire specialized loggers (Pokorny, 2013). Yet, this informal practice is generally penalized by law, except in some countries like Ecuador (Sears *et al.*, 2014). In some cases, smallholders sell standing timber to professional loggers that have better connection with sawmills, which often approach a larger number of farmers so to compensate for the use of heavy machinery (Mejia *et al.*, 2015). In the Peruvian Amazon, commercial harvest of fallow timber is done with a chainsaw portable mill with a circular saw set up on the farm for *in situ* primary transformation. It is often the case that the rough-hewn planks are planed into the finished product in either lumber yards or workshops in the urban centers.

Systems for permitting are quite different and greatly vary in many countries in temperate zones, where land tenure systems tend to be more closely regulated with regards to private vs. public ownership. In cases of private ownership, there is variation in the freedom to decide the amount of timber to harvest, approvals required to harvest and freedom of owners to perform the actual harvesting (Nichiforel *et al.*, 2018). Freedom to decide the amount of timber to harvest can be based on a framework of general silvicultural restrictions (e.g., Norway, Austria, United Kingdom of Denmark, Ireland, Latvia, Portugal and Sweden) and size/quantity provided for in the legislation. For example, in France the forest owners maximumly harvest 50% of the standing timber on their property in comparison to

Box 3 16 The furniture industry in Indonesia.

Furniture is an important industry within the forest-based sectors for several reasons. First, micro, small and medium enterprises play a significant role in creating employment. The furniture sector provides direct employment to approximately 500,000 individuals (Munadi, 2017). Second, the industry contributes significantly in terms of foreign exchange. In 2019, Indonesia exported 1.7 billion United States dollars from the furniture exports (Bank Indonesia, 2020). Third, the furniture industry also represents Indonesian identity in international markets since Jepara is a furniture producing district with global recognition as furniture and woodcraft center (Pujiati, 2017).

Ironically, the performance of the industry at the national level is not well-known, and national estimates regarding the size of the industry are based on limited data. The most valuable data comes from the Central Statistical Agency, which reported that by 2019, there were 145,000 furniture micro, small and medium enterprises in Indonesia (wooden and nonwooden-based), representing about 3.3% of the total sample of 4.4 million micro, small, and medium enterprises (BPS, 2020). A Ministry of Industry report shows that wooden-based furniture producers represent about 80% of the total furniture producers (Munadi, 2017; Pujiati, 2017). From this information and the Central Statistical Agency data, there was an estimated 116,000 wooden-based furniture producers in Indonesia.

A survey of furniture producers in Jepara and Pasuruan in 2020 by the Center for International Forestry Research (Dermawan, 2020) estimated that one producer uses about 71 m^3 of wood annually. Multiplying this number with the estimated total national producers, the wood consumption by the furniture industry in Indonesia could reach approximately 8.2 million m^3 of wood. A high segment of wooden furniture in Indonesia uses teak as the primary raw material. Teak is mainly available in Java and some areas in other islands, such as Sulawesi. With the mean annual increment of 10 m^3/ha/year (Kallio, Kanninen, & Krisnawati, 2012), meeting the need for 8.2 million m^3 of wood would require approximately 820,000 hectares of teak forests.

Estonia, where one can harvest 20m^3 per year and Bulgaria where one can harvest 10m^3 per year. These ranges are a result of forest management planning in combination with owners' decisions. In some cases, such as in Finland and Netherlands, more restrictions apply. In other countries owners are generally required to ask for approvals and adhere to the conditions of approval (Bulgaria, Greece, Romania). Approvals may be required when forest management plans do not apply (e.g., France and Czeck Republic) or when there are special circumstances such as exceeding a given size of clear cut. There is little regulation on private forests in the United Kingdom of Denmark, and in Estonia no formal approval is required for personal use. In the majority of the countries, forest owners have the freedom to cut down trees without any restrictions, others restrict the quantity an individual can harvest by him/herself (for example in Romania where owners can harvest less than 20m^3 without a permit). Several countries in Eastern Europe only grant licenses to individuals with harvesting skills, and others such as Greece require the owner to contract a special firm for harvesting (Nichiforel *et al.*, 2018).

Ecological impacts of small-scale logging range from minimal to long-term. In Central Africa, when smallholder logging is driven by demand, chainsaw millers may penetrate deeper into the forest, and apply more effective tools such as portable saws in order to meet with a growing urban demand (Cerruti *et al.*, 2017). Fallow-forestry allows the use of timber and other non-timber forest resources, while providing multiple contributions to people to regenerate soil fertility and conserve biodiversity (Pattanayak & Sills, 2001; Pyhälä, Brown, & Neil Adger, 2006). In such systems, smallholder farmers often conserve scarce timber species, such as *Cedrela odorata, Swietenia macrophylla,* and *Dipteryx* spp.), among others (Putzel, Padoch, & Ricse, 2013).

Due to the individualized living schemes of small-scale farmers in the Amazon, there are not many social impacts of forest management. However, natural and planted forests are frequently affected by accidental fires caused during field preparation, which further reduces the attractiveness of forest investments (Hoch *et al.*, 2012) and may lead to conflicts. Less frequent are wood robbery, and forest tenure conflicts in the remoter, less accessible forest parts of smallholder properties. Although not often discussed, smallholder logging often does not involve women in the operations, which may lead to some unequal distribution of benefits in the households undertaking logging, although women develop other activities in the farm and gardens (Colfer, 2005).

Thousands of households manage forest fallows and trees as part of their customary livelihoods strategy that meets both subsistence and income needs (Pokorny & De Jong, 2015). Smallholder logging is only sustainable when it is done for subsistence or at low intensities. It constitutes a complementary activity that is shrinking over time due to the expansion of agriculture. Even with forest fallows and re-growing secondary forest, tree species composition and tree growth are affected from soil degradation caused by agricultural uses and fire.

3.3.4.3.2 Community Logging practice

Community forest management involves the use and management of forests by communities. While community forestry often involves the management of large areas of forest relative to the average size of that managed by individual smallholders, the areas are still small relative to most industrial estates (100s of hectares compared with 1000s of hectares). Furthermore, the focus on multiple use management is strong in both community and smallholder forestry compared with the focus on timber production in industrial estates.

Forest areas that are owned or managed by local communities have been increasing in the last decades and account for up to 15% of total forest area worldwide (513 million hectares) (Putraditama, Kim, & Baral, 2021). Collective forest tenure reforms in countries such as China (Yiwen, Kant, & Long, 2020) and Indonesia (Putraditama *et al.*, 2021), although criticized in terms of effectiveness (Yiwen *et al.*, 2020), are likely contributing to this upward trend in community forest area. The trend in moving away from industrial forestry towards landholder-based forest management and community forestry may be due to increased support for community forests as a form of sustainable development.

From an ecological perspective, indigenous, low-intensity forest use has little negative impact on forest ecosystems (Gómez-Pompa, Whitmore, & Hadley, 1991). The effects of informal, more intensive timber harvest by the community in more forested landscapes in the Amazon and Central Africa, are limited to the easily accessible parts of the forests, where, after a while, the valuable species tend to disappear (Ferreira, Cunha, & Parolin, 2014). The environmental damage becomes stronger with the involvement of professional loggers, as they have the means for investments into infrastructure and heavy machinery. Although logging may be highly selective, the damage to the forest could be immense as it damages the remaining stand and changes its structure and tree composition in the long run (de Avila *et al.*, 2017). However, the basic ecological functions of the forest remain as long as it is not converted for agricultural purposes.

Independently of the type of ownership or management goals, community forest management has been supported across the globe by governments and donors as a way of combining socio-economic development with forest conservation. Transferring responsibilities from public (e.g., governments) or private (e.g., companies) entities to forest communities is believed to create conditions for better conservation and more sustainable use of forest ecosystems, as well as fostering social well-being and gender equity (Nandigama, 2020). There has been a high level of support for community forests managed under communal property rights, which suggests participatory engagement in common property resource management promotes environmental sustainability through improved livelihoods for the rural poor (Bluffstone *et al.*, 2018; Okumu & Muchapondwa, 2020; Ostrom 2008, 2009) and decreases the costs of management (Gutiérrez-Zamora & Hernández Estrada, 2020; Nandigama, 2020; Shumsky, Hickey, Johns, Pelletier, & Galaty, 2014). Community forests also increase local resilience and enable better disaster preparedness for emergencies ranging from earthquakes to the COVID-19 pandemic (Gentle *et al.*, 2020).

Communities can have full, partial, or no formal ownership of the forests they manage. In the cases that communities hold ownership of the forest land, they often share forest management responsibilities, including its costs and benefits, with governments, non-governmental organizations, etc., via different legal arrangements (Hyde, 2016). The mechanisms that transfer rights to use and management of forests from public or private land to communities greatly vary across the world. Legal arrangements range from the forestry regime of “baldios” in Portugal or the “montes comunales en mano commum” in Galiza/Spain (Carvalho Ribeiro, Sónia Maria, 1998; Skulska, Duarte, Rego, & Montiel-Molina, 2020), and the “van panchayats” in the Himalayas (Thakur *et al.*, 2020). In Mediterranean European countries, the existence of common property institutions and community forests in particular dates to at least a thousand years (Cullotta *et al.*, 2015; Skulska *et al.*, 2020).

Community forest management is also associated with use of forests in indigenous reserves or designated sustainable use areas including for example the sustainable use extractive reserves, some of which were created over the last decades, granting conditional local use rights on state lands for vast areas. In South America, often communities also manage land through forest concessions. Forest concessions are defined as a formal legal agreement signed with a concessionaire for the occupation and use of a territory. In these agreements, space units are demarcated for the use and management of ecosystems for specific uses and for a fixed time. There are at least 122 million hectares of tropical forests under concessions, equivalent to 14% of the world’s public forests some of which are managed by forest communities.

The industry is operated by small and medium forest enterprises which are largely left out of the forest statistics, planning and management, and yet it is growing rapidly in many tropical countries (Hoare, 2015). The small and medium forest enterprises are characterized by low level capital, informally trained workers and having potential for value addition (Osei-Tutu, Nketiah, Kyereh, Owusu-Ansah, & Faniyan, 2010). The industry contributes directly to the local economy in the form of improved livelihoods and cheap lumber for urban consumers. Small and medium forest enterprises are the main, additional or alternative income sources for a greater proportion of the local population as compared to the large-scale formal forest subsector in countries where the forestry sector is among the major income earner (Cerutti & Lescuyer, 2011; Osei-Tutu *et al.*, 2010). This is because small and medium forest enterprises tend to accrue wealth locally, empower local entrepreneurship and seek local approval to operate (Osei-Tutu *et al.*, 2010). Small-scale enterprises tend not to be adapted to landlocked, low population density, remote markets and high transportation, costs but can compete and replace forest concessions when public road infrastructures allow them easier access to the market (Karsenty, Drigo, Piketty, & Singer, 2008) **(Box 3.15)**.

Globally, about 15% of tropical forests are managed by communities (Arts & de Koning, 2017), many managed by indigenous peoples and local communities. As of 2020, indigenous peoples and local communities in Africa, South America and Asia, customarily managed at least 31% of land area corresponding to 571 M hectares (Khare, White, & Frechette, 2020). As of 2016, in Latin America nearly 33% of forests (232 million ha) were under some type of collective tenure regime owned by communities, most of which are of indigenous peoples, and another 8% of the area had been designated for their use. An important portion of these forests are used for meeting subsistence needs, but few of the communities undertake commercial logging operations, formally or informally. Traditional forest management for subsistence uses tends to be informed by traditional knowledge and customary local regulations (Gibson, McKean, & Ostrom, 2000). In turn, community forestry for commercial purposes is informed by management plans that are based on scientific forestry with no obvious role for indigenous knowledge. Often, these plans are inspired by large-scale industrial timber-harvesting schemes.

Since informal management schemes are considered by some to be ineffective or degrading, the management of forests by communities on the basis of the Reduced-Impact-Logging principles and formally authorized management plans has been widely promoted given assumptions that it would lead to sustainable outcomes in terms of biodiversity and local income. Accordingly, hundreds of initiatives across the tropics have promoted community forestry, in some cases also labelled as social forestry or collaborative forest management (Hajjar *et al.*, 2021).

The most developed cases of community forest management in Mesoamerica include Quintana Roo, Mexico, and Peten, Guatemala (see Supplementary material Box S3.1), as well as community forestry in the Amazon including Brazil, Bolivia, Peru. In Central Africa, community forestry has mainly developed in Cameroon, and to a lesser extent in Democratic Republic of the Congo. Especially in Latin America, the promotion of community forestry was accompanied by the formal recognition of tenure rights to indigenous peoples (RRI, 2015), which has been understood as a critical condition for achieving positive outcomes (Baynes, Herbohn, Smith, Fisher, & Bray, 2015).

In developed countries, community forestry is less well established than in developing countries (Bullock & Hanna, 2007). Charnley and Poe (2007) reported only 2% community and indigenous ownership of forests in developed countries in comparison with the approximately 14% of community and indigenous owned forests in developing countries. Community forestry began to be implemented in the 1990s in Canada as a result of public controversies surrounding large-scale industrial forestry, and as of 2007 existed in Ontario, Quebec, and British Columbia **(Box 3.17)**. In the Canadian context, forests remain state property but communities receive key management rights and responsibilities. In the United States of America community forestry initiatives have been supported through joint efforts across private, tribal, and public lands across the country. Despite widespread support for increased public participation in environmental decision-making in the United States of America, there has been resistance from the government and environmental groups to yielding actual control over land to local communities. Thus, in the United States of America collaborations between state and federal forest management agencies and local communities has been more common (Charnley & Poe, 2007).

The specific outcomes of community forestry initiatives largely depend on the biophysical conditions, tenure right situation, community characteristics, and the type of intervention. For the majority of cases, positive environmental and income-related outcomes are reported, but the need for formalization and the related bureaucratic and technical requirements negatively affect forest access and resource rights (Hajjar *et al.*, 2021) and the attractiveness for the local resource users (Pokorny, 2013). Accordingly, the long-term success of community forestry initiatives largely relies on continuous external support, but only in some limited cases (Pokorny, Johnson, Medina, & Hoch, 2012).

Box 3 17 **Community forestry on public lands in Canada.**

Community forestry has been a legally recognized form of forestry governance in Canada for over 50 years. While area in community forestry is small compared with industrial tenures, it makes important contributions to community development and diversifying the beneficiaries of forestry (Bullock & Hanna, 2007; McIlveen & Rhodes, 2016; Teitelbaum, 2016).

Three provinces in Canada have institutionalized forms of community forestry on public land: British Columbia, Ontario, and Quebec. Since 1998, British Columbia has granted 25-year renewable licenses to more than 50 organizations and indigenous communities under the British Columbia Community Forest Agreement (Government of British Columbia, 2020). British Columbia also has a tenure specific to indigenous communities, the First Nations Woodland Licence (Government of British Columbia, 2020).

Quebec was the second province to adopt a community forestry policy. Although implementation has been slow (Ministère des Ressources naturelles et de la Faune, 2011), a number of community forests have been created across the province (Bissonnette, Blouin, Bouthillier, & Teitelbaum, 2020; Teitelbaum, Beckley, & Nadeau, 2006). Many are located in proximity to small rural communities and are run by municipal or regional governments (Chiasson & Leclerc, 2013). A handful have also been allocated to indigenous communities and are largely run by the band council.

The province of Ontario has a network of county and municipal forests, as well as forests owned and managed by Conservation Authorities (Teitelbaum & Bullock, 2012). Under this model, forestlands are owned outright by local government entities and have strong authority over management decisions. In contrast, the provinces of Quebec and British Columbia retain considerable control over management decisions such as allowable timber cut, wild species management and gathering. Community forestry entities in Quebec and Ontario also face substantial administrative burdens from a regulatory system designed for much larger operations (Ambus & Hoberg, 2011; R.L Trosper & Tindall, 2013).

Timber harvesting is a main objective for many community forests and, in at least one case, generates significant employment in its region of British Columbia (McIlveen & Rhodes, 2016). However, there is considerable diversity in management values and priorities, with some strongly focused on protection of ecological functions and nature's contributions to people. Some community forests have diversified their activities through development of recreation and/or alternative forest products. For example, one community forestry initiative in Quebec developed an innovative approach combining timber and wild blueberry production. Revenues have been sufficient to support research on optimal conditions for co-habitation of trees and blueberries (Fournier, 2013).

Traditional forest management for subsistence uses tends to be informed by traditional knowledge and customary local regulations (Gibson *et al.*, 2000). In turn, community forestry for commercial purposes is informed by management plans that are based on scientific forestry with no obvious role for indigenous knowledge. Often, these plans are inspired by large-scale industrial timber-harvesting schemes.

In the Amazon, there are four general schemes of community timber harvesting: (1) traditional harvesting of forest products aimed to meet subsistence needs; (2) locally devised schemes to carry out commercial timber harvesting; (3) harvesting agreements between communities and loggers; and (4) formal community forestry on the basis of legally authorized management plans as described above (Sabogal, de Jong, Pokorny, & Louman, 2008). The species, volumes, areas, and management schemes of the forest operations vary strongly between and within these schemes and the context under which they occur. A key contextual factor is tenure. For example, some indigenous people and communities have been granted collective tenure, and others hold collective rights in extractive reserves, yet others have not been recognized with customary collective or individual tenure, which affects the community's possibility to legally use timber.

Informal logging operations tend to be highly selective of high-value species such as for example *Swietenia macrophylla, Manilkara huberi, Mezilaurus itauba, Handroanthus serratifolia*. While traditional logging practiced by communities works with motor-manual practices and concentrates on small areas up to 20 hectares with extraction volumes of around 50m³ in the entire area, formalized community forestry operations may cover extraction areas of up to 1,000 hectares and volumes extracted range from 5-20m³ per hectare. In accordance with Reduced-Impact-Logging principles, many in the Amazon region follow formally approved management

Box 3 18 **Coomflona in Flona Tapajos, Para, Brazil.**

Tapajós National Forest is a government-owned land with community use designated as protected area with sustainable use of natural resources. Located in the state of Pará, in the Brazilian Amazon, Tapajós National Forest occupies an area of 527,319 ha of mostly dense tropical forest characterized by the dominance of large trees under a climatic regime of high temperatures and intense precipitation distributed throughout the year (Humphries, Andrade, & McGrath, 2015; IBAMA, 2004; Silva, de Carvalho, & Lopes, 1985). Over 24 forest-based communities are based in the area. Approximately 500 indigenous people and 5000 local people live within Tapajós National Forest. They have diversified livelihood strategies which include agriculture, non-timber forest products, timber, and fishing (Andrade, de Carvalho, ilva-Ribeiro, & Dantas, 2014; ICMBio, 2015).

Tapajós National Forest residents founded a local timber cooperative, the Mixed Cooperative of the Tapajós National Forest (Coomflona) to manage tropical timber resources. The members include 150 forest residents from the 24 communities. With few exceptions, Coomflona hires external labor for work such as lawyer, forest engineer, and forestry machinery operators. The access to forest is collective, since every cooperative-member has the right to vote and make decisions over forest resource management. Decisions are made during general assemblies, held during the first three months of the year, and a cooperative executive committee operationalizes management decisions (Espada & Vasconcellos Sobrinho, 2019; Humphries, 2016).

Coomflona has a permit to manage timber with non-onerous (zero-cost) concession from the federal government, and every year it has to submit an operational plan to get the approval from the government to execute timber-harvesting operations.

Currently, the total timber harvest area covers 44,000 ha, and represents 8% of the total area of the Tapajós National Forest. Annually, Coomflona now manages an area of 1,500 ha, which has steadily increased since its first year of timber-harvesting operations in 2006. They manage for a cutting cycle of 30 years (Espada & Vasconcellos Sobrinho, 2019; Humphries, 2016). Coomflona implements reduced impact harvesting techniques, removing 3 to 4 whole trees per hectare. The main log extraction equipment in a skidder, and around of 30,000 meters cube of roundwood are harvest every year (Humphries *et al.*, 2015).

Coomflona, with the support of its partner organizations, achieved several notable accomplishments. First, the cooperative has secured financial resources for forestry operation costs, critical in community forestry. Second, the cooperative created an innovative system of funds in which to allocate net profit from timber sales to benefit timber workers, their families, and communities, and beyond, local people that do not participate directly in the cooperative. Third, Coomflona invested in a portable sawmill and small-scale carpentry to verticalize timber production, aggregate value to timber products, expand market strategies, and engage additional community members in timber production. Fourth, Coomflona has established long-term and strong partnerships with diverse organizations. Fifth, Coomflona has become a model for other community-based groups aiming to manage timber resources in sustainable-use protected areas, as in the cases of extractive reserves. Sixth, Coomflona is running timber management with good practices considered in the forestry sector; the Forest Stewardship Council certification, for instance, certifies that Coomflona is maintaining forest health and ecosystem functions while provisioning both local social and economic benefits.

plans drawing on timber inventories and respect defined cutting cycles (Sabogal *et al.*, 2008). However, most frequently, timber on communal lands is harvested by local loggers on the basis of informal arrangements that pay the communities or the communitarian leader a lump sum for the right to harvest the forests. While these arrangements typically are unfair and often the logger doesn't hold his promise, it provides communities the opportunity for an easy income (Medina, Pokorny, & Campbell, 2009) (**Boxes 3.18** and **3.19**).

In protected areas an important timber management experience operated collectively by community-based enterprises takes place in the Mayan Biosphere Reserve, a protected area of 2.1 million ha established in 1990s (Radachowsky, Ramos, McNab, Baur, & Kazakov, 2012). A total of 12-community concession contracts (for areas ranging from 7,000 ha to 85,000 ha for a total of 390,000 ha) were signed between 1994 and 2001 (Stoian, Rodas, Butler, Monterroso, & Hodgdon, 2018). All concession contracts required collective organization: three forms emerged i) limited liability companies or civil societies *(Sociedades Civiles)*, ii) civil associations, and iii) cooperatives. Community concession contracts are legal agreements between the state and an organized group composed of members living in a given community. These

Box 3 19 Ejido Petcacab-Quintana Roo, Mexico, drawn from (Wilshusen, 2005a, 2005b).

Local communities own approximately 45% of Mexico's forests and have relative autonomy to manage them. Some of these communities have established community forest enterprises in order to generate benefits, such as jobs (Frey *et al.*, 2019). In the Mexican state of Quintana Roo, tropical forest ecosystems dominate the landscape. Forest types vary by soil, topography and local climate: medium-stature forests (15 to 25 meters) are present on well-drained soils, while shorter forests occur on seasonally inundated wetlands depressions. Mahogany *(Swietenia macrophylla)* and Spanish cedar *(Cedrela odorata)* were historically the most important commercial tree species, but in recent decades lesser-known tropical species have come to constitute around 70% of the harvest (Ellis *et al.*, 2015). As of 1992, the harvest was managed by four associations of forestry ejidos through community forest enterprises with a combined allowable cut of 10,580 m^3 per year of mahogany and cedar from 393,481 ha of permanent forest areas (Flachsenberg & Galleti, 1999). Up until 1983, logging was carried out first by small private concessionaires and later by a parastatal company, with wildly fluctuating annual volumes between 10,000 and 50,000 m^3. Beginning in 1984, community management produced a striking reduction and stabilization of harvests of mahogany and cedar going from 10,000 m^3 annually, a 78% reduction from the last five years of the parastatal to around 5,000 m^3 in 2018, and foresters consider this to be sustainable (Bray, 2020; Navarro-Martínez, Ellis, Hernández-Gómez, Romero-Montero, & Sánchez-Sánchez, 2018).

Petcacab is an ejido, a common property land grant in Mexico's agrarian system, inhabited by Mayan indigenous peoples. It is located in Central Quintana Roo, with an estimated population of 947 and 206 legal members of the ejido. The property regime is communal with a total land area of 46,000 hectares and permanent forest area of 32,500 hectares. Petcacab's community forest enterprise was initially organized in the mid-1980s with external support from the Forest Pilot Plan, supported by the Mexican government and German foreign assistance. Petcacab initially organized its community forest enterprise as an entirely community-administered operation, supervised by community authorities. However, due to concerns about corruption, in 1996 Petcacab reorganized its community forest enterprise to be administered by what are termed "work groups" or coalitions of community members based on family clans and individual families. Access to communal lands by community groups, approved by the community assembly, was permitted by a 1992 reform to agrarian law. By 2000, Petcacab had 11 work groups who each received a proportional share of the annual authorized volume, and essentially managed themselves as small, separate community forest enterprises or microenterprises.

All of the work groups operated under a single management program prepared by a professional forester and approved by the Mexican environmental agency. In the 2000s Petcacab had authorized harvest volumes of 1,499 m^3 of mahogany, 2,545 m^3 of tropical softwoods, 3, 927 m^3 of tropical hardwoods and 10, 328 m^3 of polewood, with a decline in the volume of mahogany in more recent years. Production is small-scale industrial, with the use of tractors and skidders for extraction and logwood is sold to a community sawmill or intermediaries. Harvests are regulated by Mexican forest and environmental laws and compliance is considered good.

The work groups sell both logwood and sawnwood, after processing at a community sawmill. Most timber is sold domestically in Mexico. Benefits go to the individual work groups, with little or no reinvestment in the community or the community sawmill. Harvests are conducted according to the management programs with little or no input from indigenous knowledge. Harvests of mahogany have declined in recent years but are considered sustainable at current more reduced levels. Socially, the work groups have allowed for increased incomes at the work group and household level, but with a corresponding decline in community investments in public goods. The apportionment of the authorized volumes to individuals has led to a market in the shares of authorized volumes and increasing economic inequality with the ejido as some members purchase others shares. Economically the work groups appear to be profitable, and the work group arrangements appear to be sustainable.

25-year concession contracts allowed concessionaire members rights to manage and extract timber and non-timber forest products recognizing also rights to implement nature-based tourism activities in protected areas. This system of community concessions in the Multiple Use Zone (MUZ) represents about 15% of the country's total forest cover, including national parks (IARNA/URL/ILA, 2006). The area under forest concessions covers more than 480,000 hectares. To date, nine community concession contracts remain active (around 350,000 hectares) (Stoian *et al.*, 2018).

Compared to Latin America and South Asia, relatively little information on Africa was available. In Central Africa, the number of communities formally embracing community forest management has greatly increased over the last twenty years as all countries have included this management option in their forest legal frameworks. There are now more than a thousand of them, about 90% of which are in Cameroon. However, most of the community forestry operations validated by the authorities are either inactive, as in Cameroon (G. Lescuyer, Cerutti, & Tsanga, 2016), or oriented towards conservation, as in the Democratic Republic of Congo (Bauer, 2016). In total, only about 150 forestry communities are authorized in Cameroon, and about a hundred are created or in the process of being created in both Gabon and the Democratic Republic of Congo. There are no regional statistics on timber production from community forestry in Central Africa. Yet, based on case studies in each country, a maximum of 50,000m^3 would be legally extracted from community forestry harvesting operations in the Congo Basin (Beauchamp & Ingram, 2011; Julve *et al.*, 2013). Due to inadequate regulatory texts that are costly to apply for (Cuny, 2011), the vast majority of community forests that exploit timber do so illegally, destined for domestic markets that do not require timber of legal origin. There are no statistics on these illegal practices, but numerous reports from Cameroon (Nzoyem, Vabi, Kouokam, & Azanga, 2010) and Gabon (Ondo, Medik, Mijola, & Boussougou, 2020) indicate that informal timber-harvesting from community forestry operations far exceeds the volume legally extracted.

Generally, South Asia's forest dependent communities are the ones who are also the more disadvantaged in the region. Two categories of forest dependent people are identifiable. First, those who are traditionally the forest communities residing in and around forest areas for generations, such as tribal communities in India. Altogether, the population of this group is estimated to be 150 million in the region (World Bank, 2005). Second, people who depend on forests for a variety of products and nature's contributions to people but do not directly reside inside or in the vicinity of the forest. There are around 400 million forest users in this category (Poffenberger, 2000). More recently, rapid rural and urban and even overseas migration of youths has led to change in the conventional patterns of forest dependence, with reduced use of forest products in livelihoods (Ojha *et al.*, 2017).

In terms of policy shifts, South Asia has notable community forestry initiatives in terms of scale and demonstrated outcomes, although the actual form and operational modalities vary greatly across the countries and sub-national regions. Likewise, a variety of local regimes of community forestry are found: formally handed over state forests, jointly managed forests, 'sacred groves' with cultural values, community plantations, and other forms of collective land use for trees and pastures. The beliefs and rituals linked to sacred groves have helped to conserve biodiversity, although they are under threat due to changing values and perceptions (see section 3.3.5).

Forest harvesting status and trends data for community forestry are not readily available for most countries in South Asia. Available evidence suggests all the countries and their sub-national authorities are struggling to optimize forest harvesting in a sustainable, efficient, and equitable way. A significant part of the forest landscape is under protected area management, and there have been participatory and co-management shifts in this regime too, especially since early 1990s. Studies show that such participatory shift in protected area management has resulted in more active use of resources, as found by a study in Bangladesh (K. Islam, Nath, Jashimuddin, & Rahman, 2019).

In Bhutan, conservation rather than sustainable use mindsets dominate forest management policy and programs, and strategies and methodologies to promote sustainable harvesting are slow to develop (Phuntsho, 2011). Despite having nearly 70% of area under forest, Bhutan has kept harvesting level to a minimum, favoring import of forest products. The most significant wood-based import item is charcoal. In 2012, Bhutan imported charcoal worth 16.8 million United States dollars, comprising 1.4 percent of total imports and 60 percent of wood-based imports (MoF 2013, cited in World Bank (2019). The first national forest inventory published in 2017 and provides detailed data and information on Bhutan's forests, showing the Bhutan is under harvesting its forest below the potential (World Bank, 2019). A bottom-up approach to forest management in Bhutan began after the 1979 royal decree that called for the involvement of local people in tree planting activities (Phuntsho, 2011). Bhutan's community forestry policy emphasizes protection, conservation and sustainable use of forest resources in the country, together with contributions to poverty reduction and local democratization (Phuntsho, 2011). The level of harvesting in community forestry is no different than the national scenario. Following the adoption of a more decentralized and people-centered approach to forestry in the early 2000s, the number of community forestry management groups has increased rapidly since 2007. By 2018, there were 781 community forest management groups involving 32,402 rural households managing 92,165 hectares (3.0 percent) of forest land (MoAF, 2018; World Bank, 2019). Although the government is supportive to the implementation of the community forestry program, community forest management groups continue to face administrative hurdles with regard to timber marketing, leading to under-harvesting of forest stock (Samdrup, 2011).

Community forestry in Nepal emerged in the mid-1970s and led to the devolution of management and user rights to forest user groups, largely as a shift in the approach to conserve the hill forests in the context of degradation and deforestation. During the last three decades Nepal's community forestry programme has evolved in terms of coverage and institutional innovation, supported through appropriate changes in policies and legislation. Substantial international support has also helped to sustain community participation. Community forestry has contributed to livelihoods of nearly one-third of the country's total population, improved forest conditions and biodiversity, and above all, developed itself as a self-sustaining system involving a strong base of policy champions, service providers, and critical action researchers. By the end of 2019, over 22,000 community forestry user groups had been registered across the country, with rights granted to manage nearly two million hectares of forest areas (about a third of total forest areas of the country. Community forests are vital components of environmental resilience and nature's contributions to people not only to the people close by but also to large populations downstream. Dissipating fears of desertification, community forestry has led to improvement in forest ecology, with 74% of the forest area managed by community forestry user groups reported as in "good" condition, compared to 19% in "degraded" condition (Kanel & Kandel, 2004). Community forestry user groups also compare favorably to government forests in terms of change in forest condition (Nagendra, Pareeth, Sharma, Schweik, & Adhikari, 2008). More recent empirical evidence confirms improved biodiversity outcomes from community forestry (Luintel, Bluffstone, & Scheller, 2018). Nepal has a strong legislation that allows communities to enjoy perpetual rights over designated community forest areas. Such perpetual and sustained rights of access to forests have been key for the success of community forestry in Nepal (Acharya, Adhikari, & Khanal, 2008). Though the land ownership remains with the government, the tenure of forest biomass is transferred to the community through a detailed approval process. Community forestry user groups retain 100 percent of revenues generated from their forest, but they have to allocate 25% of the income to forest development activities and 35% to programs that directly benefit the poorest households within the community forestry user group. The existing forest law provides communities with enough rights to choose their objectives of forest management and harvesting, but too often the actual practices of regulation and bureaucratic oversights hinder active management of forests beyond subsistence use. Nepal has achieved massive scale community forestry development in terms of enabling policy and institutional development, but the actual use of forest is less than 30% of the annual sustainable harvest level. Despite having 45% of the country's area under forests, the contribution of the forest sector to local and national economy has remained much less than the potential in Nepal (Banjade, Paudel, Karki, Sunam, & Paudyal, 2011; Chhetri, Lund, & Nielsen, 2012; Luintel, Bluffstone, Scheller, & Adhikari, 2017; Thoms, 2008). As the national mood has recently shifted towards active forest management, several attempts have been made to develop and scale up silviculture innovations. These have stimulated debates in scientific forest management, though outcomes on the ground have remained limited.

Community forestry in Sri Lanka has developed somewhat similarly to Nepal, but began in the 1990s. The community forestry project was initiated in Sri Lanka after a series of forest policy reforms and decentralization arrangements during the 1980s. Since 2003, the Department of Forest Conservation, a government department responsible for forestry in Sri Lanka, has been testing and trialing various approaches using the community forestry model (Ekanayake, Xie, Ahmad, Geldard, & Nissanka, 2020). Community-based forest management in Sri Lanka encompasses community-owned forests and agro-forests as well as government-owned forests managed by communities. Forests managed by communities produce timber and wood products in agroforestry systems, on agricultural lands and community lands including farmer woodlots and silvopastoral systems (De Zoysa, 2017). The home gardens, outside natural and planted forests supply more than 70% of the timber and 80% of the fuel wood in Sri Lanka (De Zoysa, 2017). Recent studies have shown that impact of community forestry development has led to positive outcomes on livelihoods (Ekanayake *et al.*, 2020).

In India, large scale shifts from state control of forest to joint management with local communities has led to a large area of forest being managed under joint forest management. Joint forest management covers more than 22 million hectares which is about third of the forest land in India, engaging 25 million people through 104, in 729 committees across more than 100,000 villages (Sundar, 2017). Like Nepal and Bhutan, a conservative approach to forest harvesting dominates forest management practices across all regimes of public forests.

India's average annual yield of forest is estimated as 85.65 million m^3, whereas the annual removal of only 5.85 million m^3, which is 6.82% (FSI, 2019). Total growing stock is estimated to be 5915 million m^3 and the growing stock of trees outside forest is 1642 m^3 (FSI, 2019, p. 117). Total forest coverage between 2010 and 2020 increased in India by 0.38% (FAO, 2020a). Timber production from public forest meets only the 3.35% of the total demand, while trees from outside areas officially classified as forests provide 45% of the demand (Ghosh & Sinha, 2016). It is suggested that community forests managed by indigenous people and local communities are more likely to harvest in sustainable ways than those managed by the government (Sundar, 2017). Despite government efforts to raise domestic productivity, India's overall timber production remains low. This is especially true for the tree species preferred by consumers such as teak, sheesham and pine (Norman & Canby, 2020). The International Union of Forest Research Organizations estimates that India was the third-largest importer of illegally logged timber in the world in 2016, after China and Vietnam (Kleinschmit *et al.*, 2016). While its own forests are under harvested, India is emerging as a major importer of timber (Vanam, 2019). The demand for timber is growing from the current gross value added of 606 billion United States dollars in 2011 (FAO, 2014a). However, restrictive policy and regulatory barriers inhibit community forestry groups from harvesting and selling surplus timber from their forests even when supported by sustainable forest harvesting protocols (Shyamsundar, Ahlroth, Kristjanson, & Onder, 2020). India's joint forest management is an arrangement for co-management between local communities and the Department of Forest. Typically, the joint forest management committee holds a joint account in the local public sector bank with the chairperson, vice-chairperson and the district forest officer or her nominee as joint signatories through which financial aid from donor and the government is channeled (Sundar, 2017). The district forest officer prepares forest management plans in consultation with the communities.

3.3.4.3.3 Industrial Logging practice

In practice there are three major types of industrial logging: (i) most frequently, so-called private concessions grounded on an agreement between a private landholder and the logger or by the landowner himself, mostly on a short-term basis and sizes of some few hundred to thousand hectares; (ii) government granted concessions in public forests (~1.5 M hectares by 2019) (J. R. Ribeiro, Azevedo-Ramos, & Nascimento dos Santos, 2020) based on a set of technical, financial, and administrative requirements, that comprise several ten thousand hectares for an entire timber-harvesting cycle; (iii) the legal use of timber from authorized forest conversion areas in private landholdings. Nearly all industrial logging is organized by sawmills to secure their supply.

Large-scale industrial logging involves felling large numbers of trees in areas of more than 50,000 ha. The loggers have felling permits, use heavy machinery, have a processing plant and sell a number of wood products including logs, sawnwood, veneer, plywood, and wooden floors, almost exclusively for export (Cerutti & Lescuyer, 2011). In the last three to four decades, industrial logging has been the major source of globally traded wood products (FAO, 2009). For example, in Papua New Guinea, large-scale industrial logging companies export approximately 90% of the logs harvested in the country (PNGF, 2009).

Large scale forestry operations occurring within managed forests and tree plantations were estimated to cover 26% of global forest area between 2001 and 2015 (mainly in America and Europe) (Curtis, Slay, Harris, Tyukavina, & Hansen, 2018). Within some tropical countries, a small number of large-scale companies who source much of their timber from small and medium sized enterprises dominate the export sector (Osei-Tutu *et al.*, 2010). Allocation of forests or trees for large scale industrial logging in public and large-scale privately owned forests is predominantly done through the provision of forest concessions (Vilanova, Ramírez-Angulo, Ramírez, & Torres-Lezama, 2012), which is a common legal tool among forest policy decision-makers (Karsenty *et al.*, 2008). Forest concessions have been carried out for hundreds of years in boreal, temperate and tropical public forests (Van Hensbergen, 2016). Forest concessions have been carried out in many of the Central African forests for over a century (since the colonial rule), and in South American countries for over three decades (Karsenty *et al.*, 2008). Within Latin America, Southeast Asia and West & Central Africa, forest concessions cover about 123 million ha accounting for approximately 14% of the publicly owned forests (Van Hensbergen, 2016).

Conventional logging is highly selective, sometimes concentrating on only one or two species. Selecting and felling trees often occurs without a complete inventory or thorough spatial planning. In large-scale conventional logging there may be issues with incorrect identification, poor labor conditions, insufficient training in best practices, and overly high felling rates. These conditions can lead to immense damage and economic losses (Piponiot *et al.*, 2019).

Since the 1950s, clear-cutting involves the use of heavy timber machinery (Boucher, Auger, Noël, Grondin, & Arseneault, 2017; Maleki, Nguema Allogo, & Lafleur, 2020; Mohr, Coppus, Iroumé, Huber, & Bronstert, 2013), which may lead to changes in tree composition and oversimplification of stand structure and species diversity (Boucher, Arseneault, Sirois, & Blais, 2009; Boucher *et al.*, 2017; Gustafsson, Kouki, & Sverdrup-Thygeson, 2010). These activities can have negative effects on wild plant and animal populations (Berg *et al.*, 1994; Gärdenfors, 2010; Hyvärinen, Juslén, Kemppainen, Uddström, & Liukko, 2019; Kålås, Viken, Henriksen, & Skjelseth, 2010). Log skidding, done with heavy machinery, can also be damaging if done during improper weather conditions or during the wrong season when soils are especially vulnerable.

Industrial logging is done with and without legally authorized management plans. A correct tracing of the logs to their point of origin varies widely so loggers may use management plans to justify harvesting adjunct areas technically not under the plan. The vast majority of sawmills in the tropics work with this kind of timber-harvesting. They have small teams of mostly non-local seasonal workers and use tractors for both the construction of access roads, secondary roads, and landings as well as for the skidding (Pokorny & Steinbrenner, 2005). Larger companies may also use skidders and stackers for loading. Sometimes machinery is owned by the sawmill, sometimes services are subcontracted. Transport distances from the forest to the sawmill may reach up to nearly 100 km (Pokorny & Steinbrenner, 2005). However, if the distance becomes too large, the saw lines are dismantled and rebuilt closer to the forest.

Alternative "sustainable forest management" schemes are meant to ameliorate several of the concerns raised regarding species identification, spatial planning, proper use of equipment, and proper application of management plans. More sustainable forest management is well planned so as to minimize damage on the remaining stand while effectively making use of costly heavy machinery. This includes infrastructure planning, spatial planning of harvesting operations, harvest planning based on an inventory of all commercially valuable trees, and skid-trail planning (Putz *et al.*, 2012).

In tropical and subtropical regions such harvesting is done by larger companies with the necessary human and financial resources and engaged in export activities often linked to Forest Stewardship Council (FSC) certification (Pokorny & Steinbrenner, 2005). Certification requires the demarcation of protected areas, and the timber-harvesting of a wider range of tree species, including the ones with lower commercial value, so as to reduce the pressure on the most valuable tree species (Putz *et al.*, 2008). The engagement of the certifier has positive effects on the quality of the operation, the treatment of the workers and the local resource users living around the management unit. However, certified companies tend to work as enclaves in the forest landscape and prefer to work with non-local workers. They prioritize fast timber-harvesting and hesitate to invest in the long-term security of the management unit once the area has been logged. These practices place the long-term sustainability of these certified operations in question.

Since the 1980s, variable retention forestry is being promoted as a sustainable forest management practice in temperate and boreal forests as opposed to clear-cutting (Fedrowitz *et al.*, 2014; Franklin, Berg, Thornburgh, & Tappeiner, 1997; Harkema & Scott, 2002; Kuuluvainen & Grenfell, 2012). It is a system in which key structural components of the original stand are retained at the time of logging through selection cutting, gap cutting and modifications of clear cutting, and become part of a new stand that regrows after logging (Franklin *et al.*, 1997; Gustafsson *et al.*, 2010; Koivula *et al.*, 2014; Koivula, Silvennoinen, Koivula, Tikkanen, & Tyrväinen, 2020; Puettmann, Messier, & Coates, 2009). Emerging research reveals that tree retention has the potential to reduce impacts of logging on forest biodiversity through creating favourable conditions that allow for complex and uneven forest structures similar to those of natural forests (Gustafsson *et al.*, 2020; Moussaoui, Leduc, Fenton, Lafleur, & Bergeron, 2019; Opoku-Nyame, Leduc, & Fenton, 2021). The practice is being adopted with rather modest retention levels ranging from 30 to 40% (Beese, Deal, Dunsworth, Mitchell, & Philpott, 2019; Scott, Neyland, & Baker, 2019). Though retaining small amounts of trees or patches is better than traditional clearfelling (Gustafsson *et al.*, 2020; Koivula & Vanha-Majamaa, 2020), only retaining a minor proportion of the volume of harvestable timber (often 1-10%) makes it practically impossible to avoid edge effects and random demographic effects on the forest stands. Maintaining more of the mature forest characteristics in production forests would require lower harvest intensities in some areas than is currently typical. Therefore, this low level of retention is still considered by some scholars as clear-felling (Fedrowitz *et al.*, 2014).

Prevailing retention practices have been reported to lack ecological credibility in safeguarding biodiversity and there are calls for their further development (Kuuluvainen, Lindberg, Vanha-Majamaa, Keto-Tokoi, & Punttila, 2019). Other studies have reported that it is not necessarily the level of retention of living trees, but rather, the microclimatic continuity, and maintenance and active increase of legacies such as existing coarse woody debris, very old trees, and tree species mixtures that significantly contribute to the conservation of forest species (Koivula & Vanha-Majamaa, 2020; Siitonen, 2001).

Diversification of silvicultural harvesting techniques is recommended to enhance specific structural or compositional elements and the diversity of species in forest stands. Either clear-cutting, partial cutting or selective cutting can be carried out to match variations in stand conditions and effects of natural disturbances, biophysical site characteristics and succession processes (Bergeron, Gauthier, Kafka, Lefort, & Lesieur, 2001; Harvey & Bergeron, 1989; Maleki *et al.*, 2020). Clear-cutting tends to allow cycling of early successional species into a single species dominant stand, while partial cutting and extended rotations can enable maintenance of a mixed species stand or stands that have some characteristics of older forests (Ruel, Fortin, & Pothier, 2013).

Post-logging forest restoration greatly relies on post logging seedling generation. The ability of a species to be sustained through rotations depends on the growth and reproduction of surviving adults, juveniles and seedling regeneration (smith *et al.*, 1997). However, many of the high-value timber species are nonpioneer light demanders whose seedlings occur at low densities in the forest understory due to limited shade tolerance (Grogan, Landis, Ashton, & Galvão, 2005; Gullison & Hubbell, 1992; Hall, Medjibe, Berlyn, & Ashton, 2003; Jones, 1956; Lamprecht, 1989; Medjibe & Hall, 2002; M Schulze, Vidal, Grogan, Zweede, & Zarin, 2005), such as wind-dispersed mahoganies and related genera in the family Meliaceae (Swietenia, Cedrela, Chukrasia, Entandrophragma, Khaya, Toona), Amburana, Cedrelinga, Couratari, Dinizia, Hymenolobium, and Tabebuia. These usually have limited post-logging regeneration (Dickinson & Whigham, 1999; Grogan, Galvão, Simões, & VerÍssimo, 2003; Gullison, Panfil, Strouse, & Hubbell, 1996; Schulze, 2003, p. 2003; Veríssimo, Barreto, Tarifa, & Uhl, 1995) and thus require adjustment in logging and silvicultural practices to promote their regeneration. To ensure sustained yield timber production from such timber species across the tropics, there are a number of silvicultural practices that should be taken into consideration (Grogan & Galvão, 2006).

Economically, timber-harvesting is most profitable for the traders, particularly if engaged in export markets. Conventional, particularly illegal, timber-harvesting, is also profitable for the owner of the sawmill, but also provides urgently required income opportunities for local people, not so much in the forest operations, but in the sawmills (Pokorny, 2013). The benefits of large-scale industrial logging to the local economy are usually limited (Gray, 1999) to some low-paid work, but loss of non-timber forest products which many local people often rely on for subsistence or livelihood diversification can have serious negative impacts (Adams, 2009). Benefits to the national economy are restricted, because while value is added to the timber when it is sawn and made into products, this typically takes place elsewhere (Adams, 2009). The main products obtained are round logs which are directly exported with very little in-country downstream processing. In instances where companies obtain concessions from private or community forests, these give royalties to the owners which depend on the tree species harvested. Nevertheless, large concessions seem to be a suitable tenure model in low-density areas where central or local governments are not capable of creating or maintaining adequate infrastructure to support regional economic issues and where only large-scale companies have the potential to do so (Karsenty *et al.*, 2008).

Social impacts are felt due to large amounts of migrant labor associated with industrial logging. A larger proportion of the workers in large-scale industrial timber-harvesting operations are permanent, but are brought from other regions and seldom become settled in a region. Concessionaires, especially including certified companies, have to effectively protect their management unit against informal harvest and encroachment. Accordingly, local resource users living in and around concessions suffer from restricted access to forest management areas. In Central Africa, the social impact of industrial timber-harvesting remains a contested issue. Taxation systems and services due to workers and the local populations are clear on paper, but there is limited transparency or availability of information about how much of the due amounts or promised services are actually paid into the State coffers or delivered locally. And while there seems to be a bit more clarity on the amounts of money that are collected and redistributed to local councils and villages, e.g., in Cameroon (Cerutti, Lescuyer, Assembe-Mvondo, & Tacconi, 2010) and the Democratic Republic of Congo (Tsanga, Cerutti, Bolika, Tibaldeschi, & Inkonkoy, 2020) much of the burden of redistributing benefits to local populations remains within the concessionaires themselves, which are not always willing or capable of playing that role (Cerruti *et al.*, 2017).

Ecological, economic and social sustainability can perhaps be achieved through continuous-cover forest management (e.g., Fedrowitz *et al.*, 2014; Franklin *et al.*, 1997; Kuuluvainen & Grenfell, 2012). This regime applies logging methods other than clear cutting and thus varies the amount and spatial distribution of retained trees, and the size of harvested openings. The logging methods include selection cutting, gap cutting and modifications of clear cutting, all characterized by maintaining a significant proportion of trees throughout the logging cycle (e.g., Koivula *et al.*, 2014; Puettmann *et al.*, 2009).

Experimental evidence suggests that even modest retention of living trees in harvested blocks is beneficial for biodiversity (Koivula & Vanha-Majamaa, 2020). Also, based on landscape preference research, retention methods may be preferred over clear cutting by citizens who use forests for aesthetic pleasure, recreation, hunting, or harvesting (see 3.3.4.4). Clear cutting decreases the aesthetic and recreational values of forests (e.g., Arnberger *et al.*, 2018; Karjalainen, 2006; Tyrväinen, Silvennoinen, & Hallikainen, 2017), whereas logging methods with a high amount of retained trees, such as selection cutting, are considered socially more acceptable (Putz *et al.*, 2008; Ribe, 1989). Citizens prefer forests with diverse tree ages, species, and sizes (Silvennoinen, Alho, Kolehmainen, & Pukkala, 2001; Silvennoinen, Pukkala, & Tahvanainen, 2002; Tyrväinen *et al.*, 2017) with not too densely spaced trees (Ribe, 1989; Silvennoinen, 2017).

Industrial logging is quite extensive in the tropics **(Box 3.20)**. It takes place legally on public and private lands, and illegally on public forests designated for conservation. Logging also occurs on indigenous people and local communities' lands and territories. Forest concessions have been widely used to allow companies to undertake large-scale timber-harvesting operations, yet these areas have been shrinking over time, particularly in the Amazon (e.g., Bolivia, Peru) and Southeast Asia (e.g., Malaysia and Indonesia). There are significant areas of public lands granted as concessions on all continents. In 2009, small properties accounted for 28% of production, medium-sized properties extracted 41% of wood and large properties supplied 31% of roundwood (Pereira, Santos, Vedoveto, Guimarães, & Veríssimo, 2010). Only 29% of production in 2009 came from areas owned or leased by the timber industries. The remainder (71%) originated in third party areas.

Throughout the tropics, forestry regulations commonly grant rights to industrial, large-scale, export-oriented timber-harvesting concessions. These concessions require management plans, which are presumed to maintain forest cover and biodiversity. All countries in Central Africa follow the concessionary model, with the Ministries of Forests granting rights and responsibilities to the concessionaire (i.e., a private entity is given permission to manage a public property) either through public auctions or directly. The duration of the contract varies. In the Central African Republic, the concession is granted for the entire lifespan of the company, in all other countries there exist legal temporal limitations to the contractual agreement, which is 15 years in Cameroon, Republic of Congo and Equatorial Guinea, 25 years in the Democratic Republic of Congo, and 30 years in Gabon (Cerutti, Nasi, & Center for International Forestry Research (CIFOR), Kenya and Indonesia, 2020). Timber-harvesting concessions comprise a total area of 50 million ha in the Congo Basin, of which about half had approved management plans by 2020 (Cerutti *et al.*, 2020). Management plans are generally based on a rotation period of about 30 years, with annual allowable cuts authorized for timber-harvesting each year by the forest administration.

Logging in the boreal and temperate forests is mainly industrial in scale (Safford & Vallejo, 2019). Approximately 90% of the forest in Fennoscandia, and perhaps 40% and 60% of Canadian and Russian forests, respectively are subject to industrial tree harvest (Gauthier, Bernier, Kuuluvainen, Shvidenko, & Schepaschenko, 2015). Prior to the 20th century, selective cutting was the dominant logging practice in the temperate and boreal forests. An intensive era of clear-cutting targeting mainly conifer trees began in the 20th century due to economic factors (Dupuis, Danneyrolles, Laflamme, Boucher, & Arseneault, 2020; Koivula & Vanha-Majamaa, 2020; Lundmark, Josefsson, & Östlund, 2013; Storaunet, Rolstad, Gjerde, & Gundersen, 2005). Clear-cutting continues to be the dominant logging

Box 3 20 Industrial logging in the Amazon.

In 1998, the Brazilian Amazon generated 10.8 million cubic meters of native wood. Twenty years later, only 57% of this volume was produced (~ 6.2 million m^3 (Lentini, Sobral, & Vieira, 2020). This was due to increasing competition with cheap supplies of forest products from tree plantations and contractions in the domestic market, as well as replacement with other materials such as plastics, steel and aluminum. An estimated 95% of the sawmills in the region are small family enterprises with very limited managerial capacity. Despite operations based on legally approved management plans, it is unclear whether this logging is sustainable. The extraction (cutting and skid trails) is performed mostly (61%) by third parties, while the rest (39%) is extracted by the processing industries themselves (D. Pereira *et al.*, 2010). The Amazon has more than 300 species of trees considered commercially valuable (Martini, Rosa, & Uhl, 1994). However, for decades the very same 15 to 20 species of commercial interest were harvested. Some of the most strained and consequently most pressured species are: *Hymenaea courbaril, Handroanthus sp, Apuleia leiocarpa, Goupia glabra Aubl., Manilkara alata, Himenolobium petreum, Couratari* sp., *Dinizia excelsa* (Lentini *et al.*, 2020). Manifold attempts to broadening this range have not been too successful. Only very few large companies have the interest and capacity to comply with the Forest Stewardship Council certification standard. It is estimated that less than a quarter of the timber produced in the Amazon is exported, as most of the timber is consumed in the big cities at the coast. Depending on the forest type and the market situation, between 10 to 25 m^3 per hectare is harvested. Increments of commercial timber are low around 0.5 to 1.5 m^3 per year and hectare, which mathematically result in harvesting cycles of around 25 to 35 years (Pereira *et al.*, 2010).

The regulations established in the Brazilian Amazon assume that a minimum harvest cycle of 25 to 30 years would guarantee the long-term sustainability of forest management. The legal requirements for industrial timber harvest include clarified tenure arrangements for the forest management unit, the formulation of a sustainable forest management plan, and annual operational plans. The volumes and products of harvested wood have to be reported in a document of forest origin (DFO) designed to accompany legally harvested wood at all stages of the transport and production chain (Waldhoff & Vidal, 2015). The regulations foresee two categories of forest management: 1) Low-intensity forest management, normally by local communities, with a maximum harvest volume of up to 10 m^3 per ha, a minimum harvest cycle of 10 years, and restrictions on the use of heavy machinery; 2) Complete forest management, which allows a maximum harvest volume of 30 m^3 per hectare and year, minimum harvest cycles of 25 to 35 years, and without machinery restrictions (Pereira *et al.*, 2010).

practice (Curtis *et al.*, 2018; Kålås *et al.*, 2010; Siitonen, 2001), although trials for partial cutting practices, such as retention silviculture have been established to test their operational and biological feasibility (Bose, Harvey, Brais, Beaudet, & Leduc, 2014). In clear-cutting, mature trees are usually completely removed, followed by regeneration through site preparation, sowing or planting, tending of the emerging cohort of even-aged trees, and often a relatively short logging rotation (Koivula *et al.*, 2020; Safford & Vallejo, 2019). An underlying rationale of clear-cutting is economic because it is seen as highly efficient and leading to sustained yields of timber (Koivula *et al.*, 2020). The concept of sustained yield has been criticized for only concentrating on the maintenance of timber stocks over time, while other forest resources that are protected with site-specific practices are not explicitly considered in the management plans, consequently leading to their decline (Berg *et al.*, 1994; Cyr, Gauthier, Bergeron, & Carcaillet, 2009; Luckert & Williamson, 2005).

Tree retention is an emerging alternative to clear cut harvesting, practiced on several continents including North and South America, Oceania, and Europe (Gustafsson *et al.*, 2020). Emerging research reveals that tree retention has the potential to reduce impacts of logging on forest biodiversity through creating favourable conditions that allow for complex and uneven forest structures similar to those of natural forests (Gustafsson *et al.*, 2020; Moussaoui *et al.*, 2019; Opoku-Nyame *et al.*, 2021). Though even leaving small amounts of trees or patches is better than traditional clear-felling (Gustafsson *et al.*, 2020), tree retention comprising a minor proportion of the volume of harvestable timber (often 1-10%), which makes it practically impossible to avoid edge effects and random demographic effects on the forest stands. Maintaining more of the mature forest characteristics in production forests would require lower harvest intensities in some areas than is currently typical. Determining exact levels that are required to secure long-term viable populations of different species, as well as the most cost-efficient implementation of these conservation measures, remains a major challenge for future research (Gustafsson *et al.*, 2010).

There is a continued increase in the amount of wood removals globally through industrial logging practices **(Figure 3.56)**. In 2019, global wood removals were estimated at 3.97 billion m^3, of which 2.02 billion m^3 was industrial roundwood and 1.95 billion m^3 fuel wood. The year 2018 had the highest level of production and trade values for global wood removals and all major wood-based products since 1947 (data from the Food and Agriculture Organization of the United Nations, http://www.fao.org/faostat/en/#data). The demand for and the consumption of wood products is escalating in line with growing populations and incomes, a trend expected to continue in the coming decades (FAO, 2010b). North American and European countries have the highest global wood yields (Chaudhary, Carrasco, & Kastner, 2017), which is partly attributed to clear-felling regimes prevalent in temperate Europe/North American countries with high yields

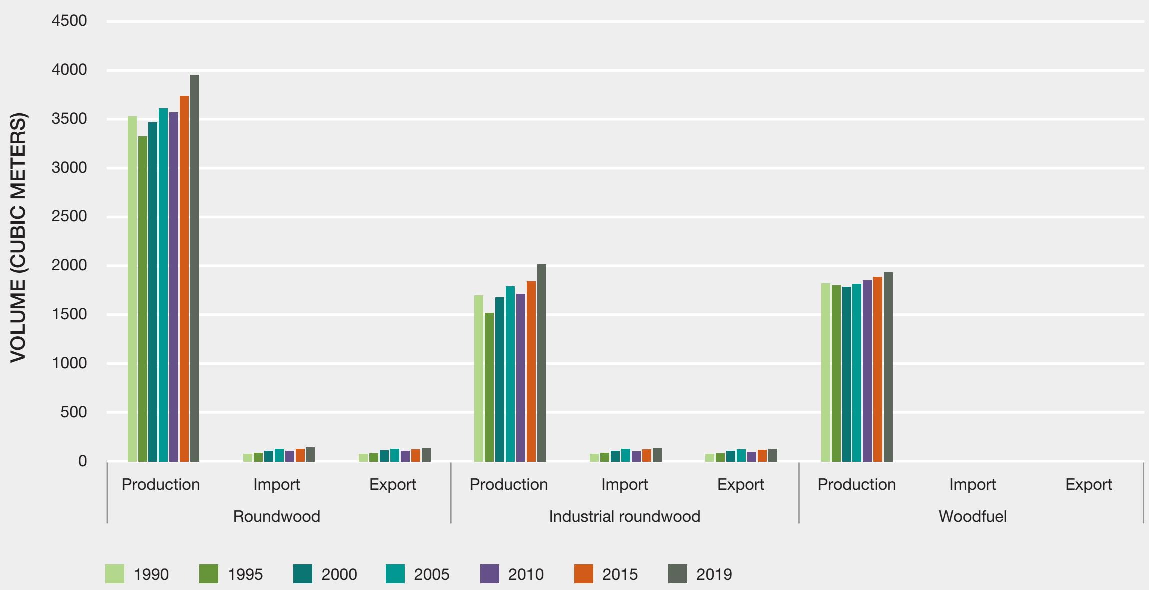


Figure 3 56 **Global wood removals 1990–2019.**

Data from FAO database (FAO, 2021b) under license CC BY-NC- SA 3.0 IGO. See data management report for the figure at https://doi.org/10.5281/zenodo.6453131.

compared with low-yield selective logging in the tropics (Chaudhary, Burivalova, Koh, & Hellweg, 2016). There are also large-scale imports of timber products by a limited number of countries especially China and the United States of America. The globalization of trade has enabled such countries to reduce local forest exploitation and achieve forest transitions from net deforestation to net reforestation (Kastner, Erb, & Nonhebel, 2011; Meyfroidt, Rudel, & Lambin, 2010; Mills Busa, 2013).

Some of the logging practices in species-rich tropical forests have been reported to resemble mining operations at the species level (Gómez Pompa, 1989; N. Johnson & Cabarle, 1993; Moad & Whitmore, 1994; M Schulze *et al.*, 2005), where a single, or group, or wider community of high value timber species are targeted for extraction. In the past, major target species included the big leaf Mahogany *(Swietenia macrophylla)*, Brazilwood or Pau-brasil *(Caesalpinia echinata)*, Brazil-nuts *(Bertholletia excelsa)*, rosewood (*Dalbergia nigra* and *Aniba rosaeodora*) and others (Martini *et al.*, 1994; Mark Schulze, Grogan, Landis, & Vidal, 2008; Veríssimo *et al.*, 1995). Due to these practices, some species were reported endangered and added to Appendix II of Convention on International Trade in Endangered Species of Wild Fauna and Flora. With scarcity and restrictions in extraction and trade of the Convention on International Trade in Endangered Species of Wild Fauna and Flora listed species, new species are targeted. This practice has led to severe and dense reductions of adult populations or old growth timber stocks, often at large spatial scales (Uhl, Veríssimo, Mattos, Brandino, & Vieira, 1991; Veríssimo, Barreto, Mattos, Tarifa, & Uhl, 1992; Veríssimo *et al.*, 1995).

Land occupation and timber extraction through conventional industrial logging has generated a culture of timber mining in many forest landscapes in the tropics, which has proved to be very persistent among some local stakeholders making an income from industrial timber extraction, which translates into low investments in operations or forest recovery. These cultural aspects of timber extraction in the tropics have been little studied, as well as shifts in social perceptions over time.

3.3.4.4 Uses

Like with the other practices reviewed in section 3.3, available knowledge on logging for a variety of uses was reviewed. In the case of logging, the relevant uses include decorative and aesthetic, energy, and shelter and construction. While it is certainly the case that many wood and tree products are used for ceremonial and cultural expression, food and feed, and medicine and hygiene, based on the definition of logging used in this assessment, these other uses (and the associated tree products) are discussed in the section on gathering (3.3.2).

3.3.4.4.1 Decorative and aesthetic

Harvesting timber for wood carvings is mainly a destructive process. The entire tree is felled at the trunk between 5 and 50 cm from the ground using a metal axe or chain saw and machetes, and the artists cut different lengths of timber from the fallen tree for their carvings (A. D. Griffiths, Philips, & Godjuwa, 2003; Koenig, Altman, & Griffiths, 2011; Purata, Brosi, & Chibnik, 2004). In other instances, only the prime section of the stem is removed, leaving the rest of wood in the forest. (B Belcher *et al.*, 2002). Tree sizes are selected based on the size and nature of the sculpture to be made. This can depend on the cultural subject matter (Koenig *et al.*, 2011). The average diameter of trees harvested for small sculptures such as birds would be smaller than those harvested to make canoes. However, the average diameter of trees harvested has significant implications on the sustainability of the tree species. Cutting down smaller sized trees before they produce and disperse seeds could affect the population of the tree species.

Wood for woodcarvings continues to be mainly harvested from the wild (Ellery, Cunningham, & Choge, 2005; Griffiths *et al.*, 2003; Purata *et al.*, 2004). These include forests on public land, communal and private forests (Ellery *et al.*, 2005; A. D. Griffiths *et al.*, 2003; Koenig *et al.*, 2011; Matose, 2006). However, there are no certain global statistics of volumes of wood extracted for wood carvings as this is primarily an informal activity (Ellery *et al.*, 2005), however that does not mean it is small in scale and scope. In 2002, woodcarving in Kenya were estimated to consume 50,000 trees per year (0.7% of the total round wood market share in Kenya). But although the amount of wood extracted for the purpose seems low, the wood carving practice relies on a selected number of species with desired qualities such as close grain, tensile strength and resistance to cracking or insect attack. In addition, a small range of different timbers are often favored as a result of social, cultural and historical factors (Cunningham *et al.*, 2005), which end up being over exploited. This has led to over exploitation of the particular wild species, especially those with other purposes, of which some are listed among the endangered species (Cunningham *et al.*, 2005; Ellery *et al.*, 2005).

From carving small household items, to carving the interior and exterior of houses and temples, ritual objects and decorative pieces, fashioning idols for various articles of furniture and for ceremonial objects (Saville, 1925), carving traditions have mainly been associated with culture, technology and change (Cunningham *et al.*, 2005). The practice was mainly associated with particular communities stretching back many generations, carving particular types of pieces that were mainly associated with long standing cultural significance. For example, in the tropics, subtropics, pre-industrial societies of Europe and some northern temperate regions, woodcarvings were and for some, are still the major sources of social and cultural materials

(Cunningham *et al.*, 2005). Whereas the practices have been socially and culturally sustainable among some woodcarving communities such as the Aboriginal wood carvers of Australia (Koenig, Altman, Griffiths, & Kohen, 2007), some communities such as those in the Mexican state of Oaxaca are engaged in carving novel creations without longstanding cultural significance (Purata *et al.*, 2004).

The wood carving industry has grown tremendously over the years, extending beyond the local and national to the international markets (Altman, 2005; Ellery *et al.*, 2005) which has increased demand of the wood carvings. The carvings are sold in a number of arenas including family workshops, markets and craft shops, either within the villages or other cities and countries (Purata *et al.*, 2004). An activity that was once predominantly a men's activity (Cunningham *et al.*, 2005) has progressively increased number of women and youths involved, becoming a family activity (Koenig *et al.*, 2007; Purata *et al.*, 2004). The women involved are mainly spouses and children of prominent wood carvers (Koenig *et al.*, 2007). However, these are mainly involved in the less labor-intensive activities such as sanding, polishing and painting (Matose, 2006; Purata *et al.*, 2004).

Other than the aesthetic values, these products have earned households, communities and national economies income. Wood carving is a major source of income through facilitating purchase of livelihood needs (Purata *et al.*, 2004) especially among communities in dry environments that suffer from lack of agricultural opportunities (Matose, 2006). However, it is not possible to obtain exact numbers of people involved (Ellery *et al.*, 2005) and the value of the industry as a whole is hard to determine (Griffiths *et al.*, 2003) due to its dynamic nature. The wood carving industry in Kenya generates about 20 million United States dollars per year in export revenue (Choge, 2002; Obunga, 1995), employing about 40% of thef national formal timber industry (Ellery *et al.*, 2005). Around the Victoria falls in Zimbabwe, the industry provides a source of livelihood to nearly a thousand households in a dry part of the country with households getting around 14 to 60 United States dollars a month.

The timber trade and woodcarving are closely linked, particularly in Asia, where timber is intricately carved to make buildings, doors or furniture. Trade in carvings is not new. It is bigger than ever before; however, it has spread internationally, rather than regionally, and has focused on a far smaller resource base (Cunningham *et al.*, 2005). Since many of the species used for wood carvings are endangered/threatened, their use and trade are restricted by both national regulations (for example sandalwood is restricted by the Kingdom of Tonga sandalwood regulations 2016, Tamil Nadu sandalwood possession rules, 1970, and the sandalwood (limitation of removal of sandalwood) order 1996 in Western Australia) and the Convention on International Trade in Endangered Species of Wild Fauna and Flora (Groves & Rutherford, 2015). Nevertheless, there are some initiatives to ensure sustainability of these species by both community and corporations. For example, some species have been in cultivation through plantation establishment and agroforestry practices; In Bali woodcarving has been put on a sound basis through shifts to a fast-growing species like *Paraserianthus falcataria*. In India there is the roadside, village-level and plantation production of *Dalbergia sissoo*. In coastal Kenya there are plantations of the neem trees. There is recommendation and adoption of community/corporate tree plantations for sandalwood (A. N. A. Kumar, Joshi, & Ram, 2012) in different parts of India with appropriate incentives and adequate protective measures. Australia has been raising large sandalwood plantations, and may be able to meet the global demands, with the world's largest plantation of *S. album* established in the Kimberly, Western Australia.

3.3.4.4.2 Energy

Energy security is one of the requirements for a good quality of life, and this includes availability and access to clean, reliable, affordable and sustainable energy without compromising health (UN, 2015). Yet globally, 1.1 billion (14%) people do not have access to electricity and 2.4 billion (approximately one-third of the global population) people rely on unclean 'traditional biomass' for energy (including charcoal, coal, crop waste, dung, kerosene and wood), with the associated health implications from household air pollution (IEA, 2017) **(Figure 3.57A)**. Wood energy contributes 75-90% of sub-Saharan Africa's household energy mix (Hoffmann, Brüntrup, & Dewes, 2016; World Bank, 2011). An estimated 880 million people globally log firewood or produce charcoal (FAO & UNEP, 2020). Reliance on wood biomass for cooking is highest in developing Asian countries and sub-Saharan Africa (IEA, 2017). One third of the world's population (2.4 billion people) use fuel wood for cooking – which provides more nutrients than raw food - and other food preservation processes (e.g., smoking, drying), and one in ten people use fuel wood for boiling and sterilizing water (FAO & UNEP, 2020).

Most wild biomass energy is derived from wood, with implications for social and natural systems (Arnold *et al.*, 2006; Bailis *et al.*, 2005; Holdren *et al.*, 2000; Miah *et al.*, 2009; Munalula & Meincken, 2009; Smith & others, 2006). Logging for energy accounts for 50% of all wood consumed globally, and accounts for 90% of logged timber in Africa (FAO & UNEP, 2020). According to the Food and Agriculture Organization of the United Nations' statistics (http://www.fao.org/faostat/en/#data/FO), global fuel wood removals have increased over time with approximately 2 billion m^3 produced in 2019. There is great variation in fuel wood production and use in the different regions.

Production is highest in Asia and Africa at approximately 713 million m³ and 706 million m³ respectively **(Figure 3.57A)**. Whereas production levels are increasing in Africa (from 445 million m³ in 1990), the opposite is happening in Asia whose production has decreased from 897 million m³ in 1990. Latin America and the Caribbean have a fairly high level of production (268 million m³). Oceania has the lowest production of approximately 10 million m³ in 2019. Although absolute fuel wood consumption is increasing, especially in sub-Saharan Africa, per capita consumption is decreasing across all regions **(Figure 3.57B)**. All regions reported minimal trade in fuel wood, implying that fuel wood are mainly consumed locally and in domestic markets. However, wood-based energy industry has the potential to grow in a number of countries. This has motivated the investment in biomass-based energy generation, and research and development of new energy products such as biodiesel (Asikainen *et al.*, 2010). Although alternative energy sources reduce demand for fuel wood, in some areas fuel wood use persists due to habits, taste and custom (FAO, Schure, Ingram, & Yoo, 2017).

In several industrialized countries, wood energy provides nearly 25% of total energy supply, and the leading renewable energy source in Europe accounting for about 45% of primary energy from renewable sources (Francisco X. Aguilar, FAO, & UNECE, 2018). With the requirement of European Union states to have 27% of their energy generated from renewable energy by 2030 (European Commission, 2014), Europe's wood consumption for energy generation is expected to grow and reach 752 million m³ in 2030 (Mantau *et al.*, 2010). Logging for energy in North and Central America has been growing to meet increasing export demand for wood pellets.

In Europe and North America, wood energy utilization is commonly integrated in forest management practices and the wood products industry. Wood energy feedstocks can be considered a co-product of forest management as part of silvicultural treatments inclusive of thinning, final integrated harvests and salvage logging, as well as a by-product of the forest industry during the production of sawn goods (Asikainen *et al.*, 2010). Most of the wood used for energy comes indirectly through the forest industry as a co-product (58%) and a little over of a third of the wood mobilized for energy comes directly from forests (36%). Data from the Joint Wood Energy Enquiry for 2013 shows that the forest-based industry was the largest consumer of wood energy (44%), followed by the residential (36%) and combined heat and power (17%) sectors (F. X. Aguilar, Glavonjić,

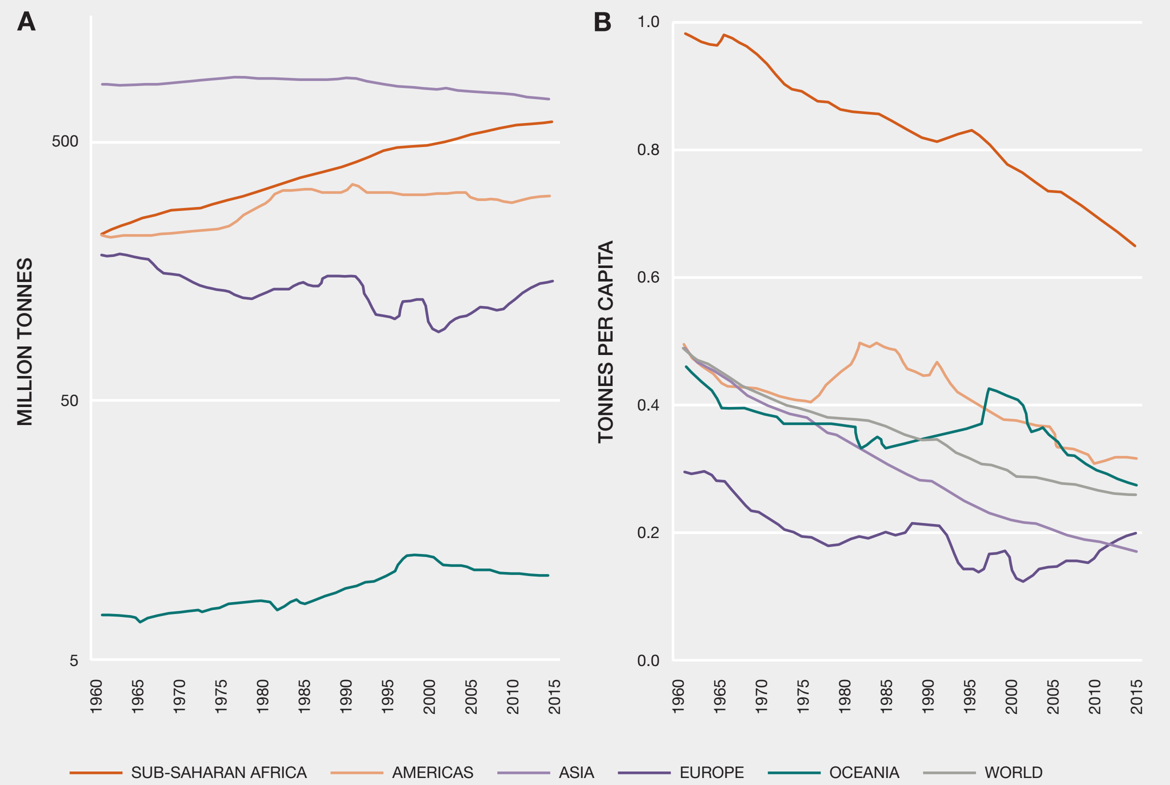


Figure 3 57 **Total (A) and per capita (B) fuel wood consumption trends by region.**

Source: (FAO, 2017b) © FAO, 2017.

Hartkamp, Mabee, & Skog, 2015). Use of wood for energy creates job opportunities not only along the supply-chain of woody biomass feedstocks, but also through investments in technology development and energy conversion and final consumption (Francisco X. Aguilar *et al.*, 2018). The FAO and the United Nations Environment Program (2020) estimated that over 40 million people are involved in commercial fuel wood activities to supply urban centers. Furthermore, fuel wood production generated an estimated 33 billion United States dollars in 2011 global revenue (FAO & UNEP, 2020). The number of jobs and net earnings is influenced by production method and organization of energy systems. For instance, the utilization of 390,000 dry tons of woody biomass estimated to feed a 100 megawatt power facility in the southern United States of America has been estimated to support 585 direct and 481 indirect jobs through the recovery of logging co-products, while direct and indirect employment associated with operation of the power plant were 281 and 115, respectively (Perez-Verdin, Grebner, Munn, Sun, & Grado, 2008).

Global fuel wood demand peaked in the mid-1990s (Arnold, Köhlin, Persson, & Shepherd, 2003), instigating a declaration of a 'fuelwood crisis'. However, the projected fuel wood supply-demand models predicting fuel wood stock collapse were an overestimation due to limited data and an incomplete understanding of social, economic and ecological interactions around wood energy (Dewees, 2020). Nevertheless, the available amounts of fuel wood may not be sufficient to meet local energy needs (Fabian, Volkmer, & Wiedemann, 2011; Swinkels, 2014). Household energy consumption is usually higher than fuel wood reported in official statistics, which mainly refer to wood from forests sources while leaving out other forms of wood biomass that contribute to household energy production. These include (for example in Europe) all by-products (sawmill by-products, other industrial wood residues and black liquor), solid wood fuels and post-consumer wood (Mantau *et al.*, 2010).

Although fuel wood demand can be met at a global, national or even regional scale, when comparing supply-demand balances, localized wood fuel scarcity persists (Arnold *et al.*, 2003; FAO *et al.*, 2017; Masera, Bailis, Drigo, Ghilardi, & Ruiz-Mercado, 2015). Fuel wood-scarcity 'hotspots' occur in areas where fuel wood is crucial for subsistence use and household-level well-being **(Figure 3.58A)** (Arnold *et al.*, 2006; Sampson *et al.*, 2005). In these areas, fuel wood users have few to no alternatives for cooking and heating, posing localized fuel wood driven challenges as most fuel wood (particularly firewood) is produced, harvested and consumed at a local level (Sampson *et al.*, 2005). In addition, regions where logging rates exceed growth rates are likely to cause degradation or deforestation (Robert Bailis, Drigo, Ghilardi, & Masera, 2015; Masera *et al.*, 2015). In 2009, 27-34% of fuel wood logging exceeded growth rates, predominantly in hotspots in South Asia and East Africa, affecting over 250 million rural people reliant on wood energy **(Figure 3.58B)** (Masera *et al.*, 2015). The FAO estimate that one third of fuel wood logging was unsustainable and a major cause of forest degradation (FAO *et al.*, 2017; FAO & UNEP, 2020). However, the link between fuel wood logging and deforestation or forest degradation is challenging to quantify and varies temporally and geographically (Rob Bailis, Wang, Drigo, Ghilardi, & Masera, 2017). Demand depends *inter alia* on household level preferences and economic context, vegetation species composition and physiognomy, the availability and cost of alternative energy sources (Rob Bailis *et al.*, 2017). Supply may vary with land use, productivity (and associated edaphic and climatic determinants), and accessibility of wood (Rob Bailis *et al.*, 2017). To further complicate quantification of fuel wood extraction, logging locations are not always from forests but are derived from many types of land cover (e.g., farms, roadside commons, home gardens), and may be a primary (logging specifically for fuel wood) or secondary activity (e.g., wood cleared from farms) (Rob Bailis *et al.*, 2017). Thus, fuel wood logging is often not the sole cause of forest degradation, but unsustainable fuel wood logging in Africa, particularly charcoal logging in open access systems with uncertain or unclear forest tenure, can be the primary driver of forest degradation (FAO *et al.*, 2017). The FAO found that fuel wood sustainability is strongly related to forest management rights and access, especially through permitting and/or taxation systems developed with local participation (FAO *et al.*, 2017). However, beyond these areas of localized shortages, sustainably logged fuel wood has the potential to be a viable, renewable, energy source that provides income (FAO *et al.*, 2017) and may be the preferred fuel source for cultural and economic reasons (P. Munro, van der Horst, & Healy, 2017), provided air quality (indoor and outdoor) and climate change emissions are mitigated (Rob Bailis *et al.*, 2017).

In low-income countries, fuel wood use occurs predominantly at the household scale for lighting, cooking and heating, but can support local and village-level industry. Commercial involvement with fuel wood, both firewood and charcoal, provides supplemental or an occasional income source (Arnold *et al.*, 2006), or an activity to fall back on as a 'safety net'. For example, firewood trading (and household subsistence use) increased in South Africa during economic shocks, such as loss of urban employment or breadwinner death as a result of HIV/AIDS (Human Immunodeficiency Virus) (Shackleton & Shackleton, 2004) or in response to the covid-19 pandemic, such as the switch from liquefied petroleum gas to fuel wood in Kenya and Malawi (Shupler *et al.*, 2020; Zalengera *et al.*, 2020). Eastern Afghanistan's forests have been an important energy resource during conflict-related crises in the region, although the forest wood stocks have been severely depleted (UNDP, 2014).

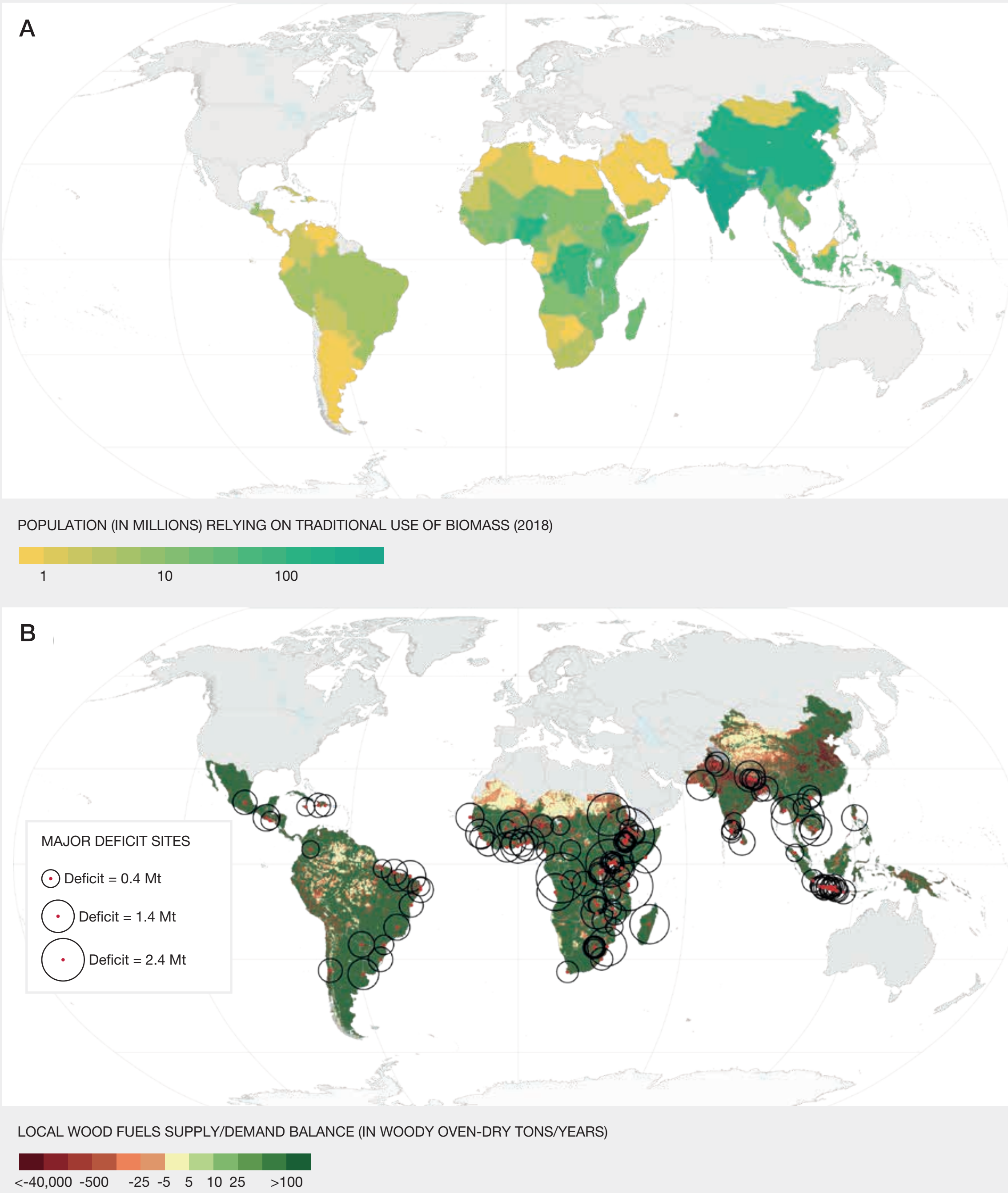


Figure 3 58 **(A) Global population reliant on traditional biomass, including fuel wood and animal waste, and (B) fuel wood supply/demand balance with circles on major deficit "hotspots".**

Sources: **(A)** based on data from (IEA, 2020) © OECD/IEA; **(B)** (Masera *et al.*, 2015). See data management report for the figure at https://doi.org/10.5281/zenodo.6453135.

Firewood trade can occur in conjunction with farmland clearing, with fuel wood being sold to wealthy urban clients through formal channels (Chidumayo & Gumbo, 2013; FAO *et al.*, 2017; Gandar, 1994). Firewood is normally logged on foot, limiting the logging radius to 1-3km, although this distance can be higher in arid regions with low tree cover (Cardoso, Ladio, & Lozada, 2013). Increasing firewood demand, locally and in urban areas has resulted in logged wood being collected and transported vehicle or horseback (Cardoso *et al.*, 2013; Matsika, Erasmus, & Twine, 2012), and may be by 'outsiders' (W. Twine, Saphugu, & Moshe, 2003). Localized shortages have also resulted in increased harvest time, changes in collected species or size classes, or harvest of live wood during deadwood shortages, often in violation of local traditional knowledge (Findlay & Twine, 2018).

Higher income is associated with a reduction in both firewood and charcoal use, although there is substantial variation between countries (Arnold *et al.*, 2003). The expectation that provisions of cleaner, more efficient energies and stoves would result in traditional energy users transitioning up the 'energy ladder' have largely not occurred, with households 'stacking' fuel, i.e., using multiple devices and fuels (Arnold *et al.*, 2006; Hiemstra-Van der Horst & Hovorka, 2008; Masera *et al.*, 2015; van der Kroon, Brouwer, & Van Beukering, 2013). Fuel wood use at household level is inelastic for a variety of reasons, including cultural and household taste preferences, high capital cost of appliances and energy, poor infrastructure and reflects dynamic, complex decision making at a household level (Arnold *et al.*, 2006; Hiemstra-Van der Horst & Hovorka, 2008; IEA, 2017; Masera *et al.*, 2015; van der Kroon *et al.*, 2013). There are indications that energy stacking, and the availability of a diversity of energy resources represents the adaptive capacity of communities, favors the conservation of local species and contributes to broader social-ecological resilience (Cardoso *et al.*, 2013). Energy stacking is a complex phenomenon, illustrated by a case in Patagonia, Argentina. A local village reliant on costly firewood, received subsidized liquefied petroleum gas (Betina Cardoso & González, 2019). This drastically reduced the amount of fuel wood collected in the region and reduced household air pollution, but not only did households continue to use wood burning stoves, their gas consumption to heat poorly insulated houses was extremely high (650 kilowatt-hour/m^2) incurring substantial operational and environmental costs (M. Betina Cardoso & González, 2019). The study's recommendations were two-fold: insulate the houses and receive a return on investment in liquefied petroleum gas savings in 2.2 years, and consider subsiding a household preference-determined mix of cheaper fire wood and gas to reduce subsidization costs (M. Betina Cardoso & González, 2019). This example clearly demonstrates the complexities in altering relative energy mixes and the potential trade-offs to social, economic and the environmental conditions.

Although firewood use is slowly decreasing, charcoal demand in urban areas is growing, doubling over 25 years to about 207 million m^3 wood for charcoal per annum in 2000 (Arnold *et al.*, 2003). In tropical South America charcoal use varies across the region with Brazilian use mainly for manufacturing and Central American for the food industry and limited domestic use (Chidumayo & Gumbo, 2013). In sub-Saharan Africa charcoal is mainly used for household cooking, particularly in urban areas (Chidumayo & Gumbo, 2013). Rural-urban charcoal trade is increasing as wealthy, urban firewood users 'transition' to charcoal (Arnold *et al.*, 2006). For example, 81% of energy use in Mozambique is fuel wood, with charcoal the predominant use in urban areas with the capital city, Maputo, garnering the highest prices for charcoal (Cuvilas, Jirjis, & Lucas, 2010). It is estimated that 91-99% of charcoal production is illegal (Cuvilas *et al.*, 2010). The high value and demand of charcoal in urban areas further incentivizes increased production (Cuvilas *et al.*, 2010). Charcoal production from plantations is increasing in the global tropics and charcoal is still predominantly derived from wild species in natural forests, and frequently related to deforestation or forest degradation (Chidumayo & Gumbo, 2013). Charcoal logging alone have resulted in the loss of 3 million hectares of forest cover in 2009 (Chidumayo & Gumbo, 2013). In Southern Africa charcoal production is valued at about 2-3% of gross domestic product (Malimbwi *et al.*, 2010) and forms a significant income source with households able to earn 1000 to 10,000 United States dollars per annum although studies suggest this income is not sustained over the long-term and does not provide improvements in human well-being (Baumert *et al.*, 2016; Smith *et al.*, 2019). Like firewood, the broad extent of charcoal logging and impacts remain unknown due to the informal and dynamic use of the resource.

Wood for firewood and charcoal are usually cut from main branches or the main stem, leaving the stump rooted in the ground. Many tree and shrub species logged for energy regenerate vegetatively, sprouting from the cut or damaged trunk, although the rate of coppice growth varies across species (Neke, Owen-Smith, & Witkowski, 2006), environmental context, post-logged land-use (Chidumayo & Gumbo, 2013), and the type of logging (Shackleton, 2000). Resprouting is a major source of regeneration in dry tropical forests and woodlands (Chidumayo, 2013; Tredennick & Hanan, 2015) and temperate forest regions, forming part of rotational logging management. A review of charcoal production reported 9-12 years logging rotations for Mali, Niger and Burkina Faso, 10-15 years for Mexico, 20-30 years in Zambia, and a wide 8–23-year range in Tanzania (Chidumayo & Gumbo, 2013). Underestimated coppice regeneration post-firewood and charcoal logging is one of the reasons that the 'fuelwood crisis' in which biomass stocks were predicted to collapse, has not occurred (Arnold *et al.*, 2003; Mograbi *et al.*, 2019; Twine & Holdo, 2016). Despite the significant productivity of woodlands and forests, fuel wood logging can alter floristic composition and vegetation structure (Mograbi *et al.*, 2015; Tredennick & Hanan, 2015). Depending on the fuel wood logging intensity, these ecosystem changes can alter the amount or type of nature's contributions to people derived from the forests (Chidumayo & Gumbo, 2013). For example, in Mozambique, charcoal production led to a reduction in firewood and construction material resources, with other natural resources such as wild food, medicinal plants and grazing mostly unaffected, although these trade-offs were mediated by village position and woodland resource characteristics (Woollen *et al.*, 2016). Variable ecosystem regeneration potential and context-specificity of environmental impacts and trade-offs in fuel wood logging are challenging to incorporate into large scale policy and management plans because of the social-ecological complexity and non-linear responses occurring across spatial and temporal scales.

Gender is one of the predominant features of traditional energy harvest, use and management of wood energy, including their (lack of) involvement in trade of fuel wood. While much of the research and interventions on inequality in forest resource use and management, many of the same challenges and barriers are faced by other vulnerable groups, such as minority ethnic groups, migrants, indigenous peoples, youths, landless people and other socially-differentiated groups such as lower castes (Chaudhary, McGregor, Houston, & Chettri, 2018; Kristjanson *et al.*, 2019). Gender gaps exist in almost aspects of natural resource use and management, including: disparities in participation; leadership; resource, land access, and tenure; forest use; division of labor and workloads; skills; access to technologies and inputs; access to information; access to forest services; access to benefits; access to credit; access to markets; policy engagement; and forest laws and regulations (Kristjanson *et al.*, 2019). With respect to the use of wood fuel for energy, women bear the majority of the responsibility for logging and using wood fuel (Clancy, Ummar, Shakya, & Kelkar, 2007; IEA, 2017; Murphy, Berazneva, & Lee, 2018). Households spend 1.4 hours a day harvesting fuel – a significant amount of time for women and children that could be used on other livelihood activities and education (IEA, 2017). The physical burden of headloads is not insignificant with a bundle weighing between 25-50 kg (IEA, 2017). Lack of access to clean cooking methods also has implications for household health (IEA, 2017), with women and children the most vulnerable to household air pollution which is a major cause of death and illness in low-income countries (Masera *et al.*, 2015).

An innovative approach to track how women are benefitting from interventions in forest resource use, trade and management is the W+ certification standard (WOCAN, 2020). The standard was created to measure women's empowerment and to accelerate investment to address gender inequality in access to resources and capital, specifically targeting improvements in: time, income and assets, health, leadership, education and knowledge, and food security (WOCAN, 2020). The standard provides certification for economic development and environment projects that improve socio-economic conditions for women. Benefits accrue to women through involvement in certified projects as well as from direct payments to women from the sale of W+ certification credits (WOCAN, 2020). Successful application of the W+ programme has demonstrated that interventions that save time and improve wood fuel efficiency are especially beneficial to women (Kristjanson *et al.*, 2019). W+ certification involving biogas digester projects in Nepal and Indonesia have resulted in tangible time and energy savings for women with improvements in income, assets and leadership capacity (Kristjanson *et al.*, 2019). Uptake of more fuel-efficient stoves has the potential for environmental benefits too. A case study in China documents a successful social media campaign to improve fuel-efficient stove uptake (DeWan, Green, Li, & Hayden, 2013). After two years, 43% of households had incorporated the stoves into their use, saving 40.1% on gathering time, and in the process saw a 23.7% reduction in newly felled trees in areas crucial to the conservation of the Sichuan Golden Snub-nosed Monkey (DeWan *et al.*, 2013).

Whilst gender inequalities are certainly a rights-based issue (S. Chaudhary *et al.*, 2018; Clancy *et al.*, 2007; Rights and Resources Initiative, 2014), investment in targeted programmes for women are opportunities for the sustainable management of forests and poverty relief (Kristjanson *et al.*, 2019). Ingram *et al.* (2016) document cases where male and female headed households harvest the same amount of wood, yet male households earned over three times more. In a Kenya study, women earn less than men in trading wood, and woodlots were mainly managed by men (Murphy *et al.*, 2018). Yet women's expenditures and increased roles in household expenditure decisions are associated with improvements in household nutrition, health and education (Ingram *et al.*, 2016). Women's income is a major determinant of household fuel choice and use (van der Kroon *et al.*, 2013). Thus, gender responsive interventions in training and enabling women to access markets and boost income can serve as leverage points for improving community well-being (de Groot, Mohlakoana, Knox, & Bressers, 2017; Ingram *et al.*, 2016). Similarly, opportunities for improved ecosystem health as empowering women's leadership and technical capacity building have been found to improve sustainable management of forests (Mwangi & Mai, 2011; Mwangi, Meinzen-Dick, & Sun, 2011). However, if fuel wood demand declines significantly, there are many women reliant on fuel wood sale income that will have reduced earnings in the event of a lack of alternate opportunities (IEA, 2017).

3.3.4.4.3 Material and construction

To have an idea of the amount of timber converted into wood for different purposes (sawn wood, energy, industrial round wood and paper and paper board), the assessment utilizes statistics on the production and trade of forest products over 245 countries and territories (FAO Stat, 2018). However, this does not include products from illegal timber trade.

Industrial round wood

Industrial round wood is all roundwood used for any purpose other than energy. It comprises pulpwood, sawlogs and veneer logs. Global industrial roundwood removals have increased from 1.7 billion m^3 in 1990 to 2.0 billion m^3 in 2019 **(Figure 3.59)**.

The increase in production is across all the regions except North America. Europe and North America had significant decreases in production in 1995 and 2010 respectively, while Asia had its greatest increase in production in 2010. There is a slight increase in trade of industrial roundwood. In 2019, approximately 144 million m^3 and 138 million m^3 were imported and exported respectively, while 83 million m^3 and 83 million m^3 were imported and exported respectively in 1990. Asia is a net importer, importing about 30 million cubic meters higher in 2019 than in 1990. Other regions are net exporters. Europe is the main exporter followed by Oceania. Africa and the Latin America and the Caribbean import and export very minimal quantities of industrial round wood (data from the Food and Agriculture Organization of the United Nations; http://www.fao.org/faostat/en/#data).

Sawnwood

Sawnwood encompasses planks, sleepers (cross-ties), beams, joists, boards, rafters, scantlings, laths, boxboards and "lumber". There was an increase in sawnwood production from 463 million m^3 in 1990 to 488 million m^3 in 2019, with the largest increase in Asia **(Figure 3.60)**. There are significant decreases in production between the two points in time happening in 2000 in Asia, 1995 in Europe and 2010 in North America. There is an increase in trade of sawnwood with 149 million m^3 and 156 million m^3 imported and exported respectively in 2019 as compared to 84 million m^3 and 78 million m^3 imported and exported respectively in 1990. Asia and Africa are net importer of sawnwood while the rest of the regions are net exporters. Asia is the major importer, importing about 47 million m^3 more in 2019 than in 1990, while Europe is the major exporter, exporting 71 million m^3 more in 2019 than it exported in 1990 (data from the FAO; http://www.fao.org/faostat/en/#data).

Wood based panels

The wood-based panels' product category consists of plywood (including blockboard and laminated veneer lumber), particle board, oriented strand board and fibreboard. In 2019, approximately 358 million m^3 of wood-based panels were produced globally **(Figure 3.61)**. This is an increase of 234 million m^3 from a volume of 124 million m^3 reported in 1990. The major producers of wood-based panels are Asia, Europe and North America, with Asia reporting the most tremendous increase of production from 25 million m^3 in 1990 to 196 million m^3 in 2019. Trade in wood-based products has also increased between 1990 to 2019 from approximately 28 million m^3 of imports and exports in 1990 to 88 million m^3 of imports and exports in 2019. Europe is the major trader of the product, followed by Asia and North America (data from the FAO; http://www.fao.org/faostat/en/#data).

Paper and paper board

The paper and paperboard product group comprises graphic papers (newsprint, printing and writing paper) and other paper and paperboard. There is an increase in global production of paper and paper boards **(Figure 3.62)**. In 2019, approximately 404 million tons were produced, an increase of 165 million m^3 from production volumes of 1990. The major producers and traders of paper and paper boards are Asia, followed by Europe and North America. Production levels of North America have fallen by approximately 11 million tons between 1990 and 2019, while those of Asia and Europe have increased by 138 million tons and 25 million tons respectively within the same time intervals. Trade in paper and paper boards has also increased with 110 million tons and 113 million tons imported and exported respectively. Asia is a net importer while Europe and North America are net exporters (data from the FAO; http://www.fao.org/faostat/en/#data).

3.3.4.5 Emerging issues in logging and timber management

3.3.4.5.1 Covid-19 pandemic

The COVID-19 pandemic has led to disruptions in international trade and supply chains of timber and its products globally. Many developing countries are heavily dependent on international trade of these products and the pandemic is having a significant effect on production and consumption patterns. For example, recent developments in the timber markets have increased the dependency on Chinese demand. With the pandemic triggered decline of exported round timber to China, stockpiles of export products are being built up in some places. This is further exacerbated by limited demand in typically strong markets such as Austria and Germany, while export markets in France, Italy and Spain are essentially at a standstill. Together these factors have resulted in a decrease in export incomes in developing countries (FAO, 2020c). As a result, the least developed timber-producing countries, in particular, may suffer directly from plummeting export volumes of roundwood and other wood products (FAO, 2020c). Nevertheless, in the post-COVID-19 environment, the trade and consumption of legal and sustainable wood products may be promoted through sustainable forest management for wood production, and can play a crucial role in economic recovery, especially considering efforts to promote a circular bioeconomy and climate change mitigation (FAO, 2020b).

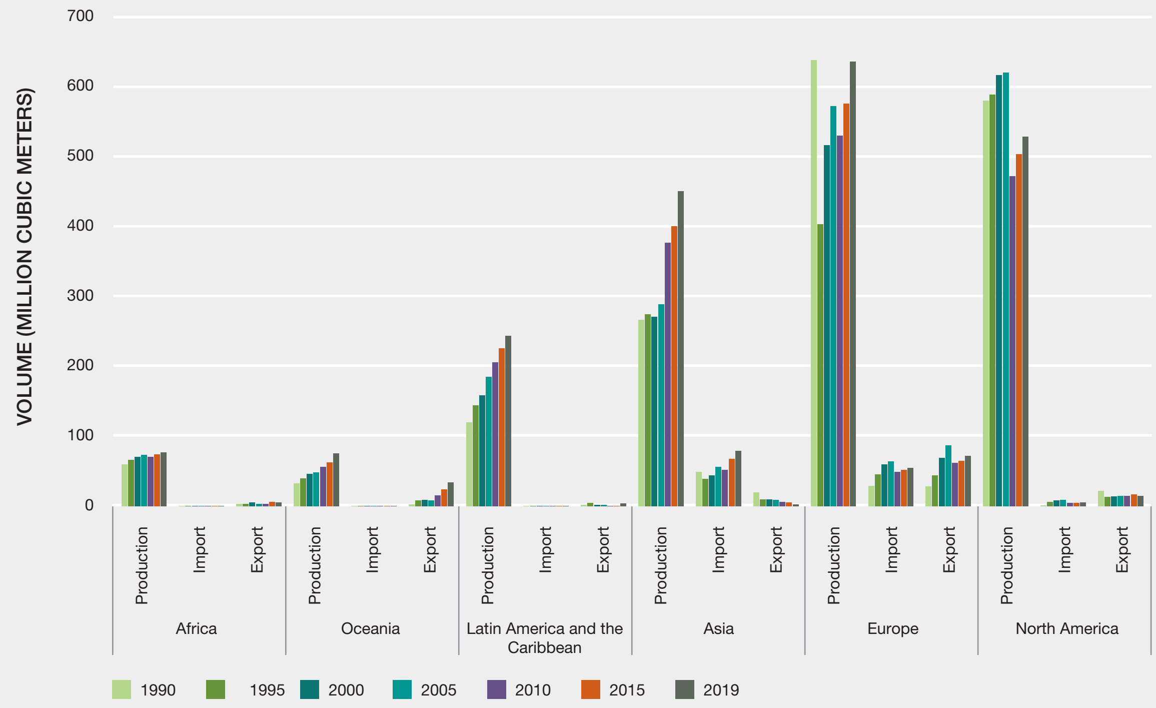


Figure 3 59 **Global trends in industrial roundwood use.**

Data from FAO database (FAO, 2021b) under license CC BY-NC- SA 3.0 IGO. See data management report for the figure at https://doi.org/10.5281/zenodo.6453131.

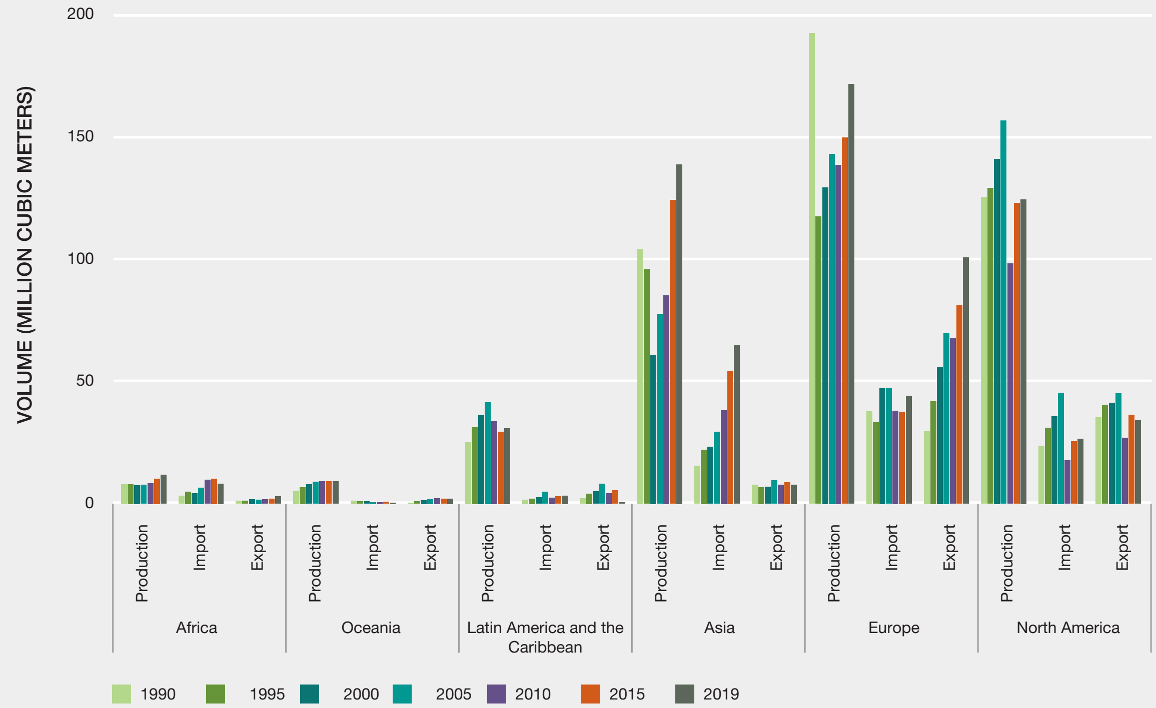


Figure 3 60 **Global trends in sawnwood.**

Data from FAO database (FAO, 2021b) under license CC BY-NC- SA 3.0 IGO. See data management report for the figure at https://doi.org/10.5281/zenodo.6453131.

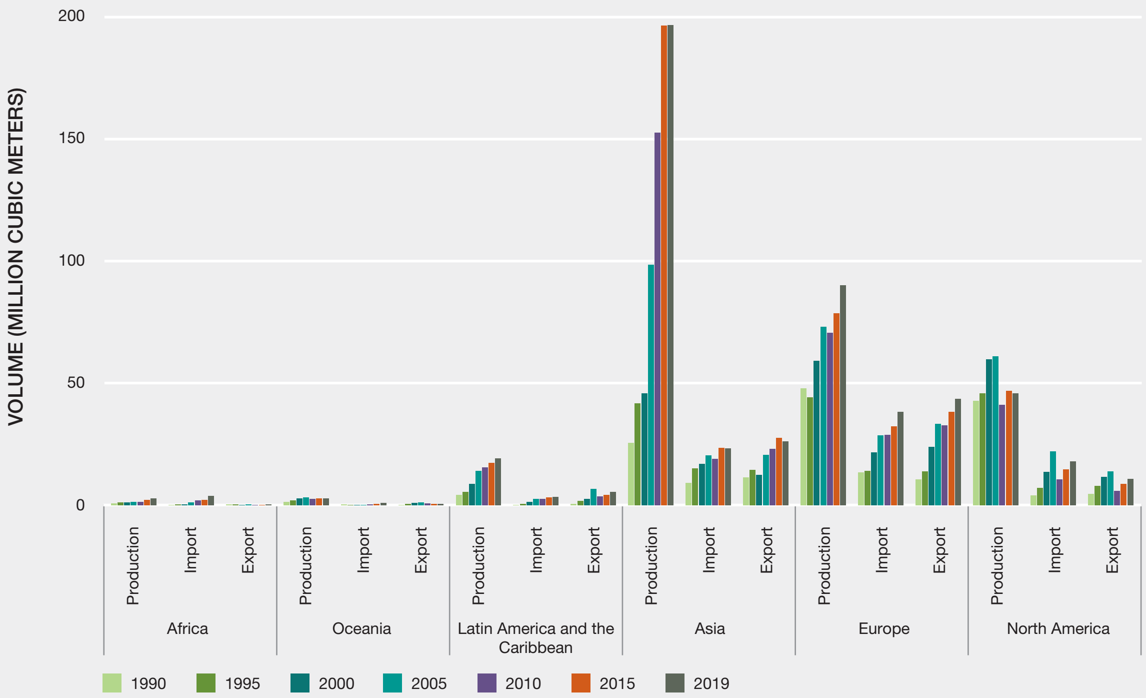


Figure 3 61 **Global trends in wood based panel production.**

Data from FAO database (FAO, 2021b) under license CC BY-NC- SA 3.0 IGO. See data management report for the figure at https://doi.org/10.5281/zenodo.6453131.

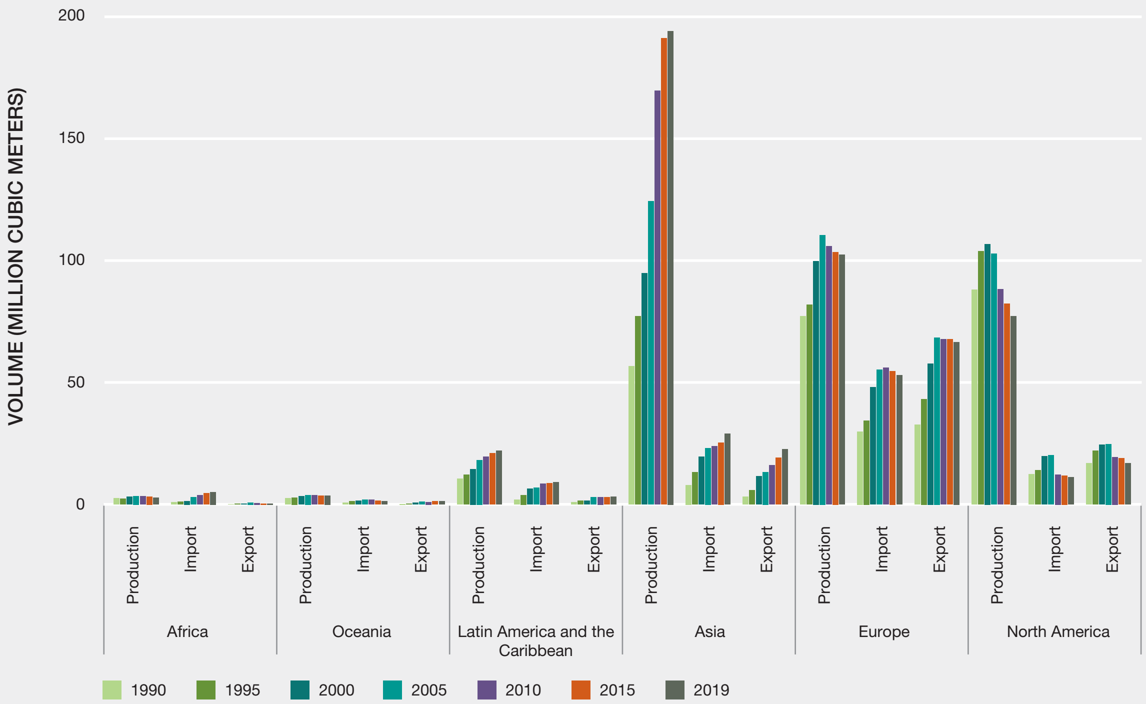


Figure 3 62 **Global trends in paper and paperboard production.**

Data from FAO database (FAO, 2021b) under license CC BY-NC- SA 3.0 IGO. See data management report for the figure at https://doi.org/10.5281/zenodo.6453131.

For indigenous people and local communities, negative effects of the COVID-19 pandemic on vulnerable communities, including women have been observed. Although, there has been steady progress made to date to empower women by supporting their participation in legal and sustainable fuelwood and charcoal production, the COVID-19 crisis is expected to put increasing pressure on forest resources through illegal charcoal production. Situations where livelihoods are put under significant pressure often tend to result in a shift towards activities with quick economic gains at the sacrifice of legal activities. In some countries, restrictions on travel and movements may affect the transportation and trade of fuelwood (particularly charcoal) from production sites to market centers (mostly urban areas). This may affect reliable access to energy for cooking in urban areas (FAO, 2020b). The COVID-19 pandemic also set the progress of universal access to electricity and clean cooking back, with the number of people without electricity access forecast to rise by 2% in 2021 (IEA, 2021). The economic shock of the pandemic also resulted in a return to fuel wood, with many people unable to pay for modern, clean fuels (IEA, 2021).

3.3.5 Non-extractive practices

3.3.5.1 Introduction: Significance of non-extractive practices

Non-extractive practices are widespread across the globe, occur in all ecoregions, and are essential to maintaining *inter alia* human relaxation, spiritual and cultural identity, connection to nature, belonging, sense of place, physical and psychological health, and inspiration.

The contributions of wild species to people from non-extractive practices are often intangible and resist commodity-based valuation (with the exception of recreational tourism). Yet many of the non-extractive contributions from nature are core to the human experience and contribute to the well-being of people (Millennium Ecosystem Assessment, 2005; Russell *et al.*, 2013). Knowing and experiencing nature is the foundation of cultural expression and identity; is inherent in the concept of biocultural diversity; forms the backdrop for social connections, religious experience and beauty; as well as contributing substantially to gross domestic product and local livelihoods (Russell *et al.*, 2013).

Although extractive practices are often the focus in the debate on what constitutes sustainable use of wild species (e.g., Abensperg-Traun, 2009; Di Minin *et al.*, 2019; Link & Watson, 2019; Nijman, 2010; Zeller & Pauly, 2019), non-extractive practices may also have sustainability implications, both for wild species and for human well-being. Although, non-extractive practices, by its very definition, are viewed as having less of a direct impact on wild species and ecosystems than extractive practices, there are many documented detrimental impacts and sustainability concerns in this practice. This is particularly well-documented for the use of wild species for tourism and recreation (see Section 3.3.5.2.3.). However, many of the adverse impacts may be avoided or mitigated through context-based understanding and collaborative engagement with all stakeholders.

Non-extractive benefits from wild species and nature are similar conceptually to the definitions of cultural nature's contributions to people (Costanza *et al.*, 1997) and non-material benefits (Millennium Ecosystem Assessment, 2005). In this assessment, non-extractive practices are defined based on the observation of wild species in a way that does not involve the harvest or removal of any part of the organism. The observation can imply some interaction with the wild species, such as the activities of wild species tourism and whale watching or no interaction with the wild species, such as photography (see Chapter 1).

Just as in extractive practices, the social contextual heterogeneity in the contributions from wild species to human well-being through their non-extractive use has implications for equitable environmental decision making (Martín-López *et al.*, 2012). The contributions from wild species to human well-being are perceived and valued differently, which influences the type and extent of use (Pascual *et al.*, 2017; Satz *et al.*, 2013). This also means that there may be conflict between different users of wild species (Pascual *et al.*, 2017). For example, one study documented interpersonal conflicts both within and between two recreational user groups in Hawaii, scuba divers and snorkelers, that held different nature-oriented values, those who valued nature intrinsically and held protectionist beliefs *versus* an anthropocentric-utilitarian value (Philips, Szuster, & Needham, 2019). Similarly, local residents near North American ski resorts placed high emphasis on recreational access and came into conflict with city residents who preferred that the area remain pristine wilderness, unaffected by tourism activities (Saremba & Gill, 1991). One proposed solution to avoid these types of conflicts is to spatially or temporally partition regions that can cater for different stakeholder's values (e.g., demarcated fishing and diving zones).

There can also be a disconnect between the importance placed on non-extractive practices of nature at a local level, where they are used on a daily basis, and the level to which they are incorporated into regional, national and global decisions on ecosystem management, which are made from a more distanced level (Brondizio, Ostrom, & Young, 2009; S. Chaudhary, McGregor, Houston, & Chettri, 2019). Thus, governance systems play a large role in which non-extractive contributions from nature are delivered to people, by identifying which stakeholder group's expectations and values are recognized (Gladkikh, Gould, & Coleman, 2019; Martín-López *et al.*, 2012; Pert *et al.*, 2015).

3.3.5.2 Uses

Regarding non-extractive practices, the following uses are well-documented in the literature and available data sources: ceremony and cultural expression (section 3.3.5.2.1), medicine and hygiene (section 3.3.5.2.2.), recreation (section 3.3.5.2.3.), education and learning (section 3.3.5.2.4).

The documentation of the non-extractive practices of nature, especially the use by indigenous peoples and local communities, often does not include species described at a species level, but frequently as part of a functional group (e.g., trees in urban green spaces; worship of sacred groves). For many indigenous and local communities their worldview and experiences are intimately connected with nature (Klain, Satterfield, & Chan, 2014; Pert *et al.*, 2015). Indigenous and local knowledge is premised on the interdependence of what scientific knowledge may identify as distinct components of nature and culture, such that worship of sacred groves is a holistic practice that includes the species in the grove (e.g., forest plant and animal species community), the ecosystem processes (e.g., primary production, pollination), the landscape features (e.g., rocks, rivers), and the particular cultural practices and language of the human community. However, in order to keep the scope of this section pragmatic and practical, literature that deals with quantifiable and measurable use of wild species up to the taxa level (e.g., trees) has been included in this review on non-extractive practices and literature on landscapes and landscape components (e.g., sacred pools, sacred mountains) was excluded.

The following uses are not relevant to this practice and/or were not documented: decorative and aesthetic, energy, food and feed, materials and shelter. Although aesthetic beauty and inspiration of nature are a form of non-extractive practice, this was excluded to maintain the scope of the assessment to quantifiable and measurable impacts of sustainable use. Keyword searches and methods for each review are detailed in each subsection.

3.3.5.2.1 Ceremony and cultural expression

Ceremony and cultural expression refer to the use of wild species in spiritual observances and practices, valued for their role in maintaining cultural identity (Chapter 1). In the context of non-extractive practices, the use of wild species can be through worship of religious or culturally important species. In urban areas, similarly urban green spaces have become important analogues for worship and ceremonial rituals (Ngulani & Shackleton, 2019). Thus, wild species can underpin cultural and religious identity by supporting spiritual, intellectual and emotional features and contribute to literature, lifestyles, value systems, traditions and beliefs, and ways of living together. The use of wild species for ceremony and cultural expression supports social cohesion, belonging and identity (Satz *et al.*, 2013). Wild species form part of history and cultural narratives (Pascual *et al.*, 2017; Satz *et al.*, 2013). Thus, unsustainable use of wild species central to cultural and ceremonial engagement can harm social relations (Millennium Ecosystem Assessment, 2005). Inversely, restoration of degraded forests and landscapes provides an opportunity for cultural and indigenous value revival (Constant & Taylor, 2020).

The text below is based on a literature review (Web of Science) using the following strings of terms ("non-extractive use*" OR "cultural ecosystem service*" OR "non-material contribution*" OR "non-consumptive use") AND (spiritual* OR ceremon* OR religion* OR ritual*), generating 51 hits (see the data management report for Chapter 3 systematic literature review at https://doi.org/10.5281/zenodo.6452651). Articles that were recommended by citation databases were also considered, as well as harvested from personal libraries and recommendations from experts. The scope of this section is limited to the non-extractive practices of wild species for ceremonial and cultural expression where the impact of the use can be measured or assessed. After reviewing the title and abstract, 36 papers were selected for a full-text read. Relevance was determined by mention of either the status (current), trend (historical), or impacts of use of wild species (or taxa) in at least one dimension of sustainability (social, economic, or ecological). After a full text read of these papers, eight were deemed relevant and assessed for the literature review. These data form the basis of the text below.

Of the reviewed articles, half (4 out of 8) covered the importance of trees for ceremonial use, particularly sacred groves in Africa. Research on sacred groves was mostly anthropological and social data of long-term (>10 years) trends on a regional (<100 km) scale.

Sacred natural sites, such as sacred groves and burial sites, are an important feature across the world that can play a central role in biodiversity and resource conservation. These sites exhibit large diversity in their form and function, showing strong local context of both ecology and culture (Fournier, 2011; Juhé-Beaulaton & Salpeteur, 2017). Sacred groves are places of spiritual and cultural importance, protected by the authority of tribal taboos and spiritual "caretakers". In general, restrictions forbid cutting down or harvesting any part of the trees, including dead wood, to burn or harm the fauna and flora, or to remove soil or stones (Fournier, 2011). Depending on the tribal custom, picking herbaceous plants and grazing livestock near the sacred trees may be permitted (Fournier, 2011). Taboos also vary depending on the type of sacred grove. For example, the Tandroy clan in Madagascar allows the use of fire in honey groves – kept for medicinal, food, and spiritual purposes – but has more stringent taboos in burial forests (von Heland & Folke, 2014).

The current status of sacred groves, or indeed of any wild species used for ceremonial and cultural purposes, is not well documented in the literature. But the limited data that do exist are mixed, with some evidence that taboos and traditional beliefs have protected sacred groves. Sacred groves can an important role in community-based conservation of biodiversity, acting as refugia for species. For example, India possesses relict populations of certain threatened tree species (*Actinodaphne lawsonii, Hopea ponga, Madhuca neriifol*i, and *Syzygium zeylanicum, Myristica fatua* and *Gymnacranthera canarica*) in numerous riparian groves. Sacred groves in the Karnataka state also shelter a high diversity of macrofungi, 49 out of 163 species are unique to sacred groves (Bhagwat & Rutte, 2006). Similarly, in central Tanzania, greater woody plant species richness was found in sacred groves than in a state-managed forest reserve (Mgumia & Oba, 2003). Despite droughts and pressure to use resources inside sacred forests, the ancestral forests in Ambonaivo have been preserved whereas elsewhere in Madagascar, sacred groves have been cut down for charcoal production (von Heland & Folke, 2014). Similarly, sacred groves in Morocco have been effectively conserved as a result of their sacred status (Frosch & Deil, 2011).

Despite their significance, the protection offered to wooded shrines may be limited in extent and may only be for a certain period of time (Fournier, 2011). In Burkino Faso, the clearing of wooded shrines has also been blamed on "foreigners" fleeing worsening climatic conditions in the Sahel, who are (either intentionally or not) ignorant of local traditions (A. Fournier, 2011). In Benin most sacred groves have been neglected or cut down, but a few have been restored and are being managed for nature's contributions to people (Juhé-Beaulaton & Salpeteur, 2017). A vegetation assessment of wooded shrines in West Africa found more groves were cut down than restored and although they were less used for extractive purposes than similar secular forests, they were still being used for extractive purposes (Fournier, 2011). Sacred groves are also not necessarily ecologically 'pristine' by conservation standards. Whilst the preference by locals is for sacred groves to "have trees", preferably dense vegetation, the species of tree is considered unimportant and wooded shrines range from almost natural to highly modified (Fournier, 2011).

The literature suggests the future of sacred groves is strongly dependent on how spiritual and religious practices adapt to changing socio-political conditions. The degrading social contract with nature and the erosion of ancestral and natural connections threatens the sustainability of sacred groves. Cultural trends show taboos around sacred groves are eroding as the elder "spiritual caretakers" who play an active role in supervising use of the groves, are not replaced (Fournier, 2011; von Heland & Folke, 2014). There are also changes to "social-ancestor contracts" which are being modernized, and more of the local people have converted to global religions (Juhé-Beaulaton & Salpeteur, 2017; von Heland & Folke, 2014). The increasing assimilation of local peoples' moral order into one more closely aligned with modern, Western, democratic ideals governed by the nation-state has eroded the traditional and ancestral social-ecological system central to their identity, as well as the associated protection afforded to their land and the species it contains (Findlay & Twine, 2018; von Heland & Folke, 2014). As local protection erodes for sacred sites, there is an opportunity for more formal protection, such as the promising case developing in Estonia where Estonian indigenous peoples and local communities (Maausk) and the government are planning to confer legal protection to approximately 550 sacred groves (Kaasik, 2012). There are also opportunities to conserve sacred groves for purposes other than cultural worship provided the other uses are compatible and respectful of the sacred status. This has occurred in West Africa where sacred groves are also used for heritage and cultural tourism (Juhé-Beaulaton & Salpeteur, 2017).

The literature review on the use of wild species for ceremonial and cultural expression also described the use of urban green spaces for religious worship. However, increasing urbanization threatens the loss of green spaces used for worship, especially as these sacred sites are not associated with formal religious structures or buildings (Jackson & Ormsby, 2017). Use of urban green space for ceremonial purposes has been documented in Zimbabwe (Ngulani & Shackleton, 2019), Accra (Okyerefo & Fiaveh, 2017) and India (Gopal, von der Lippe, & Kowarik, 2019), but it is an underreported form of use of either formal or informal urban green spaces and has not received adequate research or policy attention (Jackson & Ormsby, 2017; Ngulani & Shackleton, 2019). No information on the use of urban green spaces for worship described whether this was an increasing phenomenon, or the sustainability of this use.

Overall, the use of wild species for ceremonial and cultural purposes is likely widespread but poorly documented. There are little to no data on the status and trends, or sustainability of this use. However, the literature on sacred groves do suggest that cultural erosion is driving a decreasing trend of ceremonial use, and thus also an erosion of traditional protection that the use afforded these species.

3.3.5.2.2 Medicine and hygiene

This section relates to the non-extractive practices of wild species for human health, both psychological and physical. The scope of this section is limited to the non-extractive practice of wild species for restorative and/or preventative effects and the impact this use has in terms of a measurable effect on the species. The text below is based on a literature review (Google Scholar) including the following search terms: ("non-extractive use*" OR "cultural ecosystem service*" OR "non-material contribution*" OR "non-consumptive use") AND (sustainab* OR "forest therapy" OR "human well-being" OR "human health" OR stress OR happiness OR dose-response) generating over 1 million hits (see the data management report for Chapter 3 systematic literature review at https://doi.org/10.5281/zenodo.6452651). Articles that were recommended by citation databases were also considered, as well as collected from personal libraries and recommendations by experts After a title and abstract read, 24 papers were selected for a full-text read. After a full text read of these papers, 13 were deemed relevant and assessed for the literature review. Relevance was determined by mention of either the status (current), trend (historical) or impacts of use of wild species (or taxa) in at least one dimension of sustainability (social, economic, or ecological). These data form the basis of the text below.

Relevant material from the review covered mostly the use of trees (10 papers) for health purposes, with a few mentions of terrestrial mammals and birds (4 papers). 46% of the studies on this topic were global overviews, with regional studies mostly representing Asia-Pacific and Europe Central. Relevant research was overwhelmingly short-term studies (<1 year), but spanned a variety of spatial scales: global, national, regional and local.

There was an absence of information in the literature on trends in the non-extractive practices of wild species for health and hygiene. From the review, only one paper tracked trends in use for health over time. This was a paper that documented current and past trends in the use of forests for forest therapy in Korea (Shin *et al.*, 2017). Similarly, no information was found reporting on the sustainability of health-based use of wild species on species or ecosystems. Although undocumented, negative impacts on wild species used for medicine and health likely include the effects of trampling during nature visits (see section 3.3.5.2.3. Recreational). It is possible that the health benefits obtained from wild species motivate people to support and protect their natural spaces. Research on environmental education supports that the more people learn from (and in) nature, the more likely they are to develop pro-nature behavior (M. Richardson, Cormack, McRobert, & Underhill, 2016). But this has not yet been documented for the non-extractive practices of wild species for medicine and hygiene use. It is not a given that the benefits provided by wild species always confer protection. For instance, street tree vandalism is a significant driver of urban tree mortality (e.g., Richardson & Shackleton, 2014) despite the numerous benefits provided by urban greening.

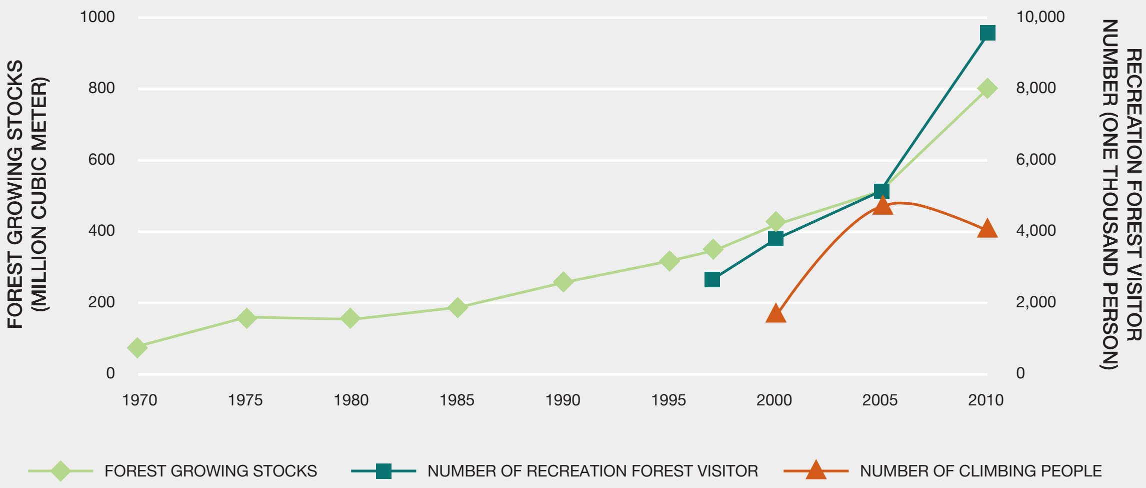
The literature on this topic extensively deals with the beneficial impacts of nature, especially forests, on *individual* human well-being. A significant research gap exists on the impacts of health-based use of wild species on human community health (Nesbitt, Hotte, Barron, Cowan, & Sheppard, 2017). The rest of this section will describe the evidence and examples of the impacts of health use of wild species on human individual's well-being.

Rapid urbanization and industrialization have been related to the rise in chronic mental and physical health problems, mostly associated with stress (Ashworth, 2017), costing millions in healthcare-related expenses and lost work days (Moore, Gould, & Keary, 2003). Thus, preventive measures, including nature-based remedies, to deal with the modern-day health crisis are economically prudent, and are supported formally by some governments, such as the legislation passed by the Korean government for the use of forests for health (Kotte, Li, & Shin, 2019; Shin *et al.*, 2017), or *shinrin-yoku* ("forest bathing") by the Japanese Forestry Agency (Rajoo, Karam, & Abdullah, 2020). There are also documented case studies of forest therapy, and the increasing demand for cost-effective preventive medicine and stress management using forest therapy, in Southeast Asia and Northern Europe (Kotte *et al.*, 2019; Lee *et al.*, 2019).

Shin *et al.*, (2017) documented a significant increase in the use of Korean forests for recreational visitors, primarily for forest therapy and the health benefits of spending time in "healing forests". This rise in health-based forest use has been facilitated by Korean government forests restoration programs, legislating certain forests specifically as "healing forests", and substantial investment in forest therapy research (Shin *et al.*, 2017). Although other studies mention that the demand for and use of nature for restoration and health has increased (Kotte *et al.*, 2019; Lee *et al.*, 2019; Rajoo *et al.*, 2020), the quantitative change in this use has not been documented **(Figure 3.63)**.

Reviews on the effects of forest therapy on human health found that most research reported positive effects (Frumkin *et al.*, 2017; Kotte *et al.*, 2019; Rajoo *et al.*, 2020; Wolf *et al.*, 2020). The benefits of natural settings for restorative effects, such as stress relief, decreased cognitive fatigue, and happiness (see Chapter 1), have been documented in both urban and non-urban settings. Natural settings have been associated with, amongst others, better cognitive functioning, fewer symptoms of depression and lower antidepressant use, reduced stress and psychiatric disorders, reduced diabetes, and improved immune function (see review in Frumkin *et al.*, 2017). Exposure to nature has also been shown to have a positive effect on infant birth weights, reductions in childhood obesity, and improved blood pressure (Aerts, Honnay, & Van Nieuwenhuyse, 2018; Frumkin *et al.*, 2017). Exceptions include the negative

A



B **Changes of growing stocks in Korea between 1970–2010; growing stock in Korea during 40 years. Comparison of scenery between 1970 and 2000 in Young-il Gyeongsangnam-do, Korea.**

1970s

2000s

C **Visitors of recreation forests and healing forests (2010–2013); Visitors of healing forest increased 11 times than visitors of recreation forests among 4 years.**

Visitors (number)	2010	2011	2012	2013	Mean increase rate (%)
Recreation Forests	9,437,000	10,684,000	11,614,000	12,780,000	10.6
Healing Forest	76,063	157,571	314,767	787,029	117.0

Figure 3 63 **The graph (A) and photos (B) show the recovery of forest stocks in Young-il Gyeongsangnam-do, Korea from 1970–2013.**

Concomitantly, the number of "recreational visitors" to "Healing Forests" have increased over time as the number of recreational climbers have declined (A) with a mean increase of 117% in healing forest visits over 3 years (C).
Source: (Shin *et al.*, 2017) under license CC BY-SA 4.0.

effects of plant pollen and volatile organic compounds from trees (Wolf *et al.*, 2020). Findings on the benefits wild species and ecosystems provide for mental and physical health have motivated for technology to provide this form of health benefit through virtual reality, and although exposure to nature through photographs or video does improve stress levels and reduce cognitive fatigue, the real experiences in nature significantly outperform virtual experiences for restorative benefits (Calogiuri *et al.*, 2018).

The studies mentioned above used a variety of self-reported measures to assess human well-being, with little research being done on clinical outcomes (Aerts *et al.*, 2018). The research was also mostly based on a limited set of variables to describe nature. The majority of studies were based on proximity to nature (e.g., Xiao *et al.*, 2019), or the number or cover of trees (Wolf *et al.*, 2020). There were fewer studies on the diversity of wild species for human well-being (Methorst *et al.*, 2021) and none was identified on specific wild species, rather focusing on functional groups such as trees or birds. The limited research on the effects of ecological quality (e.g., species richness) of trees suggests lower correlation with life satisfaction than overall abundance and denser tree cover, suggesting people were less affected by species diversity and more by the mere presence of trees (Marselle *et al.*, 2020; Methorst *et al.*, 2021), although the state of knowledge in this field is still mixed (Aerts *et al.*, 2018). Certainly, people have expressed preference for particular species, especially those that were aesthetically pleasing or reminded them of their childhood home (C.M. Shackleton & Mograbi, 2020).

Dose-response effects of wild species on human health have been demonstrated with trees and with birds. People living within 100 m of higher street tree density had lower antidepressant prescriptions (Marselle *et al.*, 2020). This effect was even more pronounced for individuals with low socio-economic status (Marselle *et al.*, 2020). A study exploring self-reported life satisfaction across Europe in relation to several taxonomic groups and socio-economic indicators found that bird species richness was highly correlated with life satisfaction, comparable with that of net household income **(Figure 3.64)** (Methorst *et al.*, 2021). Methorst *et al.*, (2021) hypothesize that the direct multisensory experience of birds and/or the supporting landscape properties that support bird diversity benefit human life satisfaction. Another study found that vegetation cover and afternoon bird abundance was positively associated with lower depression, anxiety and stress (Cox *et al.*, 2017). Cox *et al.*, (2017) modelled neighborhood vegetation cover thresholds at which population prevalence of mental health issues were significantly lower: more than 20% for depression and stress, and more than 30% for anxiety. A dose-response model suggested that visits to nature of 30 minutes or more a week could reduce population prevalence of depression by 7% and high blood pressure by 9% (Shanahan *et al.*, 2016). A significant reduction, especially considering that depression alone in Australia, where this study was conducted, was estimated at 12.6 billion Asutralian dollars per year (Shanahan *et al.*, 2016). A study from the United Kingdom of Great Britain and Northern Ireland found that individuals spending at least 120 minutes a week in nature reported better health and well-being relative to people spending no time outdoors; positive associations peaked at 200-300 minutes a week (White *et al.*, 2019). These "Green Prescriptions" highlight the importance of species presence and diversity to human well-being, a cost-effective means of supporting a healthy population.

There are significant socio-economic disparities in urban green space access, both as a result of restricted access (e.g., private space) and as a consequence of socio-economic class differentiation in urban planning (Venter, Shackleton, Van Staden, Selomane, & Masterson, 2020;

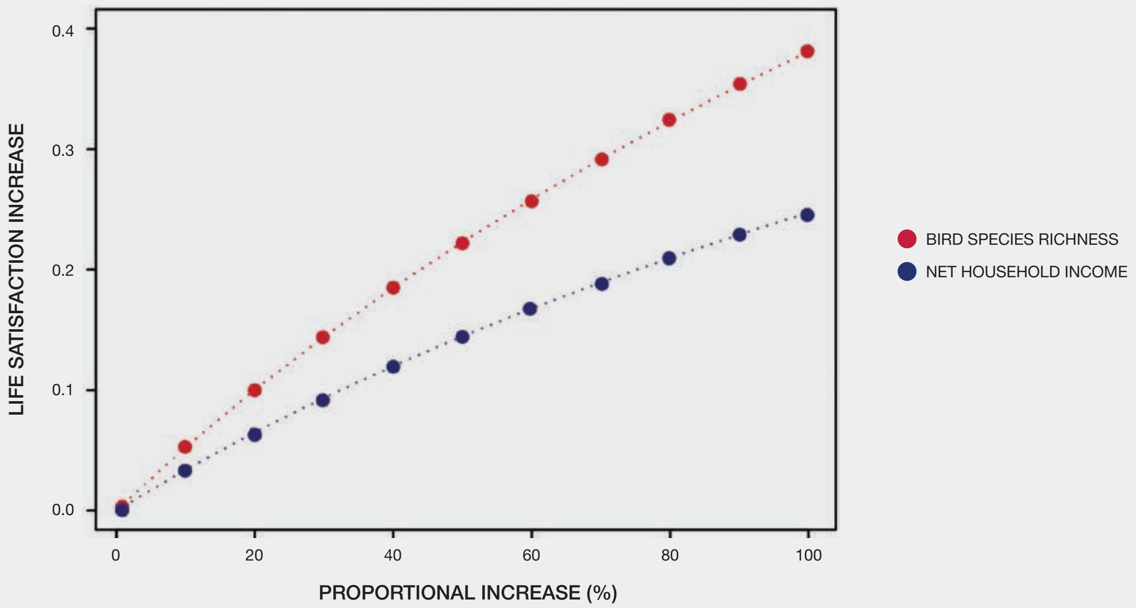


Figure 3 64 **Estimated life-satisfaction increase correlates to bird species richness and income across Europe.**

Estimates are based on the coefficients for log-transformed mean bird species richness and log-transformed net household income, both corrected for socio- and macro-economic factors. Source: (Methorst *et al.*, 2021) under license CC BY-NC-ND 4.0.

J. Wu, He, Chen, Lin, & Wang, 2020). Gentrification, while making cities more attractive to wealthy residents and attracting investment, has environmental justice implications, especially on urban green space access by lower class or income communities (Kronenberg *et al.*, 2020). Public space is also increasingly being 'corporatized', where public space maintenance is sponsored by private interests, and the urban green space is redesigned and highly controlled to meet the needs of the 'owner' rather than the general public (S. Schmidt, 2004). Research in North American cities on parks, urban forests and tree canopy cover found race, ethnicity and income disparities in tree distribution with non-white communities living in areas of lower tree density and lower quantity and quality of parks (Heynen, Perkins, & Roy, 2006; Rigolon, Browning, & Jennings, 2018). The disproportionate access to and distribution of urban green spaces creates inequitable health benefits derived from exposure to nature, with lower income and minority communities in cities more likely to live in "riskscapes" – environments that increase the vulnerability of these communities to pollutants and hazards (Jennings, Johnson Gaither, & Gragg, 2012).

"Green prescriptions" such as "forest bathing" are increasing as they improve human health, but there are also win-win opportunities for people and ecosystems through "reciprocal restoration". Pilot initiatives with urban youth working in habitat restoration programs have shown greater anti-inflammatory capacity, cardiovascular fitness, resistance to endoparasites, resistance to infectious diseases, reduced sensitivity to allergens, reduced frequency of nervous and musculoskeletal disorders and a wide range of positive effects on mental health (Nabhan, Orlando, Smith Monti, & Aronson, 2020). Concurrently, habitats are restored including vegetation cover and soil microbial content (Nabhan *et al.*, 2020).

The hypothesized mechanisms for the documented improvements in mental and physical health include the Microbiome Rewilding Hypothesis where restoring soil microbial diversity enhances human gut microbiome health and boosts immune functioning, and the Psycho-Evolutionary Restoration Hypothesis where humans exposed to forested systems exposes them to phytoncides that may reduce depression and lower cortisol levels (Nabhan *et al.*, 2020). In a critical review of the effects of environmental diversity on human health, Sandifer *et al.*, (2015) found the only unambiguous causal relationship was the maintenance of a healthy immune system and reduction of inflammatory diseases through exposure to environmental microbial diversity. There is also a limit in research on the sustained, long-term effects of nature-based therapies (Rajoo *et al.*, 2020). However, despite the limited information about the causal nature underlying the benefits of nature and biodiversity to human health, protecting and restoring a diversity of natural habitats seems crucial for maintaining human health in a developing world (Sandifer *et al.*, 2015). Indeed, the improvement of human health is a powerful tool to leverage support from multiple stakeholders to enhance social-ecological health in a variety of ways.

3.3.5.2.3 Recreation

Wildlife watching is an activity that involves the watching of wild species (animals and plants). Watching wild species is essentially an observational activity, although in some cases it can involve interactions with the animals being watched, such as touching or feeding them (UNEP/CMS, 2006). These recreational activities include nature-based tourism, hiking and nature walks, photography and cinematography, game watching and safaris, and snorkeling and scuba diving. The use of wild species for recreation is primarily for enjoyment but may also provide relaxation, restoration, physical exercise (see section 3.3.5.2.2. Medicine and hygiene), and educational experiences (see section 3.3.5.2.4. Education and learning).

The scope of this section is limited to the non-extractive practices of wild species for recreation where the impact of the use can be measured or assessed. The text below is based on a literature review (Web of Science) using the following strings of terms ("non-extractive use*" OR "non-consumptive" OR "cultural ecosystem service*" OR "non-material contribution*" OR "touris*" OR "community based tourism*" OR "ecotourism" OR "eco-tourism" OR "sustainable tourism" OR "recreational" OR "nature-based tourism" OR "wildlife watching" OR "wildlife viewing") AND (sustainab* OR trend*), generating 16117 hits. Articles that were recommended by citation databases were also considered, as well as collected from personal libraries and recommendations from experts. After a title and abstract read, 82 papers were selected for a full text read. After a full text read of these papers, 27 were deemed relevant and assessed for the literature review. Relevance was determined by mention of the status (current), trend (historical) or impacts of use of a wild species (or taxa) in at least one dimension of sustainability (social, economic, or ecological). These data form the basis of the text below.

The literature covered fairly equally (4-6 papers each): vegetation (trees and shrubs); terrestrial mammals; birds (terrestrial and marine); marine mammals; fish, rays and sharks; and arthropods (marine and terrestrial). The temporal scale of the research articles was 10 short term (<1 year), 1 medium term (1-10 years), and 9 long term (>10 years) studies. The review included articles from every IPBES region.

Most of the information in the text below relates to wildlife watching tourism, as 74% of relevant articles focused on tourism specifically. Wildlife watching does occur around people's homes (Wilkinson, Waitt, & Gibbs, 2014; Zarazúa-Carbajal *et al.*, 2020), but this is less well documented than wildlife watching tourism. Wildlife watching tourism overlaps with various types of tourism, such as tours focused on seeing a specific kind of wild taxa **(Table 3.19)** and tourism where wildlife watching is an added advantage but not the main focus of the activity (e.g., adventure and sports tourism) (UNEP/CMS, 2006). Similarly, a specific type of nature-based tourism is eco-tourism, where the tourism activity aims to contribute to the conservation of natural and cultural heritage through the involvement of indigenous peoples and local communities (UNEP/CMS, 2006). Eco-tourism has relatively low numbers of tourism and is suited to small groups and independent tourists (UNEP/CMS, 2006).

Social aspects

Enjoyment of nature for tourism and recreation is recognized as the most prominent cultural nature's contributions to people (Balmford *et al.*, 2015). Over the last half a century the demand for nature-based tourism experiences has been on the rise, with the ever-increasing breadth and depth of its global penetration, integrating more and more natural areas into commercial processes (Balmford *et al.*, 2009, 2015; Elmahdy, Haukeland, & Fredman, 2017; Hall, Harrison, Weaver, & Wall, 2013; Mowforth & Munt, 2015; D. Scott & Gössling, 2015). For example, according to rough estimations, world terrestrial protected nature areas currently receive approximately 8 billion visits per year, of

Table 3 19 **Examples of species- and taxa-based wildlife watching across the globe.**
Source: (UNEP/CMS, 2006) under license CC-BY.

Species being watched	Tourism Activity	Location example
Butterflies	Butterfly watching	Monarch butterflies in Mexico, United States of America and Canada
Glow worms	Glow worm watching	Springbrook National Park, Australia
Crabs	Red crab migration	Christmas Island, Indian Ocean
Corals and fish	Snorkel/scuba diving	Bunaken, Indonesia; Sian Ka'an, Mexico; Soufriere Marine Management Area, St. Lucia; Bonaire, Caribbean; Red Sea, Egypt
Sharks	Snorkel with whale sharks	Seychelles; Ningaloo Reef, Australia
Sharks	Underwater watching/feeding of sharks	Dyer Island, South Africa
Stingrays	Feeding and close interaction with stingrays	Cayman Islands; Maldives; Australia
Komodo dragons	Watching Komodo dragons	Komodo Island, Indonesia
Snakes	Watching pythons	Bharatpur, India
Crocodiles	Watching crocodiles	Black River, Jamaica; Kakadu Park, Australia
Turtles	Watching turtles	Projeto TAMAR-IBAMA, Brazil; Akumal, Yucatán Pennisula, Mexico; Cape Verde; Maputaland, South Africa; Sri Lanka; Indonesia
Birds	Independent or organized visits to reserves for bird-watching	Bempton Cliffs, United Kingdom; Keoladeo, India; Pantanal, Brazil
Albatrosses	Independent or coach tours to see breeding colonies	Taiaroa Head, New Zealand
Cranes	Watching cranes	Müritz National Park, Germany; Platte River, United States of America
Penguins	Watching penguins and penguin colonies	Antarctica; Peninsula Valdés, Argentina; Phillip Island, Australia
Large African mammals	Vehicle safaris to see large concentrations of mammals	Serengeti National Park, Tanzania; Masai Mara, Kenya
Tigers	Tiger watching from hides or elephant back	Chitwan National Park & Bardia National Park, Nepal
Gorillas	Mountain trek and camping to observe habituated gorillas	Bwindi National Park, Uganda; Virunga National Park, Democratic Republic of Congo; Volcanoes National Park, Rwanda
Orangutans	Watching orangutans	Sepilok Orangutan Centre & Danum Valley, Sabah Semenggok Wildlife Centre, Sarawak, Borneo
Polar bears	Watching polar bears	Churchill, Canada
Bats	Watching bats	Texas, United States of America
Dolphins	Watching dolphins	Red Sea, Egypt; Mon Repos, Australia
Whales	Watching whales	Peninsula Valdés, Argentina; Kaikoura, New Zealand; El Vizcaino, Baja California, Mexico; New England, United States of America; Plettenberg Bay, South Africa; Canary Islands

which 80% are in Europe and North America (Balmford *et al.*, 2015). In general, nature-based tourism and recreation are affected by the following global drivers of change, i.e., megatrends: *social trends* (population growth, urbanization, changes in household composition, aging population, health and well-being, changing work patterns, gender equality, values and lifestyle); *technological* (transportation, high-tech equipment, information and communication technologies); *economic* trends (economic growth; sharing economy; fuel costs); *environmental* (climate change; land use and landscape change); *political* (political turbulence; changes in border regulations; health risks; geopolitics) (Elmahdy *et al.*, 2017). The complexities of these drivers are discussed in Chapter 4 of this assessment. For the purposes of Chapter 3, it is important to point out that a combination of these interconnected global trends is and will be significantly affecting demand for nature-based tourist experiences and the way people engage with nature.

There is concern that the aforementioned global trends contribute to increasing disconnectedness of large masses of populations from natural phenomena and processes in their daily life, which generates interest to experience nature as a leisure activity in an organized, often commercialized setting (Buckley, 2000; Buckley, Gretzel, Scott, Weaver, & Becken, 2015; Curtin, 2005; Dwyer, 2003; Elmahdy *et al.*, 2017). It has been observed that nature-based tourism and recreation are increasingly characterized by the importance of experiences, achievement, adventure and well-planned activities rather than simple leisure and social interaction. Many studies indicate that tourism and recreation in nature are becoming more specialized, diversified, motorized, sportified and adventurized (Öhman, Öhman, & Sandell, 2016; Sandell, Arnegård, & Backman, 2011). In this context nature is transformed into a setting, a scenic backdrop for tourist experiences (Fossgard & Fredman, 2019; Margaryan, 2017). This also affects tourists' expectations regarding the availability of tourism-related services in nature. There is a growing demand for 'wild', 'unspoiled', 'pristine' nature in combination with high levels of comfort, accessibility and high-quality experiences (Elmahdy *et al.*, 2017; Fredman, Wall-Reinius, & Grundén, 2012). These pristine landscapes are advertised for tourism in brochures with backgrounds of teeming game, but absent of the human communities that live alongside wild species (Montgomery, Borona, Kasozi, Mudumba, & Ogada, 2020).

This marketing perpetuates that indigenous peoples and local communities are separate from the social-ecological system, constitute a threat to wild species conservation, and drives the alienation and displacement of indigenous peoples and local communities, often with indigenous peoples and local communities on the boundaries of conservation areas receiving few benefits from tourism activities taking place (Montgomery *et al.*, 2020; Saarinen, Moswete, Atlhopheng, & Hambira, 2020). Indigenous peoples and local communities also suffer from the negative aspects of tourism, for example disease and predation adjacent to protected areas (Swemmer, Mmethi, & Twine, 2017), or tourist-related disturbance of their activities (e.g., snow mobile recreation in the vicinity of Saami reindeer herders in Lapland (Kluwe & Krumpe, 2003), or rock climbers disturbing Native American rituals on Devils Tower/Mato Tipila in Wyoming (Taylor and Geffen 2004). There may also be a conflict in values between recreational and non-recreational users, especially around expected behavior in sacred areas or around traditional hunting practices (Zeppel, 2010). Tourists can also cause degradation of culturally important sites through or illegal removal of cultural heritage items (INTOSAI WGEA, 2013). Tourism may change local identities and values, through commercialization of local culture and standardization to meet tourists' expectations (INTOSAI WGEA, 2013).

As the demand for wild species-related experiences is on the rise, wild species-related content on social media and wild species documentaries have become more popular than ever. For example, the British Broadcasting Corporation (BBC) Planet Earth I and II have been among the most watched documentaries worldwide (Jackson, 2016). The growth of media attention and circulation of wild species-related content in the social media further stimulates demand to experience wild species in real life, as well as photograph and share 'selfies' with wild species. Between 2014 and 2017 there has been a documented increase of nearly 300% in the quantity of wild species selfies shared on the Instagram platform (World Animal Protection, 2017). Of these, over 40% could be classified as inappropriate wild species selfies – featuring handling, hugging, touching, feeding or other potentially detrimental interactions between humans and wild species (World Animal Protection, 2017).

Tourism marketing and social media sharing influences the demand for extremely close interactions with wild species (Dou & Day, 2020). However, research has shown the dichotomy of tourists' desires for intimate encounters with wild species and recognition of the detrimental effects on animal welfare as a result of these interactions (Dou & Day, 2020). Environmental education and increased awareness of wildlife watching sustainability can and does play a role in changing tourist behavior, such as that demonstrated in dolphin-watching tours where tourists were willing to trade-off close interactions for the purposes of dolphin welfare (Dou & Day, 2020). Research on Mozambiquan tourists showed low awareness of cetacean-based regulations, but the tourists were supportive of well-regulated activities, therefore educated tourists could increase operator compliance with regulations (Rocha *et al.*, 2020). In their review on wildlife watching sustainability, Dou and Day (2020) caution that environmental awareness does not occur automatically from increased exposure to wild species, but rather from focused environmental education with targeted, actionable messages on biodiversity conservation.

Wildlife watching has emerged as a widespread and lucrative tourist activity and its popularity is growing rapidly (de Lima & Green, 2017; Dybsand, 2020; Hassan & Sharma, 2017; Karanth *et al.*, 2017; Mowforth & Munt, 2015; World Animal Protection, 2017). International tourism has grown year after year for the last decade, driven in part by nature-based tourism (including extractive tourism activities) (UNWTO, 2019). Between 1990 and 2000, average annual international tourism growth was 4.4%, but wild species-rich countries like Madagascar, Brazil, Cuba and South Africa all experienced averages between 10-20% annually and Vietnam and Laos between 24-36% (UNEP/CMS, 2006). Regional share of wildlife watching tourism relative to overall tourism varies globally, from 36.3% in Africa to 1.6% in Europe (WTTC, 2019a). Domestic tourism is estimated to be ten times the scale, but the numbers are uncertain. Similarly, the proportion of non-extractive nature-based tourism in recreation and leisure tourism is difficult to unpack as increasingly tourism trends have seen a blend in various types of tourism, such as family holidays that involve urban, adventure tours and wildlife watching (UNEP/CMS, 2006), or visits that combine trophy hunting and wildlife watching. But comparisons of protected area visitation rates mirror overall tourism rates in low-income countries (Balmford *et al.*, 2009).

A global study estimates that protected areas receive 8 billion visits per annum, generating 600 billion United States dollars (Balmford *et al.*, 2015). Revenue generated from tourism in protected areas far exceeds the cost of managing these areas (Balmford *et al.*, 2015; WTTC, 2019a). Surveyed governments and tour operators overwhelming rank nature, national parks and wild species as their largest assets for tourism, a practice that is labor intensive and employs local communities, especially in remote areas where developing regions do not have many other employment options (UNWTO, 2015). Nature-based tourism has been increasing over the last decade as a result of increased demand (increased knowledge of wild species from media and the internet) and shrinking supply (reduced habitats and wild species scarcity) (The World Bank, 2018). This is apparent in visitation data for the iconic nature-based tourism destination, the Galapagos Islands, which has recorded an increasing trend in visitors from 1989 (<50,000 visitors) to 2019 (about 271,000 visitors) (Díaz-Sánchez & Obaco, 2020).

Recreational use of wild species also generates significant revenue, particularly through nature-based tourism. Wildlife watching contributed 120.1 billion United States dollars in 2018 (343.6 billion United States dollars with multiplier effects) to global gross domestic product, five times the estimated value of the illegal wild species trade (WTTC, 2019a). Wildlife watching also sustained 21.8 million jobs (WTTC, 2019a). The global monetary potential value of whale watching was estimated at over 2.5 billion United States dollars in 2009 and supporting 19,000 jobs (Cisneros-Montemayor *et al.*, 2010). Shark and ray watching generated over 314 million United States dollars per annum, directly supporting 10,000 jobs and is expected to double by 2033, generating over 780 million United States dollars globally (Cisneros-Montemayor, Barnes-Mauthe, Al-Abdulrazzak, Navarro-Holm, & Sumaila, 2013). In contrast the value of shark fisheries was estimated at 630 million United States dollars and has been on the decline over the last decade (Cisneros-Montemayor *et al.*, 2013). The expected revenue from entrance tickets to the Galapagos Islands in 2020 was about 18 million United States dollars, although significant losses have been predicted as a result of the COVID-19 pandemic (Díaz-Sánchez & Obaco, 2020). This revenue is mainly allocated to Galapagos Island conservation programs (Díaz-Sánchez & Obaco, 2020). International tourism arrivals in Africa, in large part for wild species tourism (including extractive recreational tourism), in 2013 were 56 million people, generating 34.2 billion United States dollars, and 134 million tourists are expected in 2030 (World Tourism Organization, 2014). During 2000 in East Africa alone, 1 billion United States dollars was generated from foreign tourist arrivals (UNEP/CMS, 2006). In the United States of America, wildlife watching engaged 86 million people in the vicinity of their homes, and 81.1 million people travelled away from home to view wild species, generating 75.9 billion United States dollars (United States of America Department of the Interior, United States of America Fish and Wildlife Service, United States of America Department of Commerce, & United States of America Census Bureau, 2018). Recreation represents over 75% of the value of the United States of America national forests, higher than the value of timber extracted (Groom *et al.*, 2006).

Although tourism revenue is significant at the national level, for economic benefits to alleviate poverty, the World Tourism Organization (UNWTO) found local level employment, infrastructure benefits, supply of goods and services and support by the tourism enterprises, as well as other pro-poor approaches were important (UNEP/CMS, 2006). If local communities and suppliers are able to meet the standard needed to cater to international tourists, considerable benefits can accrue to the local economies (Twining-Ward, Li, Bhammar, & Wright, 2018; UNEP/CMS, 2006). However, if supplies and expertise are sourced on imports, then 50% or more of the tourism revenue “leaks” from the local and national economies (UNEP/CMS, 2006).

Wild species which have the biggest importance for the tourism and recreation practices are those which attract interest from the widest spectrum of tourists, i.e., the ‘flagship’ species – most often the megafauna, ‘charismatic’ mammals and birds, the ‘cute and cuddly’, dangerous predators, threatened species, and species that are believed to display intelligence (Aguilera-Alcalá, Morales-

Reyes, Martín-López, Moleón, & Sánchez-Zapata, 2020; Carr & Broom, 2018; Devillers & Beudels-Jamar, 2008; Higginbottom, 2004; Newsome, Moore, & Dowling, 2012). The growing awareness of biodiversity loss has created a demand to see places and wild species that might disappear, including "endangered experiences", highly exclusive tourism packages offering unique opportunities (WTTC, 2019b). For example, in Eurasia and Africa, national parks that hold large mammals have much higher visitation rates than those which do not (Devillers & Beudels-Jamar, 2008).

Difference in preferences for wild species has its roots in a range of evolutionary as well as cultural predispositions (Jacobs, 2009). While some countries have a long history of wildlife watching tourism (e.g., in the East and South Africa), recent rapid growth of this business has been observed in many new destinations, for example in Southeast Asia and the Amazon (Karanth *et al.*, 2017; World Animal Protection, 2017). Overall, natural areas of high value for wildlife watching tourism tend to be characterized by (i) abundance of large animals, (ii) presence of charismatic species, and (iii) high biodiversity (Higginbottom, 2004; Newsome *et al.*, 2012). It is expected that presence of tourism in such areas will only be increasing, so special attention needs to be paid to aspects of sustainability in these processes.

Wildlife watching activities and tourism accrue considerable funds for conservation projects, as well as raising public awareness of the need for conservation. A case in point is Projeto Tamar which, through working with local communities and fishers, successfully promoted turtle conservation along the Brazilian coastline, improving turtle hatching success through protecting hatchery sites and establishing alternative employment and income opportunities based on tourism and turtle protection (UNEP/CMS, 2006). Similarly, a public-private partnership in a heavily poached region resulted in increased revenue for local communities and provided alternative revenue, to such a degree that wild species are again abundant in Majete Wildlife Reserve, Malawi (Twining-Ward *et al.*, 2018). Conservation of one of the last remaining nesting sites of Little Penguins *(Eudyptula minor)* on Australia's Phillip Island Nature Park, on Bunurong Aboriginal Land, is funded by an inclusive, collaborative business plan for tourism (UNEP/CMS, 2006). The business plan is revised every five years with the community and stakeholders and the Bunurong community representatives are involved in education programs and project supervision of a high-quality, high-volume tourist enterprise (UNEP/CMS, 2006).

Ecological aspects

Wildlife watching can have unintended consequences for wild species in three ways: changes to species behavior, changes to physiology, or damage to habitats (UNEP/CMS, 2006).

Behavioral changes to wild species include changes to feeding or resting time, expending energy to try and move away from the disturbance, altering interactions between different species (UNEP/CMS, 2006), aggressive behavior, increased stress, or alternatively a reduction in fear towards humans, and dependency on non-natural and supplemental food sources especially at feeding sites (Dou & Day, 2020), or preventing optimal spatial distribution relative to resources (Blanc, Guillemain, Mouronval, Desmonts, & Fritz, 2006). Short-term changes in animal behavior as a result of human-wild species interactions in tourism contexts are easier to detect and well-studied, but long-term changes are under researched (Dou & Day, 2020). Similarly, tourism effects on wild species individuals are more detectable and better documented than the repercussions of these individual effects at the population level (Blanc *et al.*, 2006).

The evasive nature of wild species together with tourists' expectations for a close contact with wild species creates a strong incentive for tourist destination managers to minimize sighting uncertainty and decrease the watching distance through invasive practices ranging from baiting, attracting, and habituating, to capturing animals (Dybsand, 2020; Knight, 2009; Margaryan & Wall-Reinius, 2017), and driving off-road (Nortje, 2014). Commercial wildlife watching activities rely on wild species being made viewable, which is often achieved through highly unsustainable and unethical practices (Dybsand, 2020; Knight, 2009; World Animal Protection, 2017). For example, high tourist volumes in the Serengeti have created serious disturbance for wild species and the large area of the park makes it challenging to enforce responsible game watching behavior (UNEP/CMS, 2006). In one case, the cubs of a cheetah were scared away by 15 vehicles and assumed to be predated on by lions as they were never seen again (UNEP/CMS, 2006).

Snorkeling and diving may also disturb the aquatic habitat and influence species behavior (Teresa, Romero, Casatti, & Sabino, 2011). The practice of fish feeding during diving may affect fish communities (Ilarri, Souza, Medeiros, Grempel, & Rosa, 2008). A long-term, intensive study of the detrimental impacts on wild species from even well-managed, low level, commercial watching and controlled feeding of bottle-nosed dolphins at Monkey Mia, Western Australia documented long-term dolphin responses to human-wild species interaction. Over decades of monitoring, dolphin abundance (immigration and mortality) and fecundity declined at the tourism sites but not the control sites (Higham & Bejder, 2008). Highly responsive management interventions were implemented based on the research recommendations (Higham & Bejder, 2008) and impacts were reduced after regulations limiting the duration of feeding events (Foroughirad & Mann, 2013). However, findings of this nature are of great concern for the unknown long-term sustainability at other, especially high-intensity and/or low management tourism sites, for cetaceans and other wild species (Dou & Day, 2020; Higham & Bejder, 2008).

A similar activity has been conducted in the Negro River, in the Brazilian Amazon, directed to feeding the freshwater pink (or red) dolphin *(Inia geoffrensis)*, but the potential effects of this activity on dolphin's behavior are not well known, but potentially increase dolphin aggression and may be harmful to both the dolphins and tourists (Pinto de Sá Alves, Andriolo, Orams, & de Freitas Azevedo, 2013). White sharks *(Carcharodon carcharias)* watching activities elicited curiosity and aggressive behaviors associated with feeding, leading the authors to advise against intentional feeding to avoid human-shark-cage associated incidents and the conditioning of sharks to boats (Becerril-García, Hoyos-Padilla, Micarelli, Galván-Magaña, & Sperone, 2019).

Even relatively innocuous recreational activities can have an impact on animal behavior. Research using camera traps to assess the prevalence of human recreational activities in association with terrestrial mammal occurrence in a Canadian protected area showed avoidance of mountain biking by moose (*Alces* spp.) and grizzly bears *(Ursus arctos)*, although all recorded mammal species avoided humans on trails, especially mountain bikes and motorized vehicles (Naidoo & Burton, 2020). Even "silent activities" such as windsurfing may have impacts as they enable off-path access to otherwise "sanctuary" areas (Blanc *et al.*, 2006). But the presence of tourists and vehicles can be reduced through spatial or temporal zonation to provide sanctuary for wild species. The adverse impacts of high volumes of tourists and vehicles on wild species is managed in the Serengeti through strict park zonation, where certain areas are designated "No-Go" zones where no wildlife watching is allowed, "Intensive" and "Low Use" zones have designated tourism activities and "Wilderness" zones where no vehicles are allowed and low numbers of tourists do walking tours (UNEP/CMS, 2006).

Habituation (stress response decreases with repeated exposure to humans) or sensitization (stress response increases with repeated exposure to humans) varies across and within species, and with the type of stressor, the type of tourism, spatiotemporal aspects, life history traits and intraspecific characteristics (Geffroy, Samia, Bessa, & Blumstein, 2015). For example, African penguins *(Spheniscus demersus)* and Magellanic penguins *(Sphenicus magellanicus)* habituate to humans but yellow-eyed penguins *(Megadyptes antipodes)* sensitize to humans (Geffroy *et al.*, 2015). Thus, the impacts of repeated exposure to humans are extremely context-dependent and need to be assessed locally, as well as monitored over the long-term. This has important repercussions for wild species, as behavioral changes as a result of tourist-exposure may compromise their susceptibility to poaching or their risk of predation by other animals (Geffroy *et al.*, 2015).

Wild species' physiology may be affected by tourism activities even though their behavioral patterns have not altered (Dou & Day, 2020). Yellow-eyed penguins *(Megadyptes antipodes)* at unregulated tourism sites showed significantly higher stress-induced corticosterone concentrations, with lower breeding success and lower fledgling weights than penguins visited for monitoring purposes only (Ellenberg, Setiawan, Cree, Houston, & Seddon, 2007). The presence of roads and traffic can also increase animal stress levels (Lunde, Bech, Fyumagwa, Jackson, & Røskaft, 2016). A well-studied intensive tourism site at the Grand Cayman Islands where stingrays *(Hypanus americanus)* are visited and fed by recreational scuba divers since 1986 have shown haematological changes, increased parasite loads, high injury rates and open wounds from boat collisions, and major behavioral changes from being normally solitary to forming schools of 12-15 individuals, as well as switching to diurnal feeding at the dive sites (UNEP/CMS, 2006). Most of the stingrays' food now comes from divers and the reduced dietary diversity has compromised their disease resistance and immune response (UNEP/CMS, 2006). However, these kinds of examples of poor tolerance of tourist activities by species are species, habitat, tour operator and regulator specific. For example, a comparison between the effects of provisioning and viewing on the Cayman stingrays, which has been shown as detrimental, against the highly self-regulated and limited number of shark-feeding tour operators in Fiji suggests no effects on bull shark (Carcharhinus leucas) fitness and health (Healy, Hill, Barnett, & Chin, 2020).

The trend in using wild species as photo props for "selfies" as photographic souvenirs has driven an increase in captive and handling of wild species, like slow lorises (*Nycticebus* spp.) in Asia, which have their teeth clipped to reduce the risk of injury to tourists, and results in early death (Osterberg & Nekaris, 2015). A study of three-toed sloths *(Bradypus variegatus)* in Brazil and Peru found each sloth was held by on average five tourists, often by the claws and had their limbs stretched and manipulated (Carder *et al.*, 2018). Wild species handled for long durations have been shown to display increased behavioral and physiological stress responses, leading to injury, stress and death (Baird *et al.*, 2016).

Tourists and other recreational users of nature, especially in high volumes, can damage the environment and species habitats. Trampling vegetation and the creation of informal trails both damages the environment and reduces the appeal and restorative impact on human health and well-being of these areas to other recreational users (Taff, Benfield, Miller, D'Antonio, & Schwartz, 2019). There is evidence that scuba diving, even without fish feeding, may cause unintentional damage to aquatic organisms, such as corals and algae, which may be hit by divers (Di Franco, Milazzo, Baiata, Tomasello, & Chemello, 2009). The sunscreen from divers and swimmers has been associated with bleaching of coral reefs (Danovaro *et al.*, 2008; Downs *et al.*, 2014) and Hawaii has banned the use of sunscreens containing oxybenzone or octinoxate from the 1st of January 2021 with similar bans predicted to follow in other coral-reef containing countries (Raffa, Pergolizzi Jr, Taylor Jr, Kitzen, & Group, 2019). Even a single vehicle driving on sandy beaches has been estimated to crush up to 0.75% of the intertidal population (Schlacher, Thompson, & Price, 2007) and beach camping zones show a 20.2% reduction in dune vegetation (Thompson & Schlacher, 2008).

A review of winter recreational activities in Alpine areas found ski resorts and associated infrastructure have negative impacts across all studied taxa, independent of geographic region or ski modification (Sato, Wood, & Lindenmayer, 2013). This is concerning as the area affected in Europe by ski-runs is large and increasing, currently spanning about 4000 km across Italy, Switzerland and Austria, although there is a suggestion that environmentally-friendly ski-run design could mitigate many of these impacts (Rolando, Caprio, Rinaldi, & Ellena, 2006). Tourism facilities (e.g., lodges, ablutions) and impacts from waste, as well as high water usage are concerns in the nature-based tourism industry (UNEP/CMS, 2006). Despite initiatives to foster sustainable travel behaviors (e.g., carbon offsetting for unavoidable travel emissions) and attempts to improve the eco-efficiencies of tourism industries, tourism carbon emissions have increased at 3.3% annually (Sun, Lin, & Higham, 2020), driven by increased travel frequency, long-haul flights and shorter stays per trip (Sun *et al.*, 2020).

Altering resource availability to wild species to increase watching potential can have unintended consequences on the surrounding ecosystem. The provision of artificial water points in Kruger National Park, South Africa, although intended to maintain herbivore numbers during droughts expanded the range of water-dependent species (e.g., zebra and wildebeest), and in association their predators (e.g., lions) to the detriment of less common species (e.g., roan antelope) (Harrington *et al.*, 1999). The widespread availability of surface water has also been implicated in the reduction of vegetation structure by homogenizing elephant impacts across the landscape (Gaylard, Owen-Smith, & Redfern, 2003).

These unintended effects to facilitate watching wild species demonstrate the complexity of tourism impacts on populations and ecosystems. As these impacts are species and context specific, there is much to be discovered about the potential of tourism impacts. Even under the best code of conduct, there might still be detrimental, often cryptic, effects on animal reproduction and long-term survival (Carr & Broom, 2018; Szott, Pretorius, Ganswindt, & Koyama, 2020; Tyagi *et al.*, 2019). A review of tourist impacts on wild species cautions that although the literature overwhelmingly reports on negative impacts, the findings are strongly dependent on the methods used and many findings (especially behavioral responses) could be interpreted as short-term coping strategies that do not necessarily have long-term repercussions (Bateman & Fleming, 2017). Considerable variation exists between and within species and locations, in tourism operator methods and regulations. Therefore, more serious and coordinated global multi-stakeholder efforts to regulate this practice, involving tourism businesses, local communities, science, governmental and non-governmental organizations, are needed.

Considerations for sustainable recreational use

Based on the current trends one can expect further growth in demand for wildlife watching experiences and, consequently, an increasing number of wild species integrated into tourism operations. Particularly vulnerable in this perspective are the megafauna and 'charismatic' wild species, which, however, also receive the most media attention and conservation support (Carr & Broom, 2018). Megafauna are the best studied taxa of animals, whereas there is a lack of research on the impacts of tourism on the lesser fauna, e.g., ground-dwelling mammals, small reptiles, insects, etc. (Wolf, Croft, & Green, 2019). The interlinkages between tourism, representations of wild species on social media, conservation and sustainability have acquired great importance and need further research and policy attention. Likewise, the role of environmental education in changing tourist attitudes and behavior requires further research attention (Dou & Day, 2020).

Specific attention needs to be paid to the emergence of the so-called tourist-driven destinations, which appear spontaneously based on a spike in media popularity and uncontrolled tourist demand, rather than coordinated marketing efforts of the local tourism actors. In addition, the expansion of tourism into remote, 'pristine' areas needs to be managed and monitored to avoid detrimental impacts to sensitive and vulnerable species (UNEP/CMS, 2006). As tourists prefer areas that are deemed 'pristine' (i.e., more ecologically and aesthetically sound), there are opportunities to increase recreational tourism by restoring ecosystems. For example, work in RAMSAR (the Convention of wetlands of international importance) listed wetlands in India suggest that annual recreational visits could increase by 13% if the water quality could be improved to maintain wild species and fisheries diversity and abundance (Sinclair, Ghermandi, Moses, & Joseph, 2019). Researchers have also highlighted the need for studies that integrate the ecological and social aspects of human-wild species interactions to inform the sustainable development of the tourism industry, local communities and wild species conservation (Dou & Day, 2020). Financial resources and operational experience are sorely needed at the human-wild species interaction interface, with many wild species populations attractive to tourists in countries least able to afford the research, management and monitoring needed in these sites (Dou & Day, 2020). Finding ways to mobilize the power of new communication technologies and channel them towards sustainable tourism practices will be crucial in achieving more sustainable wildlife watching operations.

Sustainable nature-based tourism needs to make a positive impact to the natural and social setting that tourism takes place in, and should generate benefit for the host communities and indigenous peoples and local communities in a manner that does not compromise the future human well-being needs of indigenous peoples and local communities or the ecosystems (UNEP/CMS, 2006). Well-managed wildlife watching is a significant boon to community development and revenue, as well as an important source of funding for wild species conservation (UNEP/CMS, 2006). However, tourism is only sustainable where the habitats and species being used recreationally are sufficiently resilient to the impacts related to the use, where tourism and the associated development is kept within manageable limits, where the tourism experience attracts a long-term and viable tourism economy, and where local communities and the local economy benefit from the activity (UNEP/CMS, 2006). The direct benefits range from increased income and employment through education and access to many new facilities, up to perception of pride and recognition. Although the direct economic benefits are most important to local residents (Akis, Peristianis, & Warner, 1996), the indirect benefits such as improved public infrastructure, education, public safety and healthcare facilities may reach even wider groups of people (e.g., Afenyo & Amuquandoh, 2014) and gain the support for tourism among those who do not benefit directly from the activity. Addressing the above points to plan and manage sustainable nature-based tourism requires stakeholder engagement in a process that helps identify diverse interests, provides expertise, and facilitates local commitment to managing tourism ventures and impacts (UNEP/CMS, 2006).

An exemplary case study of stakeholder engagement in wild species tourism is that of Bunaken national marine park, Indonesia. Bunaken national marine park pioneered a co-management approach that is being modelled by other protected areas (UNEP/CMS, 2006). Bunaken national marine park is a popular dive site for international tourists, as well as the home of 30,000 people whose livelihoods depend on fishing. Park management is overseen by a multistakeholder advisory board, including governmental, non-governmental organizations, representations of the villages within the park, park authorities, Tourism and Fisheries Departments, the local universities and the private tourism sector (UNEP/CMS, 2006). Local community elders advised on the location of the marine sanctuaries and no-take zones, the local community is involved in reef restoration efforts, and advised where to place marine sanctuaries which replenish both diving and fishing sites. Proceeds from park fees are managed by the multistakeholder board and are used for conservation and development programs, village development schemes, plastic and waste disposal, environmental education of villagers, rehabilitation and restoration projects, and law enforcement and patrols for destructive fishing and tourism practices (UNEP/CMS, 2006). Stakeholder needs have evolved as the social and environmental landscape has changed, and management has recognized the need to be adaptive in this regard. In Bunaken, the growing popularity for tourism is placing additional stress on the reefs and the large numbers of dive operators are not all members of the stakeholder association. They are considering a mandatory license system rather than voluntary compliance to manage the number of divers, dive operators and boats (UNEP/CMS, 2006). In another example, an unexpected repercussion of a successful public-private partnership in a heavily poached reserve has resulted in a tourism and revenue boom in Malawi's Majete wildlife reserve, but the now abundant wild species are affecting local communities' resources, increasing human-wildlife conflict (Twining-Ward *et al.*, 2018).

Stakeholders differing needs and perspectives need to be negotiated, as power imbalances between stakeholders can undercut effective collective management actions (Meza-Arce *et al.*, 2020). Recreational use may also be at odds with the extractive natural resource use needs of the local communities. This highlights the need to manage both physical and cultural conflicts between recreational users and indigenous peoples and local communities, through temporal or spatial zoning as well as by addressing the disparate cultural and social values of the respective stakeholders sensitively (Zeppel, 2010).

Significant opportunities exist for tourism revenue to support indigenous peoples and local communities that are already involved in conservation practices through local and traditional practices. The *Entim e Naimina Enkiyio* (Forest of the Lost Child) is one of few ungazetted forests in Kenya supporting abundant wild species, including threatened and highly endemic species (Tebtebba Foundation, 2010). This site is estimated to have tourism potential of up to 40,000 United States dollars per annum, notwithstanding the other benefits supporting conservation would ensure, such as catchment protection, wild algae, fungi and plants, grazing, and spiritual and cultural value (Tebtebba Foundation, 2010). The communities conserving biodiversity, as well as managing the natural resources for their subsistence, should be supported and strengthened where appropriate.

However, the benefits of tourism should not be overstated and require careful consideration of what is realistic (UNEP/CMS, 2006). For example, in a survey of World Bank Global Environnement Facility projects, most projects had positioned tourism as key to sustainable resource management and wild species conservation, but only 8% of the projects analyzed the tourism-derived income potential (UNEP/CMS, 2006). A key finding of this World Bank survey was that although tourism did generate revenue, it could not be solely relied on and was not even the most important source of funding (UNEP/CMS, 2006).

An economic model of the impacts of increased tourism revenue in the Philippines demonstrated that although economic benefits are accrued to all local households in the short-term, over the long-term increased demand for natural resources driven by the tourism industry eroded local household incomes, particularly for households directly involved in the natural resources economy (Gilliland, Sanchirico, & Taylor, 2020). Similarly, tourism in Latin America was associated with increased agricultural expansion and deforestation to service tourist consumption (Gunter & Ceddia, 2020). Providing indigenous peoples and local communities with title deeds and land rights seemed to mitigate this effect, although the research authors caution this effect should not be overstated (Gunter & Ceddia, 2020).

The Monarch Butterfly Forest Project is often lauded as a win-win-win success for tourism-livelihoods-conservation. Established in Mexico at forest locations where monarch butterflies *(Danaus plexippus)* congregate in winter, the project promoted recreation centers, established butterfly visitor centers and implemented tourism management at butterfly sanctuaries (UNEP/CMS, 2006). The project focuses on livelihood solutions for a region characterized by high unemployment, and has provided tourism training for local people, is involved in reforestation of areas critical to the butterfly habitat, and spans to managing logging impacts in Canada and the United States of America which threaten monarch summer habitat (UNEP/CMS, 2006). Without detracting from the immense strides the Monarch Butterfly Project has made in livelihoods and conservation, in some areas there is evidence of local residents returning to extractive activities as the project failed to yield the expected employment opportunities (Barkin, 2003). Although the rate of logging within the core areas of the Monarch Butterfly Biosphere Reserve have declined, logging is still present (Flores-Martínez *et al.*, 2019; Vidal, López-García, & Rendón-Salinas, 2014), particularly small-scale logging (López-García & Navarro-Cerrillo, 2020). The Monarch Biosphere Reserve zonation policies restricting community use of natural resources and the subsequent compensation for lost legal logging permits through payment for ecosystem services has had unintended consequences through provoking social conflict, often armed, in some areas (Gonzalez-Duarte, 2021). Indeed, the local communities who were ancestral inhabitants of what is now core biosphere areas do not share the biosphere reserve paradigm of a binary use/non-use landscape, but instead view the relationship with the forest ecosystem as a continuum of co-inhabitation and Gonzalez-Duarte (2021) suggests the enforced split in ancestral ecological practices has supported a fractured social compact, fostered illicit extractive activities, undermined community forest management, encouraged organized crime and has created a disregard for core areas where monarch butterflies do not overwinter. For an overview of the challenges facing Mexico's biosphere reserves, see Sada (2019)). These challenges are by no means unique to the Monarch Butterfly Biosphere Reserve and occur in many of the global biospheres reserves where integration of core conservation areas is not adequately incorporated into the multi-use, social-ecological system that the core reserves and local communities exist within (Coetzer, Witkowski, & Erasmus, 2014).

Often local livelihoods are believed to be in conflicting relation with conservation and therefore they are highly restricted by the rules and regulation that impede local economic development (Stone, 2015; West, Igoe, & Brockington, 2006). Cases of prohibition of traditional activities that involve unsustainable use of natural resources in favor of conservation were reported in Tanzania (Charnley, 2005), Bangladesh (Islam, Rahman, Iftekhar, & Rakkibu, 2013), Botswana (Sebele, 2010), and Nicaragua (de los Angeles Somarriba-Chang & Gunnarsdotter, 2012). The high dependence on natural resource for self-subsistence (Belsky, 2009; Moswete, Thapa, & Lacey, 2009; Prachvuthy, 2006; Rozemeijer, 2000; Wunder, 1999) often give communities no choice but to engage in illegal activities. For example, in a case study in Central Amazonian Rainforest, Brazil, some of the families were reported to risk starving because fishing became very difficult and the large-scale agriculture was prohibited in the conservation area (Lima & d'Hauteserre, 2011).

It should be highlighted that nature-based tourism as a complementary activity that substitutes completely, or partially, unsustainable use of natural resources requires a fundamental re-organization of a community's economic and social structure, which might trigger ideological opposition of those communities that have been relying on those activities for generations (Schweinsberg, Darcy, & Wearing, 2018). Local communities who participate in nature-based tourism and receive tangible benefits tend to become cautious in their use of natural resources and, therefore, more likely to support tourism and conservation (Lindberg, 2001). However, the employment in tourism must be high enough in terms of demand to maintain the workforce, and the financial benefits must be higher than gains from unsustainable activities (Kiss, 2004; Mbaiwa & Stronza, 2010).

In destinations where community-based tourism is already in place, but it does not provide enough employment, the unsustainable use of resources is a common practice. The limited economic opportunities reduce or disable any incentives for conservation (Simmons, 1994). Immediately after the incentives for tourism development or benefits from it decrease, local residents go back to previous livelihood-supporting extractive activities (Wilkinson & Pratiwi, 1995). Direct employment is one of the most common limitations of many community-based tourism initiatives as often a small-scale project is not able to employ many people and still remain profitable. A study by Zapata *et al.* (2011) on 34 community-based tourism projects in Nicaragua reported that they were able to generate an average of 6.8 permanent jobs and 12.2 part-time positions. However, it should be stressed that what is considered low or high employment is highly situational as it depends on the size of the community and their direct needs. When resource consumption is prohibited within the protected area, the high dependence on resource extractive activities may also have adverse effect on resources surrounding the area, as demands intensify due to a shrinking resource base (Durbin & Ralambo, 1994; Parry & Campbell, 1992). This might also have a negative effect on tourism itself that is based on supply of pristine landscape, biodiversity of animal and plant species. For example, in Wolong Nature Reserve, China, activities such as logging and clearing for fuelwood, agriculture, gathering of herbal medicinal plants, and ranching have significantly degraded and fragmented giant panda habitat that is the main tourism attraction offered by the local community-based tourism initiative (He *et al.*, 2008).

Nature-based tourism, and the associated reliance on tourism-derived funds for conservation, is also sensitive to economic shocks. For example, as a result of the COVID-19 pandemic, the predicted loss in park entrance fees to the Galapagos Islands is expected to cost between 35-55% of total revenue, monies mainly allocated for conservation (Díaz-Sánchez & Obaco, 2020). Continued conservation in the Galapagos will require alternate sources of funding or loans (Díaz-Sánchez & Obaco, 2020).

The potential for detrimental effects of human-wild species interactions needs to be closely managed, requiring local community empowerment, supportive and cooperation from tour operators and enterprises, and buy-in from tourists (Dou & Day, 2020). The management of the recreational use of wild species needs complementary enforcement and voluntary compliance measures, especially in the tourism context, managing human-wild species interactions is in effect managing people (Dou & Day, 2020). Instilling and supporting a sense of pride and custodianship of wild species amongst tour operators can facilitate responsible tourism.

In summary, for sustainable recreational use of wild species there needs to be:

1. low impact on the wild species being used
2. long-term monitoring of wild species populations and habitats
3. long-term improvement in the livelihoods of local communities
4. awareness and support for conservation from all stakeholders
5. adaptive management and limits on "acceptable change" for wild species tourism, conservation and local communities, including the ability to limit further development
6. supportive regulatory frameworks from local and national government (UNEP/CMS, 2006)

As every tourism initiative is different, there is no single set of suitable conditions that enable both conservation and local livelihoods to flourish (Beeton, 2008; Faulkner & Tideswell, 1997; Okazaki, 2008; Reimer & Walter, 2013). However, a number of factors emerged from a global analysis of over 100 community-based tourism case studies in natural areas (Yanes, Zielinski, Diaz Cano, & Kim, 2019; Zielinski, Kim, Botero, & Yanes, 2020).

Aspects that are critical for a success are:

1. the availability of financial resources
2. skills and technical expertise
3. political influence
4. local control over land and resources
5. community cohesion
6. involvement in local planning and management

The external support provided by non-governmental organizations and governmental organization is crucial for ensuring the abovementioned conditions (Beeton, 2008; Okazaki, 2008)(Beeton, 2006; Okazaki, 2008). The external factors enabling community-based tourism are the political will and decentralization of power and control to the community.

The main barriers for successful community-based tourism development are:

1. the lack of skills and expertise in areas required for tourism
2. lack of noticeable improvement of quality of life in the community (health, education, economy)
3. lack of independence in the decision-making process
4. lack of participative decision making

5. lack of community control over land and resources
6. low level of control over tourism activities in the area
7. internal conflict within community
8. high dependence on resource consumptive activities
9. lack of significant employment in tourism, among others.

3.3.5.2.4 Education and learning

This section deals with the non-extractive practices of wild species for the production of knowledge (Chapter 1). The scope of this section is limited to the non-extractive practices of wild species for learning and education where the impact of this use has a measurable effect on the species. The text below is based on a literature review (Google Scholar) using the following keywords: wildlife, nature, environmental, education, and learning generating 119 000 hits. Articles that were recommended by citation databases were also considered, as well as collected from personal libraries and recommendations from experts. After a title and abstract read, 18 papers were selected for a full text read. After a full text read of these papers, 12 were deemed relevant and assessed for the literature review. Relevance was determined by mention of either the status (current), trend (historical) or impacts of use of a wild species (or taxa) in at least one dimension of sustainability (social, economic, or ecological) (see the data management report for Chapter 3 systematic literature review at https://doi.org/10.5281/zenodo.6452651). These data form the basis of the text below.

Although the use of wild species and ecosystems for scientific research and environmental education, amongst other purposes, is certainly widespread, there is no indication whether this has increased over time or on the current status of use. Relevant articles represented all IPBES regions and most ecosystem types. The literature mostly addressed the use of 'nature' for education and learning, rather than a species/taxa specific approach, but where taxa were mentioned, they were either mammals (terrestrial and marine) or birds. There was little to no information in the literature about the sustainability or the effects of use on wild species or ecosystems. The exception was one article which mentioned concern over the routine use of outdoor teaching sites and their management plan to rotate use of environmentally sensitive areas as needed (Ernst & Stanek, 2006). Although undocumented, the non-extractive practices of wild species are likely to experience similar impacts to recreational watching of wild species such as stress-related responses from wild species and habitat damage through trampling (see section 3.3.5.2.3. Ecological aspects of recreational use).

There are two main methods of using wild species for learning and education. The first is via scientific research and the second through environmental education, mostly to school children and tourists, although a significant amount of informal, experiential learning and knowledge transfer occurs through other practices and uses of wild species, such as birdwatching (recreational use of wild species) (Zvonar & Weidensaul, 2015) and urban foraging (gathering) (Poe, LeCompte, McLain, & Hurley, 2014). Scientific use of wild species is generated through measuring faunal and floral diversity, and population structure and ecological processes. A review of "intellectual ecosystem services" generated by South African National Parks showed a bias towards research on animals, particularly mammals (Smit, Roux, Swemmer, Boshoff, & Novellie, 2017). Similarly, the journals that published research from protected areas were mostly mammal dominated, with little to no focus on social science, environmental governance or social-ecological studies (Smit *et al.*, 2017). Wild species use in education in Europe was dominated by threatened and charismatic species, such as wolves *(Canis lupus)*, brown bears *(Ursus arctos)* and Imperial Eagles *(Aquila adalberti)* (Aguilera-Alcalá *et al.*, 2020). These cases highlight the paucity of research conducted on less "popular" taxa, such as fungi and invertebrates, forbs and shrubs. Notwithstanding, the public's interest in charismatic species has been harnessed effectively for scientific research, such as in the analysis of data such as camera traps (e.g., https://www.zooniverse.org/) or in data collection such as atlas projects (e.g., http://sabap2.birdmap.africa/). These citizen science projects both solve significant big-data processing and collecting challenges facing scientists, as well as providing enjoyment and ecological education to interested citizens.

The second major use of wild species for education and learning is environmental education. Here environmental education is defined as a process that allows individuals to explore environmental issues, engage in problem solving, and take action to improve the environment. As a result, individuals develop a deeper understanding of environmental issues and have the skills to make informed and responsible decisions (EPA 2018: https://www.epa.gov/education/what-environmental-education Accessed on 9 January 2021). Most of this literature focuses on education and on nature rather than wild species *per se*.

Most children today have more access to environmental knowledge through nature documentaries and films than all previous generations (Hudson, 2001). Ironically, such media-educated children in developed countries may fervently campaign for saving polar bears, cheetahs and whales, while they have almost no contact with wild animals or plants common in their own country (Hudson, 2001). There was consensus in the research on environmental education, especially for school children, that educational programs that use the environment for learning supported improved attitudes toward the environment and a desire to look after the environment. An education program specifically on wild Bornean orangutans *(Pongo pygmaeus wurmbii)* led to 13.6-40.4% increase in student knowledge and more positive attitudes towards conservation (Freund *et al.*, 2020). In a study on primary and secondary school children in an environmental education program, 41% of students indicated their feelings about the environment had changed as a result of the nature-based excursion through a combination of observation and instruction (R. Ballantyne & Packer, 2002). Responses include: "I had a better understanding of the impact of people on the forests." (15-year-old) and "Don't feed the native wildlife." (15-year-old).

The benefits of using wild species for learning and education are considerable. In terms of ecological benefits, scientific research on wild species is applied by wild species managers to improve sustainable conservation (Smit *et al.*, 2017). Learning in (and from) nature engenders pro-nature behaviors (Richardson *et al.*, 2016) such as fostering a sense of responsibility and stewardship, and changing attitudes and behavior via increased ecological knowledge (Kwan, Cheung, Law, Cheung, & Shin, 2017). This knowledge can ripple outwards from the primary recipients and be transmitted to parents and neighbors (Vaughan, Gack, Solorazano, & Ray, 2003). Educational courses and formal training on wild species and nature can build constituencies with neighboring communities, indigenous peoples and local communities and other stakeholders, as well as capacity building for future wild species research and management (Smit *et al.*, 2017). Imparting environmental knowledge to tourists and students also provides employment, especially important when this is in local communities involved in these practices (Ternes, Gerhardinger, & Schiavetti, 2016; UNEP/CMS, 2006).

The aspects of engaging with wild species that contribute significantly to conservation education in the public include: watching wild species in their natural habitat, opportunities for close encounters with wild species, opportunities to observe natural wild species behavior, engaging the public emotionally, connection with the public's prior knowledge and experiences, convincing communication, and establishing a link between everyday actions and changes to these actions people can make to foster conservation outcomes, and providing incentives and activities to support behavior change (Ballantyne, Packer, Hughes, & Dierking, 2007).

Beyond generating knowledge and awareness, there is concern on whether knowledge translates into action, and the longevity of pro-environmental awareness and behavior changes. In terms of longevity of pro-environmental awareness and attitudes, there is limited longitudinal research on this aspect. One study on the influence of a six-week bird feeding and monitoring program on school grounds showed that a year later, several schools had continued the program themselves, suggesting that such interventions have the potential to be maintained in the longer term (White, Eberstein, & Scott, 2018). Another example illustrates the benefits of a close engagement with a wild species over the longer term. Here secondary school students reared captive-born juvenile threatened Asian horseshoe crabs *(Tachypleus tridentatus)* for 14 months, which were then released into the wild (Kwan *et al.*, 2017). Rearing involved training students to collect data, test water conditions, and provided opportunities to improve on the protocols through experimentation (Kwan *et al.*, 2017). The students were also tasked with presentations on the importance of horseshoe crabs and after the horseshoe crabs were released, tagged individuals could be tracked by students to monitor their movements and growth (Kwan *et al.*, 2017). The extended period of rearing and engagement with the horseshoe crabs engendered a strong emotional attachment and fostered a sense of responsibility, resulting in more pro-environmental attitudes and behavior (Kwan *et al.*, 2017).

Both Ernst and Stank (2006) and Freund *et al.*, (2020) highlight self-efficacy as crucial to pro-environmental behavior. Self-efficacy engenders the belief that one can personally make a difference and empowerment is key to translating environmental education into pro-environmental action. This is related to Hudson (2001) cautioning that environmental educators should avoid the "psychology of despair." The overwhelming documenting of declines in the health of the natural world and species populations can create a sense of hopelessness for the future and negate the belief that an individual can make a difference.

A drawback of environmental education is the limited reach of the programs. Although some ripple effect in increased awareness in the community (Vaughan *et al.*, 2003), in communities reliant on natural resources and living in vulnerable ecosystems, it is the children who do not attend school who are more likely to be involved in illegal and unsustainable wild species activities in the future (Breuer & Mavinga, 2010). Furthermore, education alone cannot be solely responsible for changes in behavior. Environmental education programs need to be complemented by projects that alleviate poverty and develop alternative livelihood opportunities (Breuer & Mavinga, 2010). Conservationists and local governments should also provide information on the importance of ecological functions of wild species that cause problems in human-wildlife conflict, whilst mitigating the drawbacks of close contact with 'problematic' species (Hosaka, Sugimoto, & Numata, 2017).

3.3.5.3 Emerging issues

Tourism is one of the practices most affected by the COVID-19 pandemic (Spenceley *et al.*, 2021; UNCTAD, 2021). As a result, the COVID-19 pandemic has exposed the vulnerability of nature-based revenue streams to global economic shocks, and the reliance of communities and conservation funds on international tourism (Peter Lindsey *et al.*, 2020; Rondeau, Perry, & Grimard, 2020; Spenceley *et al.*, 2021). Loss of conservation funds has been severe as a result of decreased tourism revenue. For example, the predicted loss in park entrance fees to the Galapagos Islands is expected to cost between 35-55% of total revenue, monies mainly allocated for conservation (Díaz-Sánchez & Obaco, 2020). Continued conservation may require alternate sources of funding or loans (Díaz-Sánchez & Obaco, 2020; McCleery, Fletcher, Kruger, Govender, & Ferreira, 2020). Early evidence from the COVID-19 pandemic impacts suggests that communities reliant on nature-based tourism turned to extractive activities to meet their local livelihood needs (Spenceley *et al.*, 2021), compounded by the return of migrant workers to rural areas and the associated increase in demand of local resources (Rondeau *et al.*, 2020). There are preliminary indications of increase poaching and a surge in illegal logging during the pandemic, possibly as a result of decreased conservation authority presence and no wildlife watching tourists (Rondeau *et al.*, 2020; Spenceley *et al.*, 2021). In addition to the lack of tourism revenue, it is unknown what the impacts of COVID-19 transmission from tourists on wild species will be (A. Gibbons, 2020) or from tourists to local communities (Hakim, 2020). But these early findings still need to be corroborated with more data as the effects of the pandemic on wild species, conservation funds, and local livelihoods becomes more understood.

Another emerging issue is the non-extractive use of wild species through novel finance mechanisms, such as Rhino Impact Bonds (www.rhinoimpact.com), Lion Carbon (www.lionlandscapes.org/lioncarbon), The Lion's Share Fund (www.thelionssharefund.com), or the Luc Hoffmann Institute's Innovation Challenge "Beyond Tourism in Africa" (https://luchoffmanninstitute.org/beyond-tourism-in-africa-innovation-challenge) which seek to support wild species conservation and sustainable livelihoods in the absence of recreational hunting or wildlife tourism. However, there is currently insufficient information on the use, trends or impacts of these finance mechanisms on wild species or wildlife economies.

3.4 TRADE-OFFS AND SYNERGIES

3.4.1 Introduction

Chapter 3 focuses on the status and trends of the use of wild species through its three interacting systems: the wild species themselves, the human practices by which they are obtained from nature, and the uses for which they are intended. Because it is impossible to include all wild species in the assessment, we have focused on those species which are more intensively utilized, those whose sustainable use is of particular concern, and those whose use exemplifies sustainable use in meaningful ways which are informative for overall consideration of the sustainable use of wild species discourse. Throughout the chapter we have followed the practices and uses typology outlined in chapter 1, with adaptations in each section in accordance to the standards in the various literatures and sectors reviewed. However, these use categories (and sometimes practice categories) are not exclusive. In this section we make an effort to consider the interactions among the uses and practices.

While the specific practices and uses of particular wild species have been studied in greater detail, the interactions and influences among species and the related consequences for sustainable use of wild species has been much less examined. These interactions between, within and among wild species-related practices and uses, and their cross-influences relate to the notion of trade-offs and synergies. To avoid developing a compartmentalized and regimented understanding of sustainable use of wild species, the attempt in this section is to use the notions of trade-offs and synergies as analytical perspectives to understand how the practices and uses of wild species are connected in multiple ways, how they interact with each other and, in the process, how they engage with and cross-influence each other both negatively and positively.

According to the IPBES Glossary (IPBES core glossary, 2021), a trade-off is a situation where an improvement in the status of one aspect of the environment or of human well-being is necessarily associated with a decline in or loss of a different aspect. Trade-offs characterize most complex systems and are important to consider when making decisions that aim to improve environmental and/or socio-economic outcomes. Synergies arise when the enhancement of one desirable outcome leads to enhancement of another. Trade-offs are distinct from synergies as the latter are also referred to as "win-win" scenarios.

While it is important to aim for a "win-win" synergy, this cannot be done without appropriate responses to the "win-lose" situations presented by existing and potential trade-offs between and among the practices and uses of wild species. Biophysical, economic and social factors all make it unlikely that multiple needs will be met simultaneously without deliberate efforts; so while there is still much interest in developing win-win outcomes there is little understanding of what is required for them to be achieved (Howe, Suich, Vira, & Mace, 2014; Tallis, Kareiva, Marvier, & Chang, 2008).

While win-wins may be attractive, they are not inevitable and may be unlikely in practice in the absence of carefully designed interventions (Bennett, Peterson, & Gordon, 2009). Howe *et al.*, (2014, p. 263) suggest that "taking account of why trade-offs occur is more likely to create win-win situations than planning for a win-win from the outset. Consequently, taking a trade-off as opposed to a win-win approach, by having an awareness of and accounting for factors that predict a trade-off and the reasons why trade-offs are often the outcome, it may be possible to create the synergies we seek to achieve.". Without attention to trade-offs, one is left with the notion that sustainability of wild species hinges separately on the individual practices and/or uses, which is both ecologically and socially unrealistic.

3.4.2 Conceptualizing trade-offs and synergies

Based on the ecosystem services literature, a two-fold understanding of trade-offs and synergy is proposed: First, trade-offs or synergies only occur if the considered practice and use interact with each other (Bennett *et al.*, 2009; García-Llorente *et al.*, 2015). Second, trade-offs and synergies require assessment of supply, demand and use together and not separately (Geijzendorffer, Martín-López, & Roche, 2015). Following Turkelboom *et al.* (2016) a trade-off is a situation where one use or practice directly decreases the benefits supplied by another. A synergy is a situation where one use or practice directly increases the benefits supplied by another use or practice. Both synergies and trade-offs have spatial and temporal dimensions (see section 3.2).

Trade-offs may depict an array of phenomena including conflicts, contestations, negative correlations, incompatibilities, rivalry and excludability in relation to sustainable use. The inverse of these phenomena signifies synergy. Both trade-offs and synergy are closely associated with benefits and well-being components, value dimensions, and management strategies (Iniesta-Arandia, García-Llorente, Aguilera, Montes, & Martín-López, 2014; Martín-López, Gómez-Baggethun, García-Llorente, & Montes, 2014; McShane *et al.*, 2011). Trade-offs and synergies reflect a host of interactions, connections, relationships and linkages within, between and among practices and uses. If so, achieving the goal of sustainable use of wild species depends on the level of understanding of the key trade-offs and possible areas of synergy within and across practice areas.

3.4.3 A framework to analyze trade-offs and synergies in the sustainable use of wild species

The main purpose behind exploring trade-offs and synergies is to understand their implication for sustainable use of wild species, key trends and status. It is evident from section 3.3 that the assessment considered a large number of wild species, five broad categories of practices and sub-practices, and more than nine different types of uses. A simple three-pronged approach is used to consider the various trade-offs and synergies across these practices and uses of wild species by focusing on (i) trade-offs and synergies at intra-practice and intra-use level; (ii) trade-offs and synergies between practices and uses; and (iii) trade-offs and synergies involving the social, economic and environmental aspects of sustainable use.

3.4.3.1 Trade-offs and synergies at intra-practice and intra-use level

The lack or presence of a range of scientific and indigenous and local knowledge-based methods and their effective combinations for assessing the sustainability of wild species are linked to possible trade-offs and synergies. A diverse range of methods to analyze the status and trends of sustainable use of wild species under each practice category has been discussed in section 3.3. They include both scientific methods (e.g., stock assessment, biomass estimation) and the use of a variety of indigenous and local knowledge. However, there is a predominance of scientific methods for assessment of wild species even though use of indigenous and local knowledge is quite widespread. In fishing practices scientific assessments are publicly available for roughly half of the global fish catch while there is considerable effort to better understand the status of the remaining half of the stocks. This shows how science and technology are focused on only portions of wild species and not all that are important for human use. This may trigger undesirable trade-offs between assessed and non-assessed species. Addressing this may be tricky but not impossible. For example, the state of world fisheries and aquaculture by the FAO makes scientific assessment of status of 500 fish stocks worldwide, while the remaining almost half of the world's stocks are covered through the expertise provided by expert knowledge to fill in the gap (Melnychuk *et al.*, 2017). In some cases, this might mean that small stocks, especially the unassessed ones, are in a disadvantageous position of below target levels compared to large stocks which are often covered under scientific assessment (Costello *et al.*, 2012). In order to ensure that partial nature of scientific information does not lead to ineffective decisions it may be combined with other types of knowledge, including indigenous knowledge, and use pluralistic and interdisciplinary forms of assessment of trade-offs. Further discussion on this appears in Chapter 5 and other sections of this assessment focusing on indigenous and local knowledge.

Trade-offs between multiple uses under a specific practice may reallocate science, technology, investment and innovations in favor of new or emerging uses over the already established and traditional uses of wild species. This may dramatically alter any existing synergies between use categories and significantly impact the sustainability trajectories associated with individual use types of wild species. Section 3.3. offers adequate understanding that while some uses under a practice type are well-established and traditionally recognized, others may be new or emerging in nature. Despite the potential for synergy between these multiple use categories there seem to be inherent competition and overlapping contestations amongst them, ultimately affecting the levels of their sustainable use. In the process of competing with one another, some of the uses have become more prominent than others and thereby known to drive science, technology, investment and innovations away from existing use areas to the new uses that have the potential to negatively affect sustainable use of wild species as a whole.

Several examples of these intense trade-offs ensue from the practices outlined in section 3.3. Fishing offers examples of two sets of mostly conflicting rather than complementary use categories that seem to pose significant challenges to the question of sustainability around this practice: (i) the overlapping interactions between capture fishery and aquaculture; (ii) the tussle between the invisibility of small-scale fisheries and the high visibility of large-scale / industrial fisheries.

(i) Capture fishery and aquaculture: the FAO estimates the total volume of capture fisheries as about 90 million metric tons which constitutes the largest wild food consumed by humans as well as one of the most established / traditional uses under fishing as a practice. Aquaculture as a new use category has gained momentum since the mid-1980's but already in significant competition with capture fisheries. **Figure 3.31** provides estimates of the global fish production as about 170 million metric tons with close to 50 percent coming from aquaculture. This is a consistently increasing trend in the last three decades whereby aquaculture is all set to takeover capture fisheries in the next decade or so. Not only that but aquaculture is reportedly encroaching into the dominant use of capture fisheries for the purpose of food for humans, i.e., out of the 90 million metric tons of fish obtained from capture sources over recent decades, about 60 million metric tons goes to direct human consumption and most of the rest is diverted as feed for aquaculture and livestock. Such trends might threaten to further marginalize the capture fisheries practice which is already experiencing a sharp decline in its biologically sustainable levels from 90 percent in 1974 to 65.8 percent in 2015 and stands at more than 34.2% stocks being overfished (FAO, 2020d). Even though it is not included in the scope of this assessment,

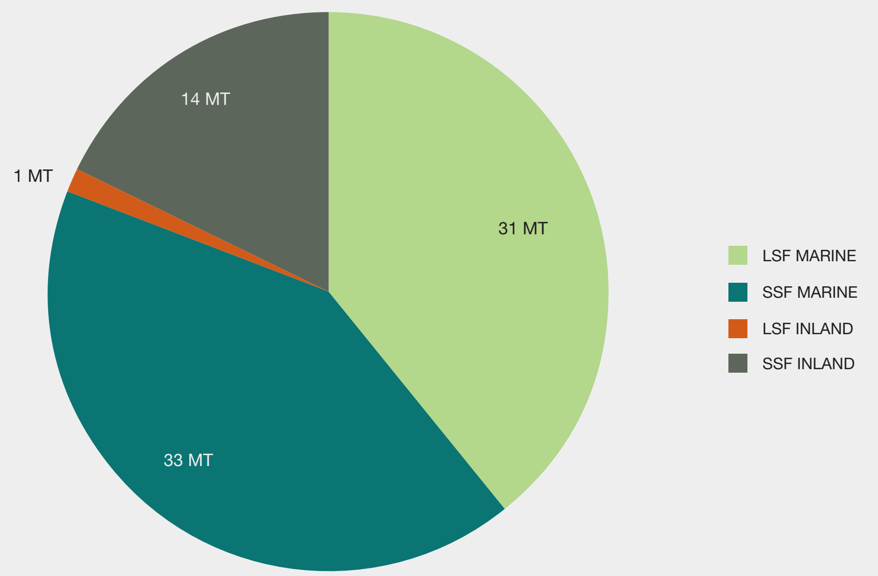


Figure 3 65 **Total annual catch in small-scale and large-scale fisheries around the world.**

Abbreviations: MT: MillionTons, LSF: Large-Scale Fisheries, SSF: Small-Scale Fisheries. Source: (Toobigtoignore.Net – Big Numbers Project Report by World Bank/FAO/World Fish, 2010) under license CC-BY.

reference to aquaculture is imperative because of the significant trade-offs it has with capture fisheries.

(ii) Small-scale and large-scale / industrial fisheries: Data in **Figure 3.65** below clearly shows that the annual catch in small-scale fisheries is higher than the large-scale fisheries both in the marine and inland fisheries practices. In the inland fisheries, small-scale fisheries have 14 times more catch than inland large-scale fisheries. Despite this lead in catch size and the significant contributions small-scale fisheries make to nutrition and food security, poverty alleviation and livelihoods, and local and national economies, especially in developing countries (Béné, Macfadyen, & Allison, 2007; Berkes, 2015; Lilian Ibengwe & Fatma Sobo, 2016), the policy attention this practice has received remains marginal. Small-scale fisheries communities have remained economically and politically marginalized, are highly vulnerable to change (including climate change), and until recently, remained largely invisible in policy debates in most countries and internationally (Berkes, 2015; FAO, 2015). These factors, together with increasing vulnerability due to climate, environmental, economic and policy drivers have created a global crisis in small-scale fisheries (Muzuka, *et al.*, 2011; Paukert *et al.*, 2017; Satumanatpan & Pollnac, 2017).

In contrast, the large-scale fisheries practice has received significant policy attention across the national and international boundaries. A major example of this attention pertains to the extent of global subsidies to the tune of 35 billion United States dollars to the large-scale fisheries practice (Sumaila *et al.*, 2019; Sumaila, Lam, Le Manach, Swartz, & Pauly, 2016). These discrepancies between the small- and large-scale fisheries signify intense levels of trade-offs between the two use types. Possible synergies can be built between these two practices within fishing practices if the small-scale fisheries can be recognized as a use type of wild species that is simply 'Too Big To Ignore' (Chuenpagdee, 2019; Chuenpagdee *et al.*, 2019). Afterall, small-scale fisheries support over 90 percent of the 120 million people engaged in capture fisheries globally, about half of them are women, and it contributes approximately 45% of the global fish catch destined for direct human consumption (World Bank, 2012). Better synergies between these two use types under the fishing practice have the potential to contribute to both goals of ecological conservation and global human development, all of which can potentially lead to sustainable outcomes.

Fisheries bycatch is a growing trend and an example of how increased use of technology and the mechanization of vessel and gear types result in trade-offs. A related area is unreported volumes of fish discarded at sea. According to **Figure 3.65**, the global catch of fish reported by individual countries include only estimates of landing and do not include non-retained catch that are discarded at sea. Globally, estimated discards accounts for about 10% of total annual catches and most discards are generated by industrial (i.e., large-scale) fisheries (Dirk Zeller *et al.*, 2018). Compared to this, landing estimates for small- scale fisheries are widely regarded as an underestimation.

3.4.3.2 Trade-offs and synergies between practices and uses

Trade-offs and synergies are inherently linked to fishing, gathering, terrestrial animal harvesting, logging, and non-extractive practices being treated exclusively or in isolation from each other. Several sections in 3.3 acknowledge the interconnections between practices. However, it was necessary to treat them in a somewhat stand-alone way for clarity in the systematic literature reviews and in reporting. This treatment exposes problems around synergies and trade-offs. Artificially created or not, the disconnections between major practice categories are not healthy for sustainable use of wild species. For example, fishing is not all about fish and fishers alone. Aquatic systems that host fish habitats are integrally connected to terrestrial ecosystems as they mutually benefit or impact each other. Or when people harvest fruits from wild trees (gathering), they may also harvest the entire tree for firewood (logging). People who are primary users, engaged in one of these practices, tend to move between multiple practices and uses either as a seasonal livelihood routine or under pressure from multilevel drivers when their primary engagement in a specific practice is disrupted.

First, several groups of wild species users are known to move between fishing, gathering and harvesting across a range of ecosystems which is influenced by their livelihood, cultural and occupational needs and complementary seasonality among the wild species, i.e., occurrence and availability. Second, even those who depend on one specific use or practice category as their primary source of food, subsistence or livelihoods are seen entering into other practices and uses of wild species due to unforeseen pressures. This includes increasing instances of how coastal inhabitants, primarily reliant on fishing, are forced to engage in the harvesting, gathering and use of wild species under non-fishing practice categories when they are faced with loss of fish and related income due to multilevel pressures. Natural disasters (i.e., cyclones, floods, tsunami, earthquakes, etc.) are known to push people from one practice and use category to others through temporary, semi-permanent and permanent displacement. It is understood that better synergies will facilitate this cross-practice mobility of users, especially in times of crisis and positively contribute to the sustainable use of wild species. It may also help resolve negative trade-offs between the practices or, at the minimum, bring them up for timely attention.

As discussed in the preceding paragraph, if trade-offs between practices are related to their separation from each other, the need to consider synergies between practices at global, regional, national and local policy and program levels cannot be underestimated.

Trade-offs between uses within a practice may be related to differential policy attention each of the uses has received. For example, the use that generates the most revenue may move up in the hierarchy and receive most policy and related attention, and this may take place at the cost of other uses. It is evident that in the tussle between capture fisheries and aquaculture the later receives significantly higher attention compared to the former (section 3.3.1). Similarly, tourism tends to grab significant policy and program attention as compared to medicinal, ceremonial and cultural uses (section 3.3.5).

The spaces and places where practices occur influence the nature of trade-offs and synergies. Section 3.3 offers numerous examples of this. Fishing practices are specific to marine, coastal, inland sectors as dominant fisheries within which multiple uses are operationalized by the users and the possibility of trade-offs and synergies are within and between these spaces. Similarly, gathering, harvesting, logging and non-extractive practices take place within multiple resource sectors that tend to interact and influence each other. Each of these resource sectors have their own social, economic, cultural, political and ecological characteristics which shape the nature of the practice and uses. Trade-offs and synergy result from how the practices and uses across these multiple sectors interact and influence each other.

It is important to recognize that trade-offs and synergies may not only be considered to be existing between or within practices and uses but also the scale at which they operate has significant role. The tussle between the small-scale and large-scale fisheries is more about scale than anything else (as discussed above). There can be multiple interpretations of how scale is linked with trade-offs and synergies in other practices and uses.

Related to scale, understanding trade-offs and synergies between and within geographical contexts within which practices occur is important. The literature review on small-scale fisheries in section 3.3 provides detailed account of geographical context specificities by characterizing small-scale fisheries within Europe, Arica, Asia, Latin America, North America, and the Pacific. It is important to note that the key characteristics and the major drivers influencing trade-offs and synergies in each of these geographical regions of the world may significantly vary.

Section 3.2 offers a systematic analysis of the role of indicators in understanding sustainable use of wild species. Examination of the sustainable indicators has a lot to offer in terms of clarifying trade-offs and synergies. In fact, many sustainable use indicators are indicators of trade-offs and synergies. Tools such as monitoring in many indigenous peoples and local communities focus on interlinked social and ecological elements and can inform the development of local and global indicators that recognize these linkages. The acknowledgement of the value of including the knowledge of indigenous peoples and local communities contribute importantly to monitoring and assessment of the species and ecosystems used by these communities.

Trade-offs and synergies between knowledge systems guiding the practices and uses is a whole new area to explore. While the politics and power dynamics between knowledge systems (Van Assche, Beunen, Duineveld, & Gruezmacher, 2017) may be inherently connected to trade-offs, systems of knowledge coproduction (Norström *et al.*, 2020) signify synergies between sustainable use practices. In this context, indigenous knowledge is increasingly challenging to pass on because the environments in which indigenous and local communities live are threatened (3.3.5). Indigenous peoples and local communities report a loss of nature that supports their local livelihoods and well-being, in part as a result of natural resource extraction by both outsiders and locals (Ichii, Molnár, Obura, Purvis, & Willis, 2019). Increasing efforts to synthesize indigenous and local knowledge have shown that the natural indicators indigenous peoples and local communities use are reasonably compatible with scientific knowledge and show their deep connection with nature, albeit it at a very local scale (Ichii *et al.*, 2019). Indigenous and local knowledge is increasingly being used to generate more accurate data on species trends, non-iconic species data and geospatially relevant data using technology (e.g., Cybertracker: (Ansell & Koenig, 2011; Liebenberg *et al.*, 2017), participatory mapping using Google Earth (Peters-Guarin & McCall, 2012) and Open Data Kit (ODK): (Jeffers, Humber, Nohasiarivelo, Botosoamananto, & Anderson, 2019)). However, it is important to note that the goal in working with indigenous peoples and local communities is to honor their knowledge in its own right, not only when it is compatible with scientific knowledge or supportive thereof (Barron, Sthultz, Hurley, & Pringle, 2015).

Changing gender roles and dynamics can lead to the disruption of existing synergies and the creation of new trade-offs. One case that shows the complexity of trying to develop sustainable use practices based on gender assignments of particular practices and uses is the gathering practices of orchids in Tanzania. The majority of gatherers of wild edible orchids are female, orphans also commonly engage in this practice, and there are slightly more boys than girls among orphans affected by HIV/AIDS (Human Immunodeficiency Virus) in villages in the southern highlands of Tanzania (Challe & Price, 2009; Challe, Struik, & Price, 2018). Children gather less species than adults and generally learn about the use and identification of wild species from their mothers and to a lesser extent also from their fathers (Cruz García, 2006; Łuczaj & Nieroda, 2011). When children's parents die before they share their knowledge, orphans teach each other and or learn from middlemen as a result of "trial and error". This can lead to the gathering of too many non-marketable orchid tubers, which may in turn negatively affect the sustainability of the practice (Challe *et al.*, 2018).

3.4.3.3 Trade-offs and synergies involving the social, economic, environmental and policy aspects of sustainable use

Sustainability is multidimensional but the essence of it can be captured by considering the social, economic and environmental aspects as inclusive categories. The questions about trade-offs and synergies are integrally linked to the three pillars of sustainability, i.e., economic viability, environmental protection and social equity (Purvis, Mao, & Robinson, 2019). Policy is also recognized as a supporting element of sustainability. In other words, economic, social, environmental and policy aspects of the sustainable uses of wild species help link practices and uses with key sustainability parameters. While negative trade-offs among and between these parameters threaten the viability of sustainable use, synergies among them provide pathways for sustainable use. In many contemporary societies, terrestrial animal harvesting has multiple functions and sustainability hinges on the synergies and trade-offs between social, ecological and economic dimensions of this specific practice. Human populations engage in animal harvesting (such as, hunting and trapping) to meet a range of nutritional, economic, medicinal, cultural and recreational needs and the level of synergy between these needs may have implications for the level of extraction of the resource, therefore its sustainability.

International and national policy instruments and guidelines, along with civil society actions have supported processes to resolve negative trade-offs and potentially build synergies between practices and uses. There is strong evidence presented in section 3.3 to support this conclusion. The impact of fishing on marine ecosystems other than the target species and their habitats is well established. Several international instruments (such as agreements, policies, protocols, treaties) have been developed to help respond to these challenges and provide guidance for action. Prominent among those are United Nations Convention on the Law of the Sea, created in 1982, which established the 200-mile exclusive economic zone and the concept of maximum sustainable yield as an international measure for sustainable fisheries management.

Given the inadequacies associated with the United Nations Convention on the Law of the Sea regarding fish stocks that range across multiple exclusive economic zones or in the high seas, the United Nations Fish Stocks Agreement, 2001 was brought into effect to offer international protocols for managing the overlapping stocks. Further, the Food and Agriculture Organization of the United Nations has put in place a range of international policy guidelines to promote sustainable use of aquatic ecosystems and facilitate the conservation of biodiversity of ecosystems by minimizing trade-offs in forms of competition, contestations and unsustainable practices. These include: the 1995 Code of Conduct for Responsible Fisheries (FAO, 1995b); Voluntary International Plan of Action on reducing the incidental capture of seabirds in longline fisheries (FAO, 1999a); International Plan of Action on the Conservation and Management of Sharks (FAO, 1999c); International Guidelines on Reducing Marine Turtle Fishing Mortality (Eric Gilman & Bianchi, 2010); International Guidelines on Managing Fisheries Bycatch (FAO, 2011); Small-Scale Fisheries Guidelines (FAO, 2015).

While these international policy measures have produced favorable results, there are gaps that still exist, such as the issue of the sustainability of non-target species relative to target fish stocks is still unclear. This indicates that species that are not covered by a treaty or international policy may be subject to overexploitation and, therefore, unsustainable or in the process of being so. In order to address this, further responses have come through the legally binding United Nations resolution 61.105 (2005) which provided for responsible fishing in vulnerable marine ecosystems and of non-target species. Additionally, the International Agreement on Port State Measures (FAO, 2016a) aims to prevent, deter and eliminate illegal, unreported and unregulated (IUU) fishing by preventing vessels engaged in illegal, unreported and unregulated fishing from using ports and landing their catches. These measures suggest that they are geared towards addressing factors (e.g., illegal, unreported and unregulated) that can trigger trade-offs and create barriers for possible synergies.

Apart from the international responses to critical trade-offs, major efforts have also come from national governments and non-governmental organizations. For example, the formation of the Marine Stewardship Council (1997) to improve fisheries sustainability along with the initiation of several environmental non-governmental organizations for marine conservation, expansion of the science and management efforts by national and regional governments including the Common Fisheries Policy in the European Union are important landmarks.

The above discussion suggests that the history of sustainable use of capture fisheries is closely tied with a number of critical international, national, regional policy guidelines, and non-governmental organizations and civil society action focusing on fisheries and their ecosystem conservation. These policy instruments and agreements provide a strong foundation for possible actions and responses to trade-offs and processes through which synergies for sustainable use can be achieved.

3.4.4 Selected case studies of trade-offs and synergies in sustainable use

The following cases studies help explore the question 'whether non-extractive uses can become an alternative to extractive uses?

3.4.4.1 Whaling and whale-watching

Whale watching is commonly seen as the global success story of a non-extractive use replacing an extractive use, and in the process encouraging sustainable use, generating economic revenue and contributing to conservation. The growth in whale watching is a result of bans on whale hunting, the decline in whale-derived products, and environmental campaigns by non-governmental organizations to support whale watching as a sustainable alternative to whale hunting (Neves, 2010). Many whale populations are recovering after the global commercial moratorium was enacted on whaling in 1985, although determining the status for some populations has remained challenging (IWC, 2020a). In addition, some countries continue to hunt whales under objection or reservation to the moritorium, or because they are not members of the International Whaling Commission (see section 3.3.1.4.5 above for additional discussion of this point).

Whale watching has undoubtedly become a lucrative industry, particularly for tour operators in developing regions who often enjoy direct income streams considerably greater than existing levels of regional gross domestic product per capita (Mustika, Birtles, Welters, & Marsh, 2012) and so, by extension, for local communities that benefit from the tourism activities. Whale watching tourism has brought additional revenue to the Maoris in Kaikoura, New Zealand (Curtin, 2003), and the inhabitants of both Lajes in the Azores (L. Silva, 2015) and Baja, Mexico (Schwoerer, Knowler, & Garcia-Martinez, 2016), through direct expenditure on tours but also through the accompanying expenditure on transport, accommodation and hospitality. It has also brought positive attitudinal effects amongst whale-watching tourists and local populations. Mintzer *et al.* (2015) note that the creation of a sustainable development reserve and the presence of dolphin researchers have had positive effects on the attitudes and behaviors of an indigenous fishing community on the Amazon towards botos, which have in the past been killed for both bait and superstition. Wilson and Tisdell (2003) report that 78% of whale- watching tourists visiting Hervey Bay, Australia, find the experience convinces them of the need for a worldwide ban on whaling, 80% of the need for greater protection of whales in Australia, and 73% to be more likely to report whales that are stranded, injured or mistreated: biocentric effects that are supported by the findings of Gowreesunkar and Rycha (2015). However, whale watching is not without negative impacts on whales and marine ecosystems. The International Whaling Commission has released a Whale Watching Handbook addressing these concerns and supporting sustainable whale watching (IWC, 2020b).

In some areas, whale watching and whaling co-exist. Whale watching is the more economically lucrative and globally accepted activity. Although whaling depends on government subsidies, some indications are that public support for whaling in whaling countries like Japan and Iceland is growing as a perceived cultural and nationalistic right (Andersson, Gothall, & Wende, 2014; Cunningham, Huijbens, & Wearing, 2012). Yet, in both Japan and Iceland, whale watching tourism is booming (Cunningham *et al.*, 2012), but there has been concern that continued whaling alongside tourism will negatively impact tourism industries (Bertulli, Leeney, Barreau, & Matassa, 2016; Cunningham *et al.*, 2012; Hoyt & Hvenegaard, 2002; Kuo, Chen, & McAleer, 2012; Orams, 2001; Parsons & Draheim, 2009; Parsons & Rawles, 2003), the extremes of which could result in tourism boycotts such as happened in St. Vincent and the Grenadines (Hoyt & Hvenegaard, 2002).

The whaling-whale watching nexus is complex and the discourse at the intersection of these activities needs further research (Cunningham *et al.*, 2012). There are contradictory tensions involved in whaling-whale watching that need unpacking. Tourists who eat whale meat are also pro-conservation and support the ban on whale hunting (Burns, Lilja Öqvist, Angerbjörn, & Granquist, 2018). Ironically, the market for whale meat is strongest for tourists (Bertulli *et al.*, 2016; Rasmussen, 2014). Iceland seems to have retained tourists who are tolerant of whaling (especially for subsistence) and who support local and cultural expression, but at the cost alienating tourists who cannot reconcile with whaling for commercial, scientific or indigenous reasons (Andersson *et al.*, 2014). Although the number of whale watching tourists has continued to grow in Iceland since 2002 when whaling resumed, the relative contribution of whale watching tourism to other tourist activities has declined (Andersson *et al.*, 2014). Overall, whaling seems likely to face increased global resistance and unlikely to generate substantial economic incentives, whilst whale watching has global support and generates substantial revenue. It would be prudent for whaling countries to assess the implications of the negative impacts of whaling on their national "image" – their biggest tourism asset (Hoyt & Hvenegaard, 2002) – and conduct a thorough compatibility analysis. Conversely, highly visible national policy for cetacean conservation can attract tourists (Parsons & Draheim, 2009).

3.4.4.2 Recreational trophy hunting and wildlife watching tourism

Trophy, sport or recreational hunting has attracted increasing negative attention, particularly since the widely publicized killing of “Cecil the Lion” in Zimbabwe in 2015. Trophy hunting has long been banned in some source countries (e.g., in Kenya since the 1970s) and efforts have been made to restrict it by banning imports of hunting trophies, at least from certain species, in consumer countries (e.g., France banned the imports of lion trophies in 2015, while the Netherlands and Australia banned imports from a wide range of species in 2016 (Ares, 2019). Trophy hunting can have negative impacts on wild species populations, particularly if offtake is too high or where infanticidal population dynamics exist (e.g. Loveridge, Searle, Murindagomo, & Macdonald, 2007; Milner, Nilsen, & Andreassen, 2007; Wielgus, Morrison, Cooley, & Maletzke, 2013) but can also positively impact conservation and local livelihoods, particularly by generating revenue from habitat and species conservation (Naidoo, Weaver, *et al.*, 2016; Snyman *et al.*, 2021). Debates have played out in the scientific literature and beyond as to the ecological, social and economic costs and benefits of hunting, but one key element of arguments against hunting has been that such extractive practices are repugnant because of ethical issues concerning certain types of harvesting of wild species. It has consequently been suggested that one solution would be to replace such practices with non-extractive uses, and in particular, with wildlife watching (e.g., photographic tourism).

This argument assumes, in the first place, that wildlife watching is indeed a non-extractive use of wild species. Some commentators would argue against this on the basis of its negative ecological impacts on some species and ecosystems. For example, Ballantyne and Pickering (2013) identify tourism as a problem for 46% of threatened vascular plant species in Europe alone, while it has also been documented as limiting cheetah reproduction (Broekhuis, 2018). In addition, wildlife watching can have wide ecological impacts, including water use and carbon emissions (Gössling *et al.*, 2012; Spenceley, 2005).

A key argument for the conservation benefits of recreational hunting is similar to that made for photographic tourism, i.e., income is generated and this plays a role in in i) directly financing conservation agencies including national parks authorities (e.g., Brink, Smith, Skinner, & Leader-Williams, 2016; Lindsey *et al.*, 2020), and ii) providing an incentive for habitat and biodiversity conservation beyond state-managed protected areas by communities and private landowners. Opponents of hunting suggest that wild species are worth far more for wildlife watching tourism than for hunting. For example, a report by the David Sheldrick Wildlife Trust (2014) estimated that a single elephant may be worth 1.6 million United States dollars over its lifetime through income from photographic tourism. A wider review by Lindsey *et al.*, (2007) highlighted that photographic tourism undoubtedly generates greater gross revenues than trophy hunting at a continental scale across Africa. Importantly though, they note that even if smaller, “hunting revenues are significant because they enable wild species production to be a viable land use across a wider range of land uses than would be possible relying on revenues from photographic nature-based tourism alone.” Unlike wildlife watching tourists (generally, obviously exceptions may apply) hunters are often prepared to hunt in areas lacking attractive scenery, and require less infrastructure, therefore minimizing habitat degradation (Di Minin *et al.*, 2016). Because wildlife watching tourism is not viable in all the places where hunting happens, the suggestion that one type of use can simply be replaced with another is thus naïve. For example, Lindsey *et al.* (2006) argue that not all land suitable for trophy hunting would be suitable for wildlife watching tourism, and that low visitor numbers would be unlikely to make it economically viable. Similarly, in Botswana, a ban on trophy hunting implemented in 2014 meant that communities were forced to shift their income earning opportunities from hunting to wildlife watching tourism (Mbaiwa, 2018). Photographic tour operators apparently had little interest in developing lodges in the concessions that lacked high tourism potential (Winterbach, Whitesell, & Somers, 2015). Consequently, there was a reduction of local benefits such as cash income, employment opportunities, scholarships and funeral insurance. This lack of local economic benefits had negative effects on conservation including negative attitudes by rural residents towards wild species conservation and an increase in poaching (Mbaiwa, 2018).

Very few studies have directly compared the benefits of trophy hunting and wildlife watching tourism to the same people, in the same location. One that has is an analysis of communal conservancies in Namibia (Naidoo, Weaver, *et al.*, 2016). The study looked at financial and in-kind benefit streams from wildlife watching tourism and hunting on 77 Namibian communal conservancies from 1998 to 2013. It found that although total benefits from hunting and tourism increased at roughly the same rate, conservancies typically started generating benefits from hunting within three years of formation compared to after six years for photographic tourism. Regarding the types of benefits, the majority (64%) of benefits from trophy hunting were in the form of cash for income for conservancy management, while 32% of benefits were meat for the community at large. In contrast, 58% of the benefits from wildlife watching tourism were in the form of jobs, with 30% used for conservancy management. A simulated ban on trophy hunting significantly reduced the number of conservancies that could cover their operating costs, whereas eliminating income from wildlife watching tourism was still negative but a less marked effect. The study concluded that maintaining both trophy hunting and wildlife watching tourism was likely to produce the greatest incentives for conservation while only focusing on one would reduce the competitiveness of wild species as a land-use option and harm the viability of community-based conservation efforts in Namibia, and possibly elsewhere.

Other comparisons that have been made between hunting and wildlife watching relate to the broader environmental impacts of the two activities. Di Minin *et al.*, (2016) argue because there are fewer trophy hunters compared to wildlife watchers and because it can generate more revenue from a smaller number of visitors, trophy hunting can have a smaller footprint than wildlife watching tourism in terms of carbon emissions and infrastructure development. One case study where an analysis has been conducted between numbers of hunting tourists compared to photographic tourists is Timbavati Private Nature Reserve in South Africa (Timbavati Private Nature Reserve News, 2020). The annual operating budget of the reserve is currently 1.26 million United States dollars which is generated primarily through wildlife watching tourism and hunting. In 2016 an analysis by the reserve's management team found that the conservation levies paid by the approximately 24,000 wildlife watching tourists who visited the reserve that year amounted to less than a third of the income earned from the 46 hunters who visited over the same period (Conservation Frontlines, 2020). The reserve has subsequently increased the fees charged to wildlife watching tourists to increase revenue without having to increase the number of bed-nights, and hence the human footprint. Similarly in Tanzania, Estes (2015) suggests that trophy hunting and wildlife watching bring in similar amounts to the Tanzanian economy but the ratio of tourists who come to see the wild species and hunters who come to shoot it is many hundreds to one with one hunting tourist paying at least 10 times as much as every wildlife watcher.

3.4.4.3 Elasmobranch tourism opportunity and shark fishing

Just as whale watching has contributed to the decline of whaling, there is opportunity for shark and ray watching tourism to mitigate shark fishing effects by providing additional income sources. In a review on elasmobranch tourism, Healy *et al.* (2020) demonstrate that the tourism value of individual sharks exceeds the fisheries value, contributing revenue to developing countries. In Palau, shark tourism contributed over 18 million United States dollars, 8% of the 2012 gross domestic product and the tourism value of sicklefin lemon sharks *(Negaprion acutidens)* in French Polynesia exceeds the payment received by fishers (Healy *et al.*, 2020). Diving, snorkeling, feeding and cage diving currently occur in 42 countries focusing on 49 target species, predominantly in tropical and subtropical Africa, Oceania, Asia and the Caribbean, but also in temperate seas such as Canada, England, Scotland, Japan and New Zealand (Healy *et al.*, 2020).

There may be unintended social-ecological feedbacks between different uses (e.g., tourism, fishing) and wild species. An interesting example is the decline in white sharks *(Carcharodon carcharias)* in South Africa as part of the greater social ecological system. The decline has been of ecological concern, but also impacts on the white shark tourism industry. Killer whale *(Orcinus orca)* presence was initially attributed to the decline, as killer whales have been shown to displace white sharks (Jorgensen *et al.*, 2019). However, continued white shark decline outside of the season niche overlap with killer whales has prompted speculation that demersal long line fishery of smaller shark species, a white shark resource, mostly exported to Australia for human consumption (Braccini, Blay, Harry, & Newman, 2020). This speculation, in turn, resulted in boycott calls against the Australian 'fish and chip' sales to protect South African white sharks, which has negatively affected an overall legitimate and sustainable Australian industry (Braccini *et al.*, 2020).

Sea horse tours and extractive sea horse harvesting

An interesting local example of non-extractive use replacing extractive use is the case of sea horse *(Hippocampus reidii)* tours by self-organized and self-governed *'jangadeiros'* in a Brazilian village (Ternes *et al.*, 2016). Here the local communities impart their comprehensive local ecological knowledge of sea horses to tourists. They take tourists out by raft boat and dive sea horse specimens out to hold in glass jars for viewing by the tourists before releasing them back into their habitat (Ternes *et al.*, 2016). The community involved in these tours no longer harvest sea horses for medicinal or ornamental purposes as they derive economic benefits from them *in situ*, unlike other villages in the region (Ternes *et al.*, 2016). The authors of this case study suggest that by careful guiding this non-extractive approach could be expanded to other villages to benefit sea horse conservation and local livelihoods (Ternes *et al.*, 2016).

These case studies show that while non-extractive uses can improve the conservation status of wild species and improve livelihoods in a sustainable fashion, it is unlikely that complete extractive use will be halted. As always, careful consideration of the context and the implications of such a shift need to guide interventions. Furthermore, the eradication of extractive activities is not necessarily desirable, especially where the extractive use fosters cultural practices that result in conservation. Yet, where extractive indigenous peoples and local communities' use occurs in conjunction with non-extractive use there is potential for conflict as a result of the opposing value systems between the user groups (see section 3.3.5.2.3). And there are nuances between different forms of extractive or non-extractive use. For example, illegal poaching has been shown to negatively impact wildlife tourism (Naidoo, Fisher, Manica, & Balmford, 2016) whereas hunting concession impacts on wildlife tourism can be avoided with careful management.

Choices around sustainable use of wild species will not always be between extractive and non-extractive use. Novel financial mechanisms such as Lion Carbon (2020) or 'rhino bonds' (Aglionby, 2019) may provide important alternatives for some areas, but these currently are only nascent initiatives. It is important to recognize that this is not just about the degree of benefits but their distribution, as benefits from one type of use may be distributed very differently compared to another. So, it may be tempting to conclude that because a non-extractive form of wild species use (e.g., shark watching) has the potential to generate more revenue and jobs than an extractive form (e.g., shark fishing) the former is more sustainable. The same applies in cases where extractive appears to be "better" than non-extractive use. However, sustainability is about more than just economics: the benefits and costs of different activities may accrue to very different stakeholder groups, which is likely to affect the degree to which each option is viewed as socially sustainable. Ultimately, wider non-economic aspects of sustainability of different uses (e.g., likely long-term impacts on the wild species population, interactions of that use with other conservation threats, resource demands of the users, perceived social acceptability etc.) should also be considered when examining trade-offs between different options. Furthermore, the likelihood of unsustainable activity should also be factored in to provide a reliable comparison, particularly the likelihood of land conversion to non-wildlife-based land uses under different scenarios. In all cases, understanding who benefits, and how, from the use of wild species is critical to designing effective policies and programmes that encourage the sustainability of that use and incentivize conservation over other land and resource use options.

3.4.5 Key attributes necessary to respond to trade-offs and strengthen synergies in sustainable use

In the use of wild species, there are synergies and trade-offs among the policies, practices and technologies used to address individually the issues of loss of biodiversity (wild species), land degradation, water pollution and climate change. Economic, ecological and social dimensions play pivotal roles in setting the context for use of wild species; the ways wild species are used differ under different economic conditions, law enforcement regimes, culture and traditional meanings and perception of users. Evidence supports that there are risks associated with the harvesting of wild populations under challenging conditions, and these are often highlighted in low-income countries (Leao, Lobo, & Scotson, 2017) although they can occur in developed countries as well. Therefore, the issues are strongly interconnected and cannot be addressed in isolation (Watson, 2005).

Better understanding of the underlying mechanisms and motivations for trade-offs and synergies can be beneficial for planning and managing sustainable use through (i) predicting and anticipating where and when trade-offs might take place; (ii) reducing undesirable trade-offs and related conflicts; (iii) enhancing desirable synergies; (iv) promoting honest dialogue, creativity, and learning between concerned user / stakeholder groups; (v) creating more effective, efficient and credible management and governance decisions; and (vi) obtaining more equitable and fair outcomes by taking into account distributive impacts of trade-offs (Turkelboom *et al.*, 2016). Key lessons on trade-offs and synergies pertaining to sustainable use of wild species include, but are not limited to:

- Trade-offs and synergies reflect a host of interactions, connections, relationships and linkages within, between and among practices and uses. Without consideration of these interactions and their effects, sustainable use cannot be adequately assessed.

- While trade-offs and synergies between uses within a practice is somewhat well understood, the exact nature of trade-offs and synergies between practices, for example the interactions among gathering and fishing, are not very well studied. This knowledge gap involving the lack of inter-practice trade-offs and synergies has the potential to adversely impact sustainable use of wild species.

- Bifurcation of existing uses and the emergence of new uses within a practice area (e.g., capture vs. aquaculture within fishing practices; ceremony and cultural expression vs. recreation (tourism) within non-extractive practices) have led to a reconfiguration of intra-practice trade-offs and synergies. These changes drive technology, science, investment, policy focus, innovation away from existing use areas to the new uses that have the potential to negatively impact sustainable use of wild species as a whole.

- Trade-offs and synergies between and among fishing, gathering, terrestrial animal harvesting, logging, and non-extractive practices are inherently linked but often treated exclusively or in isolation from each other. This exclusivity is reflected in the dominant culture of practice-specific policies leading to significant compartmentalization. Consideration of trade-offs and synergies between these practices and their use categories across global, regional, national and local policy and program levels could enhance sustainable use of wild species.

- A combination of indigenous and local knowledge and scientific knowledge is effective to better understand and respond to the trade-offs and synergies relating to status and trends in sustainable use. Knowledge co-production processes based in ongoing collaborations are useful in this respect.

Due to uncertainty and the plurality of values and information on wild species, addressing trade-offs requires inclusive adaptive co-governance that is sensitive to power dynamics, principles of justice and equity.

3.4.5.1 Levels and scales at which trade-offs and synergies occur

Trade-offs and synergies are scale-bound. IPBES Glossary defines scale as the spatial, temporal, quantitative and analytical dimensions used to measure and study any phenomenon, i.e., trade-off and synergy in this case (IPBES core glossary, 2021). The need for considering multiple scales (and levels) at which trade-offs and synergies around sustainable uses take place bears significance (Carpenter & Brock, 2006; Mayer, Pawlowski, & Cabezas, 2006). Empirical insights have been recorded from observations of modifications and reorganizations of system dynamics at the level of the ecosystem (Stephen R Carpenter & Kinne, 2003; Scheffer & van Nes, 2004). However, choices about scale of observation are not easily matched with strategies for intervention. For example, specific components of a wild species use regime can cross thresholds (understood as synergy) and lead to varying outcomes at substantially different temporal and spatial scales associated with the influences resulting from trade-offs. There is also an issue that boundaries that delineate units of scale (e.g., ecozones, *de jure*/formal administrative boundaries) do not always correspond to the reality of the ecosystems or human use which are instead (at best) 'soft' (as opposed to 'hard') boundaries (Norris, 2014; Veldhuis *et al.*, 2019). Systematic treatment of trade-offs and synergies relevant to sustainable use of wild species will require scale-sensitive perspectives, and reflection on appropriate scales of understanding and intervention (Scheffer, Westley, & Brock, 2003).

A related aspect of scale is to focus on the units of analysis for measuring trade-offs and synergies. Studies on ecosystem changes and shifts in use and management regimes tend to have mostly emphasized a single resource (or practice) type (Biggs, Carpenter, & Brock, 2009; S. R. Carpenter & Brock, 2006; Scheffer *et al.*, 2003), including marine systems (Beaugrand, 2004; Mantua, 2004; Steele, 2004), lakes and lagoons (Gal & Anderson, 2010; Scheffer & van Nes, 2004), freshwater systems (Carpenter and Kinne, 2003), forests (Ludwig, Jones, & Holling, 1978), woodlands (Dublin, Sinclair, & McGlade, 1990), dry lands (Foley *et al.* 2003), rangelands (Skaggs *et al.*, 2011), and agroecosystems (Gordon, Peterson, & Bennett, 2008), all of which act as sources of wild species. However, using individual resource systems (or practices) to define boundaries of sustainable use inevitably neglects the full range of human expectations from, and interactions with, the larger social, ecological and environmental system necessary to achieve sustainability (Nayak & Armitage, 2018). Incidentally, critical trade-offs and opportunities for building synergies may be missed if a narrow focus on the scale of sustainable use is adopted. Therefore, it is important to conceive what is an appropriate social-ecological unit within which to best capture trade-offs and synergies and why this is critical for observing trends and reporting status of wild species. Units of analysis of trade-offs and synergies in sustainable use may have both a physical (e.g., coastal line, bottom, rivers, and vegetation, landscape) and a normative (e.g., culture, rituals, law, institutions, social interactions) dimension to their boundaries. Recognizing and understanding both these dimensions are useful from scale-sensitive perspectives.

3.4.5.2 Equity and justice considerations in responding to trade-offs and negotiating synergies

How can it be ensured that the outcomes of trade-offs and synergies associated with sustainable use of wild species are distributed equitably? The procedural and distributive aspects of trade-offs and synergies offer multiple pathways to sustainability, depending on the culture and the ecosystem. If inequity and injustice reign, there are few and often no sustainable pathways.

Greater attention to equity and social justice considerations (i.e., winners and losers in the context of sustainable use) is needed to better understand the process and outcomes of trade-offs and synergies. Recognizing issues around sustainable use, trade-offs and synergies through the prism of social and environmental justice facilitates the identification of key motivations of users and main ingredients influencing these processes. Section 3.3 presents material that points towards equity and justice as both cause and effect of trade-offs and synergies. For example, an equity and social justice perspective helps clarify if outcomes from critical trade-offs disproportionately impact a multitude of users, e.g., poor, disempowered and other marginalized communities including women through a process of uneven distribution of benefits and impacts (see Walker & Bulkeley, 2006). Literature from multiple disciplines suggests that changes and shifts in ecosystem processes, structures, functions and services associated with sustainable use may redistribute benefits among stakeholders (Selkoe *et al.*, 2015), and such redistribution may lack sensitivity to equity and justice issues. It is important to recognize that these shifts and redistribution processes are inherently linked to unresolved trade-offs and the absence of synergies among practices and uses of wild species.

In economically and socially stratified social-ecological systems that host wild species, the outcomes of trade-offs and synergies pertaining to the diverse use regimes are often beneficial to some while adversely affecting others (Nayak, Armitage, & Andrachuk, 2016). For example, case studies by Nayak and Berkes (2010), Armitage and Marschke (2013), and others provide evidence that changes in the management practices in coastal and inland fisheries of Bay of Bengal and South China Sea (e.g., outcomes of the trade-offs from the introduction of aquaculture within a predominantly capture fishery system) have benefitted higher caste or wealthier aquaculture owners respectively but have proven negative for customary users. This trend is also evident in section 3.3. These experiences clarify that trade-offs around sustainable use can create new opportunities and upward social and economic mobility for some users (and in this case those that were already upwardly mobile) but simultaneously exclude others (often those already marginalized). Such discrepancies in the nature and level of impacts are related to power and authority, structural advantage and institutional and political favor. Consequently, equity and social justice conditions influence how sustainable use related trade-offs, synergies and the outcomes thereof are 'framed' by certain groups as significant or not, and to what extent that framing can be used to design strategies to respond.

An additional consideration pertains to a multi/inter species justice dimension within the trade-offs and synergy discussions. This underscores the question whether sustainable development can really be accomplished without taking animals' own interests into account (Visseren-Hamakers, 2020). It is important to consider trade-offs and synergies between human and non-human justice leading to further explorations about the types of relationships humans can cultivate with animals so as to produce just outcomes. In doing so, neglecting the spiritual and cultural can also result in the lack of attention to the ways in which dominant Western cultural and spiritual forms sustain narrow conceptions of justice (Celermajer *et al.*, 2021; Santiago-Ávila & Lynn, 2020).

3.4.5.3 Power dynamics and politics of use

The appearance and disappearance of trade-offs and synergies, and the ways in which they are responded to and negotiated upon are not politically neutral. Social relations of power expressed through institutions, the position of different users in the society, and the language adopted to characterize trends in the use of wild species are crucial to understanding trade-offs and synergies. There is tremendous scope to comprehensively articulate the implications of power for sustainable use when it is under pressure from negative trade-offs, especially within a rapidly changing social-ecological context of the wild species (see similar arguments in Boonstra, 2016; Crépin, Biggs, Polasky, Troell, & de Zeeuw, 2012; Kull *et al.*, 2018, 2018; Nayak *et al.*, 2016). Important questions to further examine trade-off and synergy issues in sustainable use include: (i) What can be gained by assessing who wins and who loses in the context of changes in sustainable use of wild species and its emerging trends under the influence of multiple trade-offs? (ii) Is it possible to better assess the chances that a wild species use regime may be deliberately steered by some towards or away from other users? Such questions help to understand that sustainable use can benefit some and adversely impact others (see Armitage, Marschke, & van Tuyen, 2011; Ho, Ross, & Coutts, 2015).

Divergent views on how a wild species use regime should be managed, who should benefit and who gets to decide on the essential features of the use system, and what needs to be done, are crucial questions with important consequences for how to respond to trade-offs and manage possible synergies. This is a highly context-specific issue, and no silver bullet exists. "What Works" in one context may be completely different in another. Further, this will require careful assessment of the dynamics associated with what Lebel *et al.* (2005) have termed as the "politics of scale" with attention to "politics of position" and "politics of place", and this construct can be well placed in the analysis of trade-offs and synergies around sustainable use of wild species. Reid *et al.* (2006) adds to this view by highlighting the importance of user perspectives in problem formulation and analysis, and user knowledge to deal with governance and management issues. Users' own views of their situation reflect a rather different narrative and reality and failure to account for these diverse perspectives that emerge at different scales and from different users and actors can potentially restrict ability to deal with trade-offs and achieve sustainable use of wild species (Andrachuk & Armitage, 2015; Barron, Hartman, & Hagemann, 2020; Narayan, D., *et al.*, 2001; Narayan, D., R. Patel, K. Schafft, A. Rademacher and S. Koch-Schulte., 2001; Nayak & Berkes, 2010). Berkes (2002) highlights numerous examples where higher scale perspectives and practices exert an influence over or dominate lower scale realities, including through centralized decision-making, limited acceptance of alternative systems of knowledge in formal decision-making, nationalization of resources, influence of national and international markets, and top-down development policies and projects. These issues have significant connection with questions about trade-offs and synergy between and across practices and uses of wild species.

3.4.5.4 Governing trade-offs and synergies for sustainable use

What can be done when the outcomes of multiple, cross-cutting trade-offs between uses and practices become untenable for achieving sustainability of wild species, and when possible, synergies between and among use regimes and practices are not readily available? What approach could be useful when unresolved trade-offs have the potential to become stubborn and act as wicked problems, and configuring innovative synergies becomes a challenge? The question of adopting a governance approach to address these situations becomes important. Kooiman *et al.* (2005, p. 7) define governance as "the whole of interactions taken to solve societal problems and to create societal opportunities; including the formulation and application of principles guiding those interactions and care for institutions that enable and control them." According to this view, governance is qualitatively different from the related task of management in directing societal and environmental processes. It adds dimensions that are absent in a hands-on management approach. 'Interactive governance' emphasizes solving societal problems and creating societal opportunities through interactions among actors (Kooiman, J., Bavinck, M., Chuenpagdee, R., Mahon, R., & Pullin, R., 2008). The emphasis on 'interactions' constitutes the main innovation that fits appropriately with the need for responding to the trade-off and synergy related questions outlined at the beginning of this sub-section. IPBES Glossary (2021) adds rules, norms and actions as crucial elements of governance that can help structure, sustain, and regulate trade-offs and synergies. These multiple elements of governance help ensure dynamic problem-solving abilities based on values, principles, institutions and practices.

Debates around sustainable use may trigger the need for biologically informed management and use targets that require an adaptive governance response (Selkoe *et al.*, 2015). Here, governance refers to the "interrelated and increasingly integrated system of formal and informal rules, rule-making systems, and actor-networks at all levels of human society (from local to global) that are set up to steer societies toward preventing, mitigating, and adapting to global and local environmental change" (Biermann *et al.*, 2009). Social and ecological processes, such as use regimes of wild species, influence and are influenced by governance arrangements in which social outcomes remain contingent upon ecological dynamics and vice-versa (Dale *et al.*, 2000; Waltner-Toews & Kay, 2005). These interacting influences are very visible, for example, in section 3.3.5 regarding the dynamics in non-extractive use and governance, social, and ecological dimension of recreational tourism. As explored in section 3.3.4 on logging, responses of social agents (users) in a given system to ecological change (wild species) have a direct bearing on outcomes (quality of life) (Following Lade, Tavoni, Levin, & Schlüter, 2013). In this respect, aggregated informal responses or coping strategies of local users to the shortage of wild species are important drivers of natural resource depletions, but often overlooked in the policy development of the natural resource management (Ehara *et al.*, 2018). These complex dynamics are visible across sections 3.3.3 and 3.3.5, for example, in relation to the interplay between the harvesting of wild meat for subsistence and protection of livestock, and the establishment of national parks in low-income countries throughout Africa to generate revenue.

Both ecological variables (e.g., biodiversity, biogeochemical cycling, hydrological processes) as well as social variables influencing sustainable use, including human agency, social relations of power, institutions and rules that influence human behavior need to be assessed. As well, humans (users and other agents) both produce unsustainable use regimes and simultaneously adapt to them. Here the focus of governance will be on navigating or adapting, but in other cases the focus will be on steering towards more fundamental social transformation to avoid unsustainable use regimes under the influence of undesirable trade-offs and ensure stronger synergies between uses and practices (see Chapter 5 and 6).

3.5 KNOWLEDGE GAPS

There is an increasing tendency today to shift the focus away from sustainable use of wild species; whereas the emphasis is to view biodiversity conservation and sustainable use through the lens of ecosystem functioning and its capacity to produce ecosystem goods and services (Heywood, 2017). Therefore, it is very challenging to compile knowledge gaps on sustainable use of wild species as there is lack of consistency among worldwide databases to quantify the harvesting and use of wild species by people in different countries across the world. This happens because different countries and organizations have different accounting methodologies, making the merging of different datasets a huge challenge. Major knowledge gaps in the sustainable use of wild species are summarized here.

(i) Across all practices, and especially in global fishing, existing data and reporting do not differentiate adequately between wild and non-wild species. Explained most explicitly in sections 3.2 (global overview), 3.3.1 (fishing), global indicators and data reported by the Food and Agriculture Organization of the United Nations and other agencies do not separate out wild and aquaculture, wild and farmed, wild and plantation, or wild and domesticated species when calculating global or regional off-takes. This makes it almost impossible to accurately assess and report on status and trends in sustainable use of wild species. There is vast legacy of available data on species taxonomy, conservation or economic value related to trade and markets rather than specifically on use as defined in the assessment. In addition, most of the datasets available lack detailed information on practices and uses of utilized and non-utilized species that challenges to make comparative account of population trends.

(ii) Knowledge gap in status of taxonomic groups and their uses at different levels and scales. Information is available on the conservation status of vertebrates, particularly with regard to mammals and birds, to a lesser extent with amphibians and fish including demersal fish; however knowledge on conservation status and use is severely lacking for invertebrates (insects), fungi and microbial species (Coleman *et al.*, 2019; Naranjo-Ortiz & Gabaldón, 2019; Willis, 2018), and in some taxa, especially invertebrates and fungi, there are still thousands of species yet to be described and being named. The knowledge gap also includes widely used and internationally traded species, for example porcini mushrooms (*Boletus* spp.).

Marine species are especially susceptible to exploitation. However, the status of half of the world's fisheries, largely from Southeast Asia, is not scientifically assessed (Costello *et al.*, 2012). We know less about inland fisheries than marine fisheries. Marine mammals are especially susceptible to exploitation due to low reproductive rates and the many other threats they face, including noise pollution and climate change (Perrin, 2009).

With regards to insects, fungi and microbes, insufficient taxonomic information makes it difficult to assess the sustainability of their use, and more generally knowledge on their roles in the supply of nature's contributions to people is limited (Kassas, 2002). For example, it is believed that more than 90% of species remain unknown to science out of 148,000 species of fungi that have been scientifically identified (Antonelli *et al.*, 2020). Sustainability of wild algae, fungi and plants harvesting is challenged by many factors and comprises interlinked dimensions such as socio-cultural, economic and political (Ghimire, 2008). Similarly, the sustainable management of medicinal trees requires knowledge on how different species respond to different harvesting techniques (Delvaux, Sinsin, Darchambeau, & Van Damme, 2009). As discussed in section 3.3.3.3.3 invertebrates provide an important source of nutrition in some areas, but data are missing on the sustainability or unsustainability of the gathering of edible insects. Overexploitation probably only concerns some species, but insects and fungi (sections 3.3.2.1; 3.3.3.2.3) on the whole are vulnerable due to the destruction of their habitats, to pesticides and other pollution, and to climate change (Arnold van Huis *et al.*, 2013).

Another limitation of indicators of sustainable use is related to spatial scales. Not all populations, taxa, systems and regions are equally or adequately represented in the scientific literature, meaning that while it is possible to assess the available knowledge, it is not actually possible to assess the sustainability of use. At the global level, there is a lack of pertinent data for many species of whales and seals, and the polar bear *(Ursus maritimus)* in the Arctic (Tierney *et al.*, 2014) and for many small-scale fisheries in tropical developing countries, such as in Africa, Asia and South America (see small-scale fisheries section). There is a lack of data on how many species in each vertebrate class are used and how much is harvested. For example, data on harvested Arctic species are biased towards marine mammal and marine fish populations, and this could mask declines in some seabird colonies that are over-harvested (Tierney *et al.*, 2014). Relatedly, many of the conservation models, protocols, procedures, monitoring and assessments are based on experience of animals, notably mammals and birds, and do not necessarily apply to plants, invertebrates or fungi (Heywood, 2017).

(iii) Life histories and stocks of marine fish species not well understood. In most fisheries, there exist large gaps in understanding of life histories for many marine fish species, information on total cumulative anthropogenic levels of fishery removals from an individual population, knowledge of the conservation status of individual populations, and deficits in monitoring, including in data collection protocols, observer coverage rates, and sufficient time-series to detect the response in absolute population abundance of long-lived species to this anthropogenic mortality source (Gilman *et al.*, 2014, 2020; Lewison, Crowder, Read, & Freeman, 2004b; Musick, 1999). Status of fish stocks of both large- and small-scale fishing is little understood for those countries and regions where fishing management intensity is low. Further, there is data of status and trends individual fish stocks for IPBES regions such as Europe (e.g., https://www.eumofa.eu/) and North America, whereas data for other IPBES regions are missing.

(iv) Knowledge gap in direct and collateral sources of fishing mortality on associated and dependent species. While there is increasing understanding of the status of stocks of principal market species of marine capture fisheries, albeit still incomplete especially in low-income countries, there remains a very large gap in knowledge of the effects of direct and collateral sources of fishing mortality on associated and dependent species including fecund species. For example, rare-event bycatch of species such as toothed whales and some pelagic sharks are unmonitored in most fisheries, there is a lack of knowledge of which populations are captured in individual fisheries, and as a result of these data quality constraints, extremely limited understanding of the sustainability of the 'use' of these wild species. For instance, 47 of 68 fisheries that catch marine resources managed by regional fisheries management organizations have no observer coverage (Gilman *et al.*, 2014) for the vast majority of the ca. 4.6 million fishing vessels globally, information on non-retained catch is non-existent, and information on retained catch only is available in some cases. While a target stock of a relatively productive species may be determined to be sustainable when assessed against various standards, the sustainability of the fishery and when assessed against impacts on incidentally captured species is very often unknown. Stock assessments which do not incorporate recreational fishing do not provide accurate assessments of global uptake and fish mortality.

(v) Data gaps on sustainable use of wild species and their monitoring regarding small-scale fisheries, inland fisheries, marine and freshwater fisheries, and reef fisheries. One of the major challenges or data gaps to properly assess sustainable use of wild species, especially regarding small-scale fisheries and inland fisheries in tropical developing countries consists in the lack of long temporal series of data on resource use. Most of the small-scale fisheries worldwide show a chronic lack of monitoring data on time series of landings, fishing effort, biology of exploited species, among other relevant fisheries indicators (Welcomme, 2011). Similarly, there is no reliable information on value or number and diversity of sustainability of marine and freshwater ornamental fishery, and many species of reef fishes lack biological and ecological information. This indicates that conservation status of almost half of the species is still unknown (SOTWP, 2016).

This lack of data precludes a proper assessment of the sustainability of most small-scale fisheries and inland fisheries. Furthermore, those indicators based on stock dynamics or population parameters, which have been widely applied in industrial fisheries, may not be suitable to complex, multispecies small-scale fisheries, or data needed to calculate these indicators cannot be gathered on a cost-effective and timely manner to inform policy intervention in many small-scale fisheries and inland fisheries. However, these limitations have been successfully addressed, in the context of small-scale fisheries, by studies adopting a scientific approach to record and analyze local or indigenous knowledge held by small-scale fishers on resource use over broad temporal scales (see section 3.3.1.3.2).

(vi) Research gap in gathering. Estimates on the number wild plant species that are used across different regions are unclear, despite documentation from (SOTWP, 2020) and (FOC, 2020). Also, there are limited information on wild species used as food, and these come mainly from ethnological or ecological inventories. As a global phenomenon, urban gathering that promotes positive cultural, ecological, economic and health outcomes research has received little scholarly attention and due emphasis has not been equally given in all regions of the globe. For example, 70% of the studies are from the Americas, Europe and Central Asia, 20% are from Africa, and the remaining are from Asia and the Pacific based on literature search retrieved for this assessment. Recently, an emerging gap has been in high demand of collection of recently described new species or rare species when their type localities were published in particular by specialized collectors. For this reason, an increasing number of scientists warn against publishing type localities (Lindenmayer & Scheele, 2017b); and the sustainability of this form of consumer-driven use is unclear.

(vii) No data for global sale of cut flowers from wild and cultivated conditions. Cut flower or foliage of bromeliads, or ornamental plants like aloe and orchids share global market and these plant species are either gathered from cultivated or wild sources. But no data was available at the time of this assessment on the share of global market sales from wild vs cultivated plants.

(viii) Gaps in *ex situ* conservation of wild plant species. Botanic gardens gather live plant species from wild for conservation purpose, however, those botanical collections have focused mainly in the temperate parts of the world. For example, the PlantSearch database hosted by Botanic Gardens Conservation International indicates that 107,340 accepted species grow in botanic garden collections, representing 31% of vascular plant species. However, 93% of these species are held in temperate parts of the world. As a result, a temperate species has a 60% chance of being cultivated within the botanic garden network, whereas a tropical species has only a 25% chance. Similarly, the diversity of crop wild relatives is poorly represented in gene banks. For example, there are over 78,000 accessions representing about 688 species of crop wild relatives in gene banks, and over 70% of taxa are recommended as high priority for gathering so as to improve their representation in gene banks. However, gaps in gathering occur in the Mediterranean and the near East, Western and Southern Europe, Southeast and East Asia, and South America **(Figure 3.45)**.

(viii) Identification gaps in taxonomic groups of terrestrial animal harvesting. Some groups of terrestrial animals harvested mainly for trade lack proper identification. For example, more than 50% of all traded individuals of reptiles had no species-specific identification, and this makes implementation of species-based regulations ineffective. Further, scientific studies suggest that consumption of *Didelphis marsupialis*, a species of undeniable cultural significance for local communities in Latin America, but carrying a reservoir of parasites that cause severe diseases, should be the subject of further study.

(ix) Insufficient information on recreation from green hunting. Green hunting that takes place with the help of tranquilizer dart guns is cheaper and less harmful compared to traditional hunting (section 3.3.3.4.2. However, green hunting is as of yet not a significant recreational activity. There exists insufficient information on the status, trends and/or impact of the activity with regards to its potential impact on sustainable use of terrestrial animal harvesting from wild.

(x) Gap of trade of exotic pet animal species under the Convention on International Trade in Endangered Species of Wild Fauna and Flora list. Many wild animal species have been unsustainably traded to supply the international pet markets for natural breeding purpose, including rare and endemic species that are most threatened. Even with existing international regulations, the majority of species in exotic pet trade are not protected under the Convention on International Trade in Endangered Species of Wild Fauna and Flora, therefore, leaving international trade mostly unregulated and unmonitored of threatened species (Janssen & Shepherd, 2018) (section 3.3.3).

(xii) Inadequate available information on wild species informal and formal trade, and consumption. Wild species are traded in informal and formal markets. Much of this trade goes unrecorded and is difficult to monitor. A complex and nuanced temporal association between the illegal and legal wild species trades exist (Tittensor *et al.*, 2020). The gap is so great that in many cases the phrase, "we do not know what we do not know" applies. Cases where it is known that data are lacking include tropical fish for the aquarium trade, freshwater turtles and tortoises for terraria, recreational fishing (including catch and release) and spearfishing, amphibians, and reptiles (Alves, Rosa, *et al.*, 2013; Castello, McGrath, & Beck, 2011; Costello *et al.*, 2012; Schlaepfer, Hoover, & Dodd, 2005). Similarly, insects, especially butterflies and beetles, are harvested and traded all over the world, but there are few data about this exploitation and trade (Alan L. Yen, 2009). Further, the consumer-driven harvest of live specimens may have benefits for local peoples' economically, however, sustainability of use is unclear.

Wild meat harvest and trade are often excluded from official statistics (Pangau-Adam *et al.*, 2012). Overall, there is much less information available on wild meat harvest in the Asian tropics, especially outside Borneo (Swamy & Pinedo-Vasquez, 2014). A conspicuous knowledge gap concerning the causes of lion mortality has been identified, and this requires knowledge of both the existing population size and its dynamics over time and space (fecundity and mortality) (Macdonald *et al.*, 2017). In addition, where markets in such species are monitored, often it is not clear whether sources are wild or domesticated.

Existing data are available mainly for timber species traded in the global market (FAO, 2018a), but timber from illegal logging activities used within producing countries as well as across the transboundary are not available (Chaudhary *et al.*, 2016).

(xii) Knowledge gap in logging. Timbers are supplied to the markets; however, it is unclear to estimate which come from legal or illegal sources as well as differentiate timber from wild vs plantation sources.

(xiii) Knowledge gap in non-extractive practice and uses. Assessment of knowledge gap in non-extractive practice and use is challenging as the non-extractive use of nature often does not include species described at a species level, but frequently they appear as part of a functional group (e.g., trees in urban green spaces) or in terms of multifunctional landscapes (e.g., worship of sacred groves). Further research is especially needed to clarify the benefits of living in nature and focus on ecosystem elements. For example, in commercial wildlife watching, an increasing number of wild species such as megafauna and 'charismatic' wild species are integrated into tourism operations. Megafauna are well studied taxa of animals, whereas there is a lack of research on the impacts of tourism on the lesser fauna, e.g., ground-dwelling mammals, small reptiles, insects, etc. (Wolf *et al.*, 2019).

The literature on the non-extractive use of wild species for medicine and hygiene shows many positive benefits on human individuals, but there is an absence of research on the effects of wild species on human community health (Nesbitt *et al.*, 2017). There is almost no information on the global or regional trends in the non-extractive use of wild species for human health. No research has looked at the sustained, long-term effects of nature-based therapies (Rajoo *et al.*, 2020).

(xiv) Gaps in inter-practice trade-offs and synergy. It is well known that different practices interact themselves and are connected with each other; however, the knowledge gap involves the lack of inter-practice trade-offs and synergies, such as between and among fishing, gathering, terrestrial animal harvesting, logging, and non-extractive practices across global, regional, national and local policy and program.

(xv) Lack of critical linkages between nature's contributions to people and quality of life and benefit gaps. There has been broad uptake of the critical linkages between nature's contributions to people and quality of life. However, knowledge gap exists on the status of species and nature's contribution to people linked to specific ecosystem functions, and interrelationships between gender equality, nature and nature's contribution to people (IPBES, 2019). Therefore, enhanced attention is needed to develop specific variables and indicators to understand the multiple intricate ways in which peoples' well-being / quality of life and nature's contributions to people influence each other in a two-way feedback-oriented process (Chaplin-Kramer *et al.*, 2019; Diaz, Demissew, Joly, Lonsdale, & Larigauderie, 2015; IPBES, 2019). It is also important to ascertain that such a connection draws on integration of indigenous and local knowledge and their effective participation pays judicious attention to scientific knowledge and strengthens linkage between nature and nature's contribution to people ((Diaz *et al.*, 2015).

There are important methodological limitations to many of the studies exploring nature-based therapy or the presence of wild species on human health. Furthermore, the majority of the studies are correlative and the involvement of medical professionals is encouraged, as well as an increased diversity of study participants (Rajoo *et al.*, 2020; Sandifer *et al.*, 2015). The causal mechanisms that underlie the benefits people receive from health-based use of wild species is underexplored (Sandifer *et al.*, 2015). Currently, there is limited evidence for environmental microbial exposure boosting human immune system response and no causal evidence for the phytoncides hypothesis was identified.

Another important aspect to consider is that there is a poor understanding of how biodiversity affects people's well-being and health through cultural pathways, and how that is being affected by changes in the status and trends in sustainable use. A better understanding by linking biodiversity change with human culture values, well-being, and health might be profoundly important for biodiversity conservation and public health (N. E. Clark *et al.*, 2014). It is believed that diversity of positive values is important for countering negative values and support conservation action when needed.

(xvi) Inadequate economic valuation of wild species. A considerable body of valuation studies focuses on the economics of nature's contributions to people at the global scale (e.g., carbon stocks and flows) delivered to people outside the countries where natural ecosystems and wild species occur (IPBES, 2019). However, the societal values of the gathering and use of wild species in local markets have not been properly addressed. Wild species are an integral component of ecosystems, and the value they provide in terms of services should be a standard part of ecosystem assessments (Puri, Yadav, & Joshi, 2019). Though, it needs to be recognized that there is no distinct division between wild (unmanaged) biodiversity and human managed biodiversity (Tisdell, 2015). A comprehensive assessment of the contributions (current or potential) of wild species in protected areas (terrestrial and marine), such as watching of animals, recreational tourism, recreational fishing, trophy hunting, among others to promote social and economic sustainability, besides ecological sustainability is lacking. Further, several species of frogs in Africa, including endemic species (*Conraua* sp, *Trichobatrachus* sp. and *Astylosternus* sp.) are mainly harvested from wild used for local consumption and local trading; however, assessments of the value chains are poor, especially Central Africa regions (See section 3.3.3.3.3).

(xvii) Knowledge gap on global scale of sustainable use of wild species among indigenous people and local communities. The importance of wild species that contribute to livelihood strategies, in particular for indigenous people and local communities is well recognized. However, little information exist in the available global indicator sets to comprehensively quantify the spatial and temporal scales of sustainable use of wild species occurring specifically in indigenous people and local communities across the globe; and the United Nations are aware of this gap.

(xviii) Knowledge gap on quality assurance, safety and efficacy to assess traditional medicine. Wild species have been used in traditional medicinal practice for millennia. To control quality and to ensure safety and efficacy in production of traditional medicines is difficult. The World Health Organization has produced a series of technical documents in this field, including publications on good agricultural and collection practices and good manufacturing practices, along with other technical support, to assist with standardization and creation of high-quality products (WHO, 2013). The World Health Organization urges member states to cooperate with each other and to share knowledge while working to strengthen communication between conventional and traditional practitioners. Evaluation of quality, safety and efficacy based on research is needed to improve approaches to assessment of traditional medicines, a situation made difficult to remedy in light of historically inadequate public and private funding to address this growing concern (WHO, 2013).

(xix) Insufficient bridging of indigenous and science-based knowledge. The incorporation of multiple types of knowledge (e.g., science, indigenous knowledge, traditional ecological knowledge) is especially critical for the sustainable use of wild species, which can strengthen the evidence-base for policy advice, decision making, and environmental management (R. Hill *et al.*, 2017). While the benefits of incorporating multiple types of knowledge in environmental research and management are many, successfully doing so has remained a challenge. In response there have been a number of recent reviews that have sought to better understand the bridging of indigenous, local, and science-based knowledge (Barron *et al.*, 2015; Berkes, 2010; Berkes & Berkes, 2009). Yet there continues to be a need for methods, models, and approaches for integrative work (Barron *et al.*, 2020; R. Hill *et al.*, 2017). This approach seeks to examine the extent, range, and nature of the published literature (i.e., peer-reviewed and grey) that integrates and/or includes indigenous, local, and science-based knowledge in sustainable use of wild species research, monitoring, or management (Alexander, Provencher, Henri, Taylor, & Cooke, 2019).

There is no solid mechanism developed for knowledge transfer from indigenous communities to scientific communities and vice versa, and as discussed in section 3.3.2, in many cases attempts to do so have led to issues of intellectual property and biopiracy (Barron *et al.*, 2015; Berkes & Berkes, 2009; S. Devkota, 2006). Wild species are being used as a rich source of medicine because they produce a host of bioactive molecules, most of which probably evolved as chemical defenses against predation or infection. Wild plant species are chosen for pharmaceutical studies through different methods, one of the methods include ethnobotanical approach, i.e., indigenous uses of plant species based on indigenous and local knowledge that can offer strong clues to the biological activities of those plants (P. Cox & Balick, 1994). There are well-established drugs that were developed after scientists began to analyze the chemical constituents of plants used by indigenous peoples and local communities for medicinal or other biological effects.

3.6 CHALLENGES AND RESEARCH PRIORITIES

3.6.1 Challenges

Major challenges related to status of and trends in sustainable use of wild species have been discussed.

3.6.1.1 Global scale and scope

Fundamental challenges evaluating the role of sustainable use in biodiversity conservation and sustainable economic development pertain to the lack of guiding principles derived from analysis of spatial and temporal applications (Rands *et al.*, 2010; Tierney *et al.*, 2014). The scale and scope of these challenges involving sustainable use across regions and countries is immensely diverse and context specific. Studies that integrate and harmonize information from various sources and programs, where sustainable use has and has not been achieved, are needed to evaluate in order to better understand the likelihood of benefits and costs for both nature and people. However, these datasets, published from a variety of sources, are not sufficient in terms of quantity or quality or both for an assessment of sustainability of harvest. Data need to be integrated and harmonized to evaluate status and trends in global use of many species.

3.6.1.2 Informal trade of wild species

Informal trade of wild species in small quantities that do not enter the national trade or export statistics takes place through informal markets in most developing countries. Informal trade mainly includes subsistence small-scale coastal and freshwater fisheries, terrestrial animal harvesting, gathering of wild foodstuffs, medicinal plants, mushroom, and berry picking (FAO, 2019b). A challenge in such informal and largely unreported trade is that its ecological, economic and social impacts and importance to society remain invisible to decision-makers, hence unlikely to mainstream into policy-making. More research would be desirable to assess informal trade of wild species.

3.6.1.3 Fishing

Among fishing, small-scale fisheries are strongly connected with activities by local communities for their own consumption; and employ over 90 percent of fisheries workforce. Despite their importance, small-scale fisheries around the world are facing major challenges due to large-scale fisheries and increased global development activities as well as climate change. There is lack of data to evaluate the sustainability of small-scale fisheries on catches and measures of exploited stocks (i.e., size, proportion of juveniles caught, among others), especially over broader spatial or temporal scales. These challenges, in many cases, have placed the livelihoods, economy, food security, values and identity, and the viability of small-scale fisheries communities at risk.

3.6.1.4 Gathering

A wide range of organisms are gathered worldwide for meeting a variety of needs (i.e., income, livelihoods, subsistence, social safety, identity), which are also traded in both domestic and international markets. A major challenge is that gathering practices are selective and based on specific organisms/group of organisms (e.g., ornamental fish and coral, orchids, cacti, bromeliads, succulent, palms and bamboos, medicinal plants and other wild algae, fungi and plants, wild biomass energy, edible insects and small terrestrial invertebrates, etc.). As organisms become more popular to harvest because of changing commodity chains or popular fads or trends, overharvesting can occur. The mix of temporal and spatial drivers and their direct effects on wild species are difficult to quantify since these same changes in demand lead to innovations in domestication and synthesizing similar materials.

3.6.1.5 Terrestrial animal harvesting

Throughout history human populations have been engaged in hunting and trapping to meet a range of nutritional, economic, medicinal, cultural and recreational needs. A major challenge is that overhunting, which is taking place at varying degrees of hunting pressure, often results in faunal biomass collapses, mainly through declines of large-bodied species with low intrinsic rates of population increase, especially in Oceania, Africa and Asia. Trophy hunting is currently the subject of intense debate. However, some trophy hunting can produce simultaneous benefits of economic gains, and sustainable wild species exploitation and biodiversity conservation, even though well managed trophy hunting is rarely documented (Coad *et al.*, 2019), and unsustainable hunting is common. Hunting becomes unsustainable when it causes species abundance on a trajectory of ongoing declines.

3.6.1.6 Logging

Harvesting of timber for wood carvings is a challenge because it involves destructive processes, which is not carefully monitored and remain somewhat hidden. Most commercial carving enterprises are based in homes or small production units (Cifor, 2002). In the past, wood carvings were mainly carried out to attain cultural materials, often as symbols of particular cultures or regions.

Tree retention has the potential to reduce impact of logging on forest biodiversity, though determining exact levels that are required to secure long-term viable populations of different species in a natural forest in most cost-efficient conservation measures remains a major challenge for future research (Gustafsson *et al.*, 2010).

Another challenge is that harvesting has long been affected by changing tools and technology i.e., availability of axes, adzes and chisels made of iron, for example, increased both the speed with which wood could be carved and the range of species used. This includes endangered/threatened, for example sandalwood, whose use and trade are restricted by both national and international regulations.

New policy instruments are emerging in some countries, such as in the United States of America, Australia and many European countries to prohibit the sale of illegally harvested wood and wood products. These regulations require operators to provide proof of certification of the identity of the species traded and the origin of their products. However, there is a mismatch between the legislated requirements and the capacity of importers to comply fully because existing methods for documenting species identity (wood anatomy and chemistry) and origin (mostly paper-based documentation, tagging) are insufficient, ambiguous and easily falsifiable (FAO, 2014c). While extensive literatures on the using of DNA analysis for forensic investigations in animal species exist, there is unfortunately a serious lack of information on wild plant species (Iyengar, 2014).

3.6.1.7 Non-extractive uses

In the context of sustainable utilization of nature for economic and other benefits, nature tourism has created a growing demand for 'watching wild species', 'un-spoilt habitat', and 'pristine nature' in combination with high levels of comfort, accessibility and high-quality experiences. The 'flagship' species – most often the megafauna, 'charismatic' mammals and birds, the 'cute and cuddly', dangerous predators and species that are believed to display intelligence, play an important for tourism and recreation practices. The tourists' preference of visiting a pristine natural habitat contributes to new challenges and creates pressure on the ecosystems in general and wild species in particular. Consequently, special attention needs to be paid to the aspects of sustainability in these processes.

3.6.2 Research priorities

An attempt has been made to identify common research priorities for status of and trends in sustainable use of wild species. This analysis is based on the assessment of: (i) key knowledge gaps to achieve global sustainability goals (Mastrángelo *et al.*, 2019); (ii) knowledge gaps and challenges mentioned in this assessment; and (iii) selection of pertinent research questions that would substantially advance the goals of biodiversity conservation and sustainable development (Coleman *et al.*, 2019).

3.6.2.1 Practices and uses

In the practices and uses sectors, key prioritized areas include sustainable practices in fishing, gathering, terrestrial animal harvesting and logging as well as assessment of combined impact of wild species harvesting, fishing and hunting practices leading to habitat and global biodiversity loss. For example, gaps in fishing comprise: (i) amount of freshwater wild species harvested, consumed locally, and traded nationally and internationally; (ii) assessment of conservation status and sustainable small-scale fishery, and economically-important fish for food, live fish trade; and (iii) impact of international trade on fisheries and marine biodiversity, globally and regionally. Similarly, an emerging major challenge for future in logging remains to determine exact levels that are required to secure long-term viable populations of different species, as well as most cost-efficient implementation of these conservation measures (Gustafsson *et al.* 2010). It is estimated that Reduced Impact Logging provides guidelines to reduce environmental impact of logging, but it is unclear what intensity can sometimes result perverse effects; and more research is needed to clarify this type of practice.

3.6.2.2 Nature's contributions to people & human well-being

Some prioritized areas of research include: (i) evaluation of contributions of sustainable use of wild species including urban gathering to nature's contributions to people that play key roles in regional and national scales; (ii) analysis of maximum benefits of nature tourism while minimizing adverse impacts on terrestrial, aquatic and marine ecosystems; (iv) assessment of effective livelihood support programs that meaningfully support nature conservation among marginalized communities and indigenous peoples and local communities; and (v) identification of key factors underlying win-win outcomes for sustainable use of wild species and poverty alleviation.

3.6.2.3 Documenting under-researched taxa

The emphasis should be on taxonomic assessment of under-researched taxa (e.g., invertebrates, insects, fungi, species that are pollinators or pest regulators, species or habitats with cultural value, rare or endemic species that are often viewed as the most important targets for biodiversity conservation) being overlooked due to gaps in data, as well as inadequate enforcement of laws and principles that are particularly missing in the countries action plans.

3.6.2.4 Social norms that affect uses and practices

There is growing interest in how socio-ecological dynamics relate to obtaining interdisciplinary and reliable data in research priorities, such as: (i) social science methods and approaches to obtaining reliable data on scale and patterns of uses of wild species; and (ii) evaluation of social norms (at local, regional and national scales) that affect use and practices including gathering and harvesting, fishing, logging, and hunting & poaching pressure.

3.6.2.5 Integrating indigenous local knowledge

Indigenous and local knowledge research is increasingly being used to generate more accurate data on species trends, non-iconic species data and geospatially relevant data using technology. Participatory monitoring of use of wild species in close collaboration with local resource users can provide large amounts of reliable and much needed data to inform policy and management approaches in data-poor social-ecological systems. Biocultural approaches are being adopted by governments to policy that recognize both indigenous people and local communities' territorial management practices and customary governance, thus countering the drivers of unsustainable resource use and offering alternative conceptualizations of the interrelations between people and nature (Brondízio *et al.*, 2021).

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Артеага В. (2019, November 1). Старая песня про зубра. *Старая Песня Про Зубра.*

The thematic assessment report on

THE SUSTAINABLE USE OF WILD SPECIES

SUMMARY FOR POLICYMAKERS

SUMMARY FOR POLICYMAKERS OF THE IPBES THEMATIC ASSESSMENT REPORT ON THE SUSTAINABLE USE OF WILD SPECIES

ISBN No: 978-3-947851-30-0

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SUGGESTED CITATION

IPBES (2022). Summary for Policymakers of the Thematic Assessment Report on the Sustainable Use of Wild Species of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. Fromentin, J.M., Emery, M.R., Donaldson, J., Danner, M.C., Hallosserie, A., Kieling, D., Balachander, G., Barron, E.S., Chaudhary, R.P., Gasalla, M., Halmy, M., Hicks, C., Park, M.S., Parlee, B., Rice, J., Ticktin, T., and Tittensor, D. (eds.). IPBES secretariat, Bonn, Germany. https://doi.org/10.5281/zenodo.6425599

MEMBERS OF THE MANAGEMENT COMMITTEE WHO PROVIDED GUIDANCE FOR THE PRODUCTION OF THIS ASSESSMENT

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The Assessment of the Sustainable Use of Wild Species was made possible thanks to many generous contributions received during the production of the assessment including non-earmarked contributions to the IPBES trust fund from Governments (Australia, Austria, Belgium, Bulgaria, Canada, Chile, China, Denmark, Estonia, European Union, Finland, France, Germany, Japan, Latvia, Luxembourg, Netherlands, New Zealand, Norway, Republic of Korea, Slovakia, Spain, Sweden, Switzerland, United Kingdom and United States of America); earmarked contributions to the IPBES trust fund toward the Sustainable Use of Wild Species Assessment (France – French Office for Biodiversity); and in-kind contributions targeted at the Sustainable Use of Wild Species Assessment, including from the French Foundation for Research on Biodiversity (FRB) and the French Office for Biodiversity (OFB) which co-hosted the technical support unit. All donors to the trust funds are listed on the IPBES web site: www.ipbes.net/donors