Food and Agriculture
Organization of the
United Nations

COMMISSION ON
GENETIC RESOURCES
FOR FOOD AND
AGRICULTURE

THEMATIC STUDY

BIODIVERSITY FOR FOOD AND AGRICULTURE AND ECOSYSTEM SERVICES

Thematic Study for *The State of the World's Biodiversity for Food and Agriculture*

BIODIVERSITY FOR FOOD AND AGRICULTURE AND ECOSYSTEM SERVICES

Thematic Study for *The State of the World's Biodiversity for Food and Agriculture*

COMMISSION ON GENETIC RESOURCES FOR FOOD AND AGRICULTURE
FOOD AND AGRICULTURE ORGANIZATION OF THE UNITED NATIONS
Rome, 2020

Required citation:
FAO. 2020. *Biodiversity for food and agriculture and ecosystem services – Thematic Study for The State of the World's Biodiversity for Food and Agriculture.* Rome. https://doi.org/10.4060/cb0649en

The designations employed and the presentation of material in this information product do not imply the expression of any opinion whatsoever on the part of the Food and Agriculture Organization of the United Nations (FAO) concerning the legal or development status of any country, territory, city or area or of its authorities, or concerning the delimitation of its frontiers or boundaries. The mention of specific companies or products of manufacturers, whether or not these have been patented, does not imply that these have been endorsed or recommended by FAO in preference to others of a similar nature that are not mentioned.

The views expressed in this information product are those of the author(s) and do not necessarily reflect the views or policies of FAO.

ISBN 978-92-5-133168-2
© FAO, 2020

Some rights reserved. This work is made available under the Creative Commons Attribution-NonCommercial-ShareAlike 3.0 IGO licence (CC BY-NC-SA 3.0 IGO; https://creativecommons.org/licenses/by-nc-sa/3.0/igo/legalcode).

Under the terms of this licence, this work may be copied, redistributed and adapted for non-commercial purposes, provided that the work is appropriately cited. In any use of this work, there should be no suggestion that FAO endorses any specific organization, products or services. The use of the FAO logo is not permitted. If the work is adapted, then it must be licensed under the same or equivalent Creative Commons licence. If a translation of this work is created, it must include the following disclaimer along with the required citation: "This translation was not created by the Food and Agriculture Organization of the United Nations (FAO). FAO is not responsible for the content or accuracy of this translation. The original [Language] edition shall be the authoritative edition."

Disputes arising under the licence that cannot be settled amicably will be resolved by mediation and arbitration as described in Article 8 of the licence except as otherwise provided herein. The applicable mediation rules will be the mediation rules of the World Intellectual Property Organization http://www.wipo.int/amc/en/mediation/rules and any arbitration will be conducted in accordance with the Arbitration Rules of the United Nations Commission on International Trade Law (UNCITRAL).

Third-party materials. Users wishing to reuse material from this work that is attributed to a third party, such as tables, figures or images, are responsible for determining whether permission is needed for that reuse and for obtaining permission from the copyright holder. The risk of claims resulting from infringement of any third-party-owned component in the work rests solely with the user.

Sales, rights and licensing. FAO information products are available on the FAO website (www.fao.org/publications) and can be purchased through publications-sales@fao.org. Requests for commercial use should be submitted via: www.fao.org/contact-us/licence-request. Queries regarding rights and licensing should be submitted to: copyright@fao.org.

Contents

Tables

Acknowledgements

The study was prepared by Dafydd Pilling, with contributions from Kim-Anh Tempelman and Julie Bélanger, in FAO's Secretariat of the Commission on Genetic Resources for Food and Agriculture. Thanks are due to the many reviewers who provided comments on the study, including (all FAO unless otherwise indicated) Devin Bartley, Caterina Batello, Agnès Bernis-Fonteneau, Eric Blanchard (Institut de Recherche pour le Développement, France), Paul Boettcher, Stefano Diulgheroff, Simon Funge-Smith, Bonnie Furman, Rodolphe Gozlan (Centre de coopération internationale en recherche agronomique pour le développement, France), Irene Hoffmann, Jarkko Koskela, Giulia Muir, Arshiya Noorani, Manuel Pomar Cloquell, Marcela Portocarrero-Aya, Mariam Rahmanian, Suzanne Redfern, Kristina Rodina, Beate Scherf, Vladimir Shlevkov-Pronskiy, Pablo Tittonell (Instituto Nacional de Tecnología Agropecuaria, Argentina), Miriam Widmer and Liesel Wiese.

Executive summary

This study provides a short overview of the contributions that biodiversity for food and agriculture (BFA) makes to the delivery of ecosystem services. It is intended to complement material provided in the country reports submitted as inputs to the report on *The State of the World's Biodiversity for Food and Agriculture* (SoW-BFA). BFA is a subcategory of biodiversity taken for the purposes of the SoW-BFA (and in this thematic study) to correspond to "the variety and variability of animals, plants and micro-organisms at the genetic, species and ecosystem levels that sustain the ecosystem structures, functions and processes in and around production systems, and that provide food and non-food agricultural products." The study considers a range of ecosystem services across the "provisioning", "regulating", "supporting", "habitat" and "cultural" categories.

The examples presented in the various sections of the document illustrate the wide range of ecosystem services provided by BFA. They also show that the benefits that a given food and agricultural production unit (i.e. farm, fish farm, forest stand, fishery or livestock holding) gains from biodiversity generally come both from within and from outside the production unit. These services are supplied, and made more resilient, by a diverse range of interacting components of biodiversity, often including those that are used in or associated with other production units (including those in other sectors of food and agriculture) and those found on land or in waters not used for food and agriculture. It follows, similarly, that flows of benefits to one production unit can be disrupted by events, including the effects of human management or mismanagement, in others and in the wider landscape or seascape. These interactions point to the need for a more integrated management of production units and their surroundings, at least at landscape (or seascape) scale. The examples also show that the biodiversity present in and around food and agricultural production systems often provides ecosystem services whose benefits are felt far beyond the food and agriculture sector (and in some cases far away in geographical terms). While there are potential "win–win" scenarios in the management of BFA for ecosystem services, there will inevitably be cases where there are trade-offs in terms of who benefits or loses out. Efforts need to be made to develop equitable ways of addressing such issues, as well as to facilitate cooperation in the implementation of mutually beneficial actions.

Assessing the significance of diversity *per se* to the capacity of BFA to supply ecosystem services is often difficult. However, experimental evidence and theoretical considerations suggest that biological communities that are more diverse at species or within-species level will often be more effective or more resilient suppliers of ecosystem services. Diversity also provides the basis for adapting production systems to future challenges to the supply of ecosystem services.

1. Introduction

This study provides a short overview of the contributions that biodiversity for food and agriculture (BFA) makes to the delivery of ecosystem services. It is intended to complement material provided in the country reports submitted as inputs to the report on *The State of the World's Biodiversity for Food and Agriculture* (SoW-BFA).[1]

1.1 Key concept

BFA is a subcategory of biodiversity taken for the purposes of the SoW-BFA (and in this thematic study) to correspond to "the variety and variability of animals, plants and micro-organisms at the genetic, species and ecosystem levels that sustain the ecosystem structures, functions and processes in and around production systems, and that provide food and non-food agricultural products." Production systems are here taken to include those in the crop, livestock, forest, fisheries and aquaculture sectors. BFA includes plant, animal and aquatic genetic resources for food and agriculture, forest genetic resources, associated biodiversity[2] and wild foods.

The concept of ecosystems as suppliers of "services" that contribute to human well-being has gained widespread currency in recent decades. Obtaining information on the role of BFA in the supply of such services was a major objective of the country-reporting process for the SoW-BFA, which followed the Millennium Ecosystem Assessment (MEA, 2005a) in defining ecosystem services as the "the benefits humans derive from ecosystems." Such services have been categorized in various ways by different authors. For example, the Millennium Ecosystem Assessment identified the following four categories: provisioning services – "the products obtained from ecosystems"; regulating services – "benefits obtained from the regulation of ecosystem processes"; cultural services – the "nonmaterial benefits people obtain from ecosystems through spiritual enrichment, cognitive development, reflection, recreation, and aesthetic experiences"; and supporting services – services "that are necessary for the production of all other ecosystem services" (ibid.). In contrast, the framework used by the Economics of Ecosystems and Biodiversity (TEEB) initiative does not treat supporting services as a separate category, but rather as a subset of the ecological processes that underlie the delivery of other services (TEEB, 2010). TEEB, however, distinguishes a separate category, habitat services, defined as services that "provide living space for resident and migratory species." In their reports for the SoW-BFA countries were invited to focus particularly on regulating and supporting services. The present study aims to cover all categories of ecosystem services. For the sake of simplicity of presentation, services are grouped into three main groups: provisioning; regulating, supporting and habitat; and cultural. Lower-level categories are based largely on those used by TEEB (TEEB, 2010).

---

[1] The study was first drafted in 2016 in connection with the preparation of a draft version of the SoW-BFA that was presented to the Sixteenth Regular Session of the Commission on Genetic Resources for Food and Agriculture. It was revised in 2018 and early 2019 in connection with the finalization of the SoW-BFA. It therefore does not cite literature published after February 2019 (other than its companion thematic studies – Dawson *et al.* [2019] and DuVal, Mijatovic and Hodgkin [2019] – and *The State of the World's Aquatic Genetic Resources for food and agiculture* [FAO, 2019], all of which were available in advanced draft form at the end of 2018). Production figures from FAO sources were updated prior to publication to reflect the latest available data as of May 2020.

[2] Associated biodiversity is described in the country-reporting guidelines for the SOW-BFA (FAO, 2013a) as "those species of importance to ecosystem function, for example, through pollination, control of plant, animal and aquatic pests, soil formation and health, water provision and quality, etc."

1.2 Links between biodiversity and the supply of ecosystem services

The capacity of ecosystems to deliver ecosystem services is inextricably linked to biodiversity. In some cases, there is a clear and direct link between a particular species and a given service, for example the provision of a particular type of food by a particular fish, crop or livestock species or the control of a particular crop pest by a particular predator species. However, the presence of any such individual species will depend on ecosystem structures and processes involving vast numbers of other species, linked in numerous ways (e.g. via food webs or habitat creation) and over a variety of time and spatial scales. Many ecosystem services need to be thought of as products of the ecosystem as a whole, for example carbon sequestration or control of water flow and quality by a forest, grassland or coastal ecosystem.

Food and agricultural production systems benefit from a range of ecosystem services generated locally (i.e. in and around the respective systems) and at a greater distance. For example, a crop production system may benefit from the services provided by insect pollinators that live in and around the fields, from the effects of a nearby woodland on the local climate and water supply, and from global climate-regulating services provided by the world's forests, grasslands, oceans and other ecosystems.

As well as benefiting from ecosystem services, food and agricultural production systems also supply them. Production systems are largely defined by their roles in the delivery of provisioning services – most notably in the production of food, but also in the supply of fibres, fuel, timber and a range of other products – and their management typically focuses mainly on these roles. However, the significance of other ecosystem services generated in and around production systems is increasingly being recognized. On the one hand, the supply of provisioning services is underpinned by regulating and supporting services (pollination, nutrient cycling, protection against disasters, etc.). On the other, production systems generate a range of non-provisioning ecosystem services, whose significance often extends far beyond the food and agriculture sector.

Given the scale and diversity involved – cropland, grasslands used for livestock grazing, marine and freshwater ecosystems used for fishing or aquaculture, and managed or harvested forests – it is clear that ecosystems used for food and agriculture (and the biodiversity in and around them) account for a substantial share of the ecosystem services generated on the planet. By the same token, the potential of crop and livestock production, forestry, fisheries and aquaculture to disrupt the delivery of ecosystem services is also enormous. Ensuring that BFA is well managed – used responsibly and sustainably and protected by conservation measures where needed – is vital to the supply of ecosystem services, both to the food and agriculture sector and beyond.

1.3 Scope and objectives of the study

The general significance of ecosystem services to human well-being – including via their contributions to food and agriculture – has been extensively reviewed in other publications, as has the significance of biodiversity in general in the supply of ecosystem services (e.g. MEA, 2005a; TEEB, 2010). This thematic study focuses more specifically on the biodiversity found in and around production systems – in particular on associated biodiversity and wild foods, but also on crops, livestock, forest trees and aquatic species used in aquaculture and targeted by fishers. It aims to provide an overview of the range of ecosystem services to which BFA contributes, the mechanisms involved, the roles played (or potentially played) by particular components of BFA and the significance of diversity *per se* at species or within-species level.

While the focus of the study is on the services provided by BFA, it clearly has to be recognized that production systems and their surroundings harbour species that can have damaging effects on food and agriculture, other socio-economic activities and/or human health. Although some effects of this kind are noted in the text, the study does not attempt to systematically explore all the ways in which plants, animals and micro-organisms can harm humans and disrupt their activities. It is also clear that the use of components of biodiversity to deliver one kind of ecosystem service can disrupt the supply of others (or directly cause "disservices" to human wellbeing and the environment). Negative environmental effects associated with crop and livestock production, forestry, fisheries and aquaculture systems have been extensively reviewed elsewhere (e.g. Steinfeld *et al*., 2006; Gerber *et al*., 2013; Smith *et al*., 2014; Edwards, 2015; Herrero *et al*., 2015; Robb *et al*., 2017; Mateo-Sagasta, Marjani Zadeh and Turral, 2018) and are not revisited in any depth in this study. While some effects of this kind are again noted (and in some cases also the potential role of BFA in reducing them), the study does not provide a detailed analysis of possible trade-offs. It thus does not provide a basis for strategic recommendations about how the components of BFA should be deployed (e.g. the expansion or contraction of particular sectors of production) to maximize overall benefits in terms of the supply of ecosystem services. Discussion of methods for increasing or maintaining flows of ecosystem services from BFA can be found in the companion thematic studies Dawson *et al*. (2019) and DuVal, Mijatovic and Hodgkin (2019).

2. Provisioning services

2.1 Food

The world's food production depends on its terrestrial and aquatic ecosystems. Figures from FAO's statistical database FAOSTAT indicate that as of 2017 approximately 82 percent of the calories in the global human food supply were provided by terrestrial plants, 17 percent by terrestrial animals and 1 percent by aquatic animals and plants.[3] The figures for protein supply were 60 percent from terrestrial plants, 33 percent from terrestrial animals and 7 percent from aquatic animals and plants. Within each of these broad categories, a range of different species – and varieties and breeds within species – are used in food production. A far wider range of species contribute to the functioning of the ecosystems upon which food production depends.

When considering the food-supply figures quoted above, it is important to recall that global averages mask the fact that certain sectors may be extremely important in specific geographical areas or to particular sections of the population: for example, fish in small island developing states and livestock in pastoral communities. Moreover, in addition to calories and protein, food security and good nutrition require adequate access to micronutrients, essential fatty acids and minerals. These are found in varying levels in the various species and populations of plants, animals and micro-organisms used as sources of food and in the products obtained from them.

2.1.1 Terrestrial domesticated animals

The vast majority of animal-source food obtained from terrestrial ecosystems comes from domesticated mammals and birds. According to FAOSTAT figures,[4] game (meat from wild animals) accounted for only 0.6 percent of global terrestrial meat production as of 2018 (although it should be noted that wild foods are generally underreported by countries).

Food production from domesticated animals is dominated by a relatively small number of species. Cattle, sheep, goats, pigs and chickens are sometimes referred to as the "big five" species on account of their major role in food production and their widespread distribution (FAO, 2015a). Viewed purely in terms of production, the "big five" could reasonably be reduced to a "big three". In 2018, Cattle, chickens and pigs together accounted for 88 percent of meat production, cattle for 81 percent of milk production and chickens for 93 percent of egg production. Beyond these three species, the biggest contributions to meat production came from sheep (3 percent), goats (2 percent), turkeys (2 percent), ducks (1 percent) and buffaloes (1 percent). Buffaloes (15 percent), goats (2 percent) and sheep (1 percent) were also relatively major contributors to the global supply of milk. Non-chicken eggs came mainly from ducks and geese.

Again, global figures mask a good deal of regional variation in the importance of particular species. For example, buffaloes rather than cattle are the leading milk producers in South Asia. "Minor" species, such as dromedaries, Bactrian camels, yaks, llamas, alpacas and reindeer play a significant role in various harsh production environments around the world.

Other bird and mammalian species that provide relatively small amounts of food in global terms include those such as horses and donkeys that are used primarily for other purposes, small mammals such as rabbits and (on a more local scale) guinea pigs, and those such as ostriches that are relatively newly domesticated or cater to niche markets. Products

---

[3] FAOSTAT (http://www.fao.org/faostat/en/#home) accessed May 2020.

[4] Unless otherwise indicated, all figures presented in this subsection are based on FAOSTAT data (http://www.fao.org/faostat/en/#home) accessed May 2020.

from domesticated or captive-raised terrestrial animals from taxonomic groups other than birds and mammals represent only a small fraction of global food production. In 2018, global honey production exceeded 1.85 million tonnes and production of land snails was almost 20 000 tonnes. In 2017, honey contributed 2 kcal per person per day to global food supplies.

Below the species level, domesticated animal populations are often subdivided into distinct breeds. Some of these have been developed as single-purpose breeds specialized in producing a specific food product. Others are multipurpose breeds that are good at supplying more than one type of food (e.g. both milk and meat) or can combine food production with other roles such as providing draught power. The other main significance of breed diversity is that it allows production to take place across a wide range of environments. Widely distributed livestock species generally include populations that have become adapted to extremes of climate, terrain, disease exposure and other environmental variables. They also include populations that have been developed to provide maximum output in favourable conditions. As humans' capacity to control production environments has increased, breeds of the latter type have become increasingly widespread.

Food production statistics are generally not broken down beyond the species level and it is therefore difficult to determine the contributions that different breeds or breed categories make to global production. However, some conclusions can be drawn from estimates of the contributions of different production systems.

Pig and poultry production, in particular, is increasingly dominated by specialized "industrial" production systems. MacLeod *et al.* (2013) estimated that, as of 2010, 61 percent of global pig production came from industrial systems, 20 percent from "intermediate" systems and 19 percent from "backyard" systems. The same authors concluded that only 14 percent of egg production and 4 percent of poultry meat production came from backyard production. Specialized layer systems accounted for an estimated 86 percent of egg production and 6 percent of poultry-meat production and specialized broiler systems for 81 percent of poultry-meat production (ibid.). These figures imply that a large proportion of monogastric[5] livestock production comes from the narrow range of high-output breeds that are raised in specialized industrial systems. These breeds have been intensively bred for meat or egg production and tend to be widely distributed internationally. Small-scale, backyard pig and poultry production based largely on locally adapted breeds (a wide and diverse range of breeds, reflecting diverse local conditions) is nonetheless still significant. For example, according to the above-cited study, half the pig population in developing countries was being raised in "backyard, small-scale and low-input systems in which pigs represent an important source of nutrition and income."

Food production from ruminants still comes largely from grazing or mixed crop–livestock production systems (ibid.). Animals in these systems are relatively dependent on locally available feed resources and exposed to the vagaries of the local environment. Particularly where conditions are harsh, adaptedness to specific local conditions remains important and hence a wide range of locally adapted breeds continue to be raised. Nonetheless, certain high-output breeds, such as Holstein-Friesian dairy cattle, have become very widespread and provide a disproportionately large share of the global supply of animal products from ruminants.

Finally, in addition to its significance to current food production, the diversity of animal genetic resources at species, breed and within-breed levels provides options for the future development of food production systems, whether through the introduction of species and breeds into new production systems or through breeding (genetic improvement) (FAO, 2015a).

---

[5] Monogastric animals are those that do not have a rumen.

2.1.2 Terrestrial crop plants

As noted above, terrestrial plants are the main sources of calories and protein in the human diet globally. While wild plants make important contributions to many people's diets (see Section 2.1.4 for further discussion), the bulk of the world's plant-sourced food comes from domesticated crop plants. Among the world's approximately 391 000 species of vascular plants (RBG Kew, 2016), it has been estimated that a little over 6 000 have been cultivated for food (IPK, 2018). Fewer than 200 of these species are currently produced in sufficient quantities to be listed in global production statistics (FAOSTAT), with only nine (sugar cane, maize, rice, wheat, potatoes, soybeans, oil palm, sugar beet and cassava) accounting for 67 percent of all crop production by weight in 2018. Where energy is concerned, these nine crops accounted for 70 percent of crop calories in the human food supply as of 2017.[6] In the case of protein supply, wheat, rice, maize, potatoes and soybean are the dominant individual crops globally, together accounting for 67 percent of protein supply from crops in 2017.

As is the case in other sectors, global food-supply figures for crops mask variation from region to region, country to country and locality to locality associated with differences in agroclimatic conditions, culinary traditions, levels of prosperity, etc. Moreover, figures for calorie and protein supply do not account for the significance of crop diversity to the availability of micronutrients, many of which tend to be deficient in diets based heavily on a few staple crops (e.g. Welch, 2002). It is often also the case that varieties within a given species differ significantly in their micronutrient content (e.g. Burlingame, Charrondiere and Mouille, 2009). Dietary diversity is regarded as a good predictor of dietary quality, particularly in the case of children's diets (Kennedy *et al.*, 2007; Moursi *et al.*, 2008; Parlesak, Geelhoed and Robertson, 2014; Rah *et al.*, 2010). The availability of a range of diversely adapted species and varieties also means that production can occur in a range of production environments and can help reduce the levels of inputs required (e.g. irrigation water for water-demanding crops in dry areas). Growing a range of crop species and varieties at the scale of the field, farm or landscape can give rise to a range of complementarities and synergies that increase and/or stabilize output, reduce input use and reduce risks (Dawson *et al.*, 2019; DuVal, Mijatovic and Hodgkin, 2019).

Traditionally, many crop (and mixed) food production systems have been highly diverse in terms of the species and varieties grown. The overall status of within-species crop diversity on farms around the world and its precise significance in terms of food production are difficult to estimate. Relatively homogeneous, often large-scale, farms have become more widespread, and there are concerns about genetic vulnerability[7] and the loss of crop genetic diversity in many countries (FAO, 2010a). However, studies have found that many traditional varieties continue to be maintained on farm (ibid.). A large proportion of global food production comes from small farms (FAO, 2014a), many of which are relatively diverse in terms of the genetic resources they utilize. As noted above for livestock, the significance of crop diversity lies not only in its current role in production but also in the options it provides for future use in breeding programmes and in adapting farm management strategies.

---

[6] Unless otherwise indicated, all figures presented in this subsection are based on FAOSTAT data (http://www.fao.org/faostat/en/#home) accessed May 2020.

[7] "The condition that results when a widely planted crop is uniformly susceptible to a pest, pathogen or environmental hazard as a result of its genetic constitution, thereby creating a potential for widespread crop losses" (FAO, 1997). as a result of its ger

2.1.3 Aquatic species

A very diverse range of aquatic species are raised in aquaculture. As of 2016, production data for about 598 "species items"[8] had been recorded by FAO: 369 of finfish; 109 of molluscs; 64 of crustaceans; 9 of other aquatic invertebrates; 7 of amphibians and reptiles; and 40 of aquatic algae (FAO, 2018a). Moreover, many of the country reports submitted as a basis for the preparation of the report on *The State of the World's Aquatic Genetic Resources for Food and Agriculture* (FAO, 2019) indicated that more species were being farmed than had been reported via the regular FAO statistical survey. Countries also reported a number of species considered to have potential for future use in aquaculture. Despite the large total number of species items farmed, production at national, regional and global levels is dominated by a relatively small number of "staple" species (FAO, 2018a). For example, in 2016, 27 species items supplied more than 90 percent of farmed finfish production.[9]

Among[10] freshwater and diadromous fish,[11] farmed types range from low trophic-level species, such as carps, barbs, tilapia and pacu, to highly carnivorous species such as salmon, eel and snakehead. The majority of production volume comes from lower trophic-level species – relatively efficient producers of high-quality protein and thus of major significance to global food security. The salmonids are very significant in value terms, and improvements to their production systems mean that these carnivorous fishes are becoming more efficient users of feed resources. Although marine finfish represent a low proportion of total finfish aquaculture production, 33 different families are farmed. Farmed marine finfish tend to be carnivorous (e.g. snappers, groupers, pompano and tuna), but also include a few species that are omnivorous or herbivorous (e.g. mullet, scats and rabbitfish). Among crustaceans, marine/brackishwater production is dominated by the penaeid shrimp, with minor contributions from other families such as lobsters and metapenaeids. Freshwater crustacean aquaculture production comes from Chinese mitten crab, various crayfish/crawfish species and Macrobrachium freshwater prawns. Farmed molluscs are mainly bivalves and gastropods. Cephalopod aquaculture production is very limited. Other species contributing to aquaculture production include sea cucumbers, sea urchins, frogs and turtles. Crocodile production is growing quickly in Asia. Aquatic plant production is dominated by seaweeds.

Table 1 shows the contributions of different taxonomic groups to world food production from aquaculture in 2016. In the case of inland aquaculture, finfish production is very dominant, although the proportion of production accounted for by this taxonomic group declined from 97.2 percent to 92.5 percent between 2000 and 2016, because of relatively faster growth in other categories, particularly an increase in the production of crustaceans (including shrimps, crayfish and crabs) in Asia (FAO, 2018a). In marine and coastal aquaculture, in contrast, mollusc production dominates in terms of volume produced. Crustaceans account for a relatively small percentage of production volume, but are disproportionally significant in value terms. Aquatic animals belonging to other taxonomic groups are still quite marginal in terms of production volume, although some, such as Japanese sea cucumber (*Apostichopus japonicas*), are of high value. Global farmed aquatic plant production amounted to 30 million tonnes in 2016, up from 13.5 million tonnes in 1995 (ibid.).

---

[8] A species item is a single species, a group of species (where identification to the species level is not possible) or an interspecific hybrid.

[9] Detailed production statistics can be found in FAO's Fishery and Aquaculture Statistics Yearbooks: http://www.fao.org//fishery/publications/yearbooks/en

[10] This paragraph is based on FAO (2019).

[11] Fish species that migrate between freshwater and the sea.

Table 1. World food production from animal aquaculture in 2016, by taxonomic group

	Inland aquaculture	Marine and coastal aquaculture	Quantity total		Value total	
	(tonnes)	(tonnes)	(tonnes)	(Percentage by volume)	(USD billion)	(Percentage by value)
Finfish	47 516	6 575	54 091	68	138.5	60
Crustaceans	3 033	4 829	7862	10	57.1	25
Molluscs	286	16 853	17 139	21	29.2	13
Other animals	531	407	938	1	6.8	3
Total	51 367	28 664	80031	100	231.6	100

Source: Data from FAO, 2018a.

Marine capture fishery production amounted to 79.3 million tonnes in 2016, 41.9 percent of which came from 25 major species and genera (FAO, 2018a). Most of these were finfish – largest contributors were the Alaska pollock (*Theragra chalcogramma*), anchoveta (*Engraulis ringens*), skipjack tuna (*Katsuwonus pelamis*) and sardinellas (*Sardinella* spp.) – but they also included the jumbo flying squid (*Dosidicus gigas*), the Gazami crab (*Portunus trituberculatus*) and the Akiami paste shrimp (*Acetes japonicas*). Inland capture fishery production amounted to 11.6 million tonnes in 2016. A large number of species contribute to this production. However, much of the reported output is not broken down by species, i.e. production is only noted as coming from freshwater fish, molluscs or crustaceans (FAO, 2018a). Among production for which species is recorded, the predominant species are the carps and other cyprinids, tilapia, Nile perch and freshwater prawns.

In addition to their contributions to the supply of calories and protein, aquatic species are also important sources of vitamins and pigments (e.g. spirulina and artemia) and omega-3 lipids (oily fish and marine phytoplankton) and are widely used in the production of food (and animal-feed) supplements (Coutteau *et al.*, 1997; Sargent, 1997; Habib *et al.*, 2008; de Deckere, 2001; Simopoulos, 1991; Adarme-Vega *et al.*, 2012).

Production data at the level of stocks and strains within species are limited in the aquatic sector. However, within-species diversity enables production in a range of different environments and provides the basis for adaptation to future changes through natural or human-controlled selection (FAO, 2008a).

2.1.4 Wild foods

Wild foods, as defined for the purposes of the SoW-BFA, are food products obtained from non-domesticated species. However, the distinction between wild and domesticated sources is not clear cut: wild foods have been described as lying "along a continuum ranging from the entirely wild to the semidomesticated, or from no noticeable human intervention to selective harvesting, transplanting, and propagation by seed and graft" (Harris, 1989). They may be harvested, gathered or hunted in natural or semi-natural ecosystems or in and around cultivated/intensively managed production systems (crop fields, plantations, gardens, fishponds, etc.). Wild foods include a diverse variety of products, ranging from mushrooms, fruits, leafy vegetables, woody foliage, bulbs and tubers, cereals and grains, nuts and kernels, and saps and gums to honey, birds' eggs, fish and shellfish, terrestrial invertebrates such as insects and snails and meat from small and large vertebrates (Bharucha and Pretty, 2010; Shackleton *et al.*, 2010; CBD and WHO, 2015). Within each of these groups, up to several hundred different species may be eaten.

The most important category of wild food in terms of volume and protein supply globally is wild-caught fish and aquatic invertebrates (see Section 2.13). Capture fisheries are particularly significant to food security in certain regions of the world, including notably Oceania, where average national annual consumption of fish (including shellfish) per person in 2013 was 27 kg, relative to a global average of 19 kg. Figures for Melanesia (34 kg), Polynesia (46 kg) and Micronesia (72 kg) were even higher.[12] Bell *et al.* (2013) report figures of 146 kg per person per year for coastal fishing communities in Tuvalu. Freshwater capture fisheries are extremely important in many developing countries, particularly in landlocked areas such as the interiors of Southeast Asia, Africa and South America.America.

Wild foods are a major non-wood forest product (NWFP). Recent global figures for the value of NWFPs have not been published. However, in 2005, the value of recorded food products from forests (mostly fruit, berries, mushrooms and nuts) amounted to more than USD 8.6 billion globally, with wild honey and beeswax accounting for a further USD 1.8 billion, wild meat for USD 577 million and "other edible animal products" for USD 1 million (FAO, 2010b). Given that most NWFPs do not enter the commercial market and that there are many gaps in reporting and in the availability of data and relevant assessment tools at country level (FAO, 2014b; Sorrenti, 2017; FAO, 2016a), these figures are likely to be considerable underestimates of the actual value of wild foods from forests. More recent figures for the value of NWFPs, including wild foods, in Europe are given in Section 2.2.3.

Wild foods contribute significantly to the food security of very large numbers of people (Bharucha and Pretty, 2010; Rowland *et al.*, 2017; Sunderland, 2011). However, the site-specific nature of the data available on frequency of consumption, species consumed and contributions to protein, energy and micronutrient dietary intakes means that global estimates of the importance of wild foods to nutrition are difficult to establish. A lack of information on the nutritional composition of wild foods (Bharucha and Pretty, 2010; Colfer, Sheil and Kishi, 2006; Grivetti and Ogle, 2000; Powell *et al.*, 2015) and on the variability of nutritional composition within species (Stadlmayr *et al.*, 2013; Toledo and Burlingame, 2006) is another constraint. Powell *et al.* (2015) note that although the contribution of wild foods to total energy and protein intake is generally low, several studies have identified cases in which a high proportion of the dietary intake of micronutrients is obtained from wild foods. A survey of nearly 8 000 rural households in 24 countries across Africa, Latin America and Asia found that 39 percent of households harvested wild meat, most of which was used for subsistence, indicating that wild meat is a major source of protein and other nutrients for many millions of rural people in the tropics and subtropics (Nielsen *et al.,* 2018).

Wild foods are consumed for a wide range of reasons and in a variety of circumstances, including both year-round and seasonal use, the latter occurring for example when other foods are in short supply or when people have time to harvest them because of lulls in other activities. They make a range of contributions to livelihoods, food security and nutrition, including by increasing dietary diversity, increasing resilience against shocks such as crop failure and providing a source of income via sales (Bharucha and Pretty, 2010; Hickey *et al.*, 2016; Johns and Sthapit, 2004; Schulp, Thuiller and Verburg, 2014; Vinceti *et al*., 2013; Wunder, Angelsen and Belcher, 2014).

2.2 Raw materials

Crop, livestock, forest and aquatic production systems and the biodiversity used in and associated with them supply a wide range of non-food products, including materials used as fuels, in construction and in the manufacture of textiles, clothing, cosmetics and many

---

[12] Figures from FAOSTAT (http://www.fao.org/faostat/en/#home) accessed May 2020.

other goods. Ornamental products and materials used for medical and other biochemical purposes are discussed separately in Sections 2.4 and 2.5.

2.2.1 Terrestrial domesticated animals

In terms of the value of marketed products, the most significant non-food materials produced by the livestock sector are fibres, hides and skins. Global sheep-wool production in 2018 amounted to almost 2 million tonnes.[13] Fibres from other animals are produced in much lower quantities, but include high-quality products such as alpaca wool, cashmere and mohair. Within-species breed diversity adds to the diversity of fibres available. A range of species and breeds also provides diversity in the supply of hides and skins (global production of cattle, buffalo, sheep and goat hides and skins was almost 12.3 million tonnes in 2018). Animal dung, as well as being a major source of manure for use in agriculture, is widely used as a fuel, either in the form of dung cakes or as a source of biogas.

As well as providing material products, livestock are also a source of motive power. Species such as horses, donkeys, cattle and dromedaries provide transport for goods and people and traction in agriculture. At the end of the twentieth century, 30 percent of cropland in developing countries was being cultivated using draught animals (the remaining 70 percent was equally divided between hand and mechanized cultivation) (FAO, 2003). The share of animal power was predicted to fall to 20 percent overall by 2030, but to increase in sub-Saharan Africa (ibid.). Again, the availability of a range of breeds – including specialized draught, pack and riding animals – underpins the supply of these services.

2.2.2 Terrestrial crop plants

Major non-food products obtained from crop plants include biofuels and fibres. The former include liquids (e.g. ethanol and biodiesel), biogas and solid biomass. Ethanol is obtained from plant materials that contain large amounts of sugar or substances that can be converted into sugar (FAO, 2008b). Production is largely based on sugar crops (sugar cane and sugar beet) and starchy crops, such as cereals, i.e. on materials that could potentially be eaten by humans. Only a small fraction comes from lignocellulosic materials, such as wood and straw (OECD/FAO, 2016). Biodiesel is produced using oil extracted from crops such as rapeseed, oil palm, soybean, sunflower, peanut and jatropha (FAO, 2008b). Sources of biomass for heat and power include various agro-industrial and post-harvest residues and dedicated energy crops such as short-rotation perennials (eucalyptus, poplar, willow) and grasses (miscanthus and switchgrass) (ibid).

Where fibres are concerned, cotton is by far the most significant crop in terms of production volume; other major natural fibres derived from plants include jute, sisal, flax and hemp (van Dam, 2008). As with food crops, genetic diversity within fibre-producing species is vital to efforts to increase productivity and address threats such as pests and diseases (FAO, 2010a). Many fibre crops supply important by-products such as oilseeds (van Dam, 2008). Some provide materials used for an extremely wide range of purposes. The global market for hemp, for example, reportedly encompasses more than 25 000 products across the agriculture, textile, recycling, automotive, furniture, food and beverage, paper, construction and personal-care sectors (Johnson, 2018).

Crop plants provide vital raw materials for livestock production. An estimated 19 percent of dry matter fed to livestock globally consists of crop residues (straws, stovers, sugar-cane tops and banana stems), 13 percent of human-edible grains and 8 percent of "fodder crops" (grain and legume silage and fodder beets), with oil-seed cakes and other agro-industrial by-products accounting for another 10 percent each (Mottet *et al.*, 2017).

---

[13] All figures in this paragraph taken from FAOSTAT (http://www.fao.org/faostat/en/#home) accessed May 2020.

Part of the 46 percent accounted for by "grass and leaves" (ibid.) comes from sown grasses, legumes and other forages. A wide range species and varieties are grown as forage crops around the world, with the particular types and combinations grown varying depending on the climate and the nutritional needs of the animals fed (Capstaff and Miller, 2018). Genetic improvement of forage species is a relatively recently established activity as compared to that of cereals, fruits and vegetables, and has focused mainly on increasing yields and tolerance of harsh climatic conditions (ibid).

2.2.3 Forests and trees outside forests

Forests and trees outside forests supply a vast range of wood products and NWFPs. The former include wood used in construction, for pulp, in the manufacturing of a wide variety of wooden items, including furniture and tools, and as fuel. Global roundwood[14] production in 2018 amounted to 4 billion m^3, 1.9 billion m^3 of which was used for wood fuel.[15] NWFPs, in addition to food, ornamental and medicinal products (see Sections 2.1, 2.4 and 2.5), include a range of other plant- and animal-sourced materials such as bamboo (e.g. for use in construction and the manufacture of household items, tools and textiles), rattan (e.g. for use in producing furniture, canes, clothes and decorative items), cork (e.g. for use in wine bottling and in construction), bark, latexes, gums, resins (e.g. for use in producing turpentine), hides, skins and beeswax.[16] While synthetic alternatives to many NWFPs have been developed, in places there is now a resurgence of interest in natural products that are less polluting or higher in quality or that embody aspects of local culture, including in the context of hobby interest in traditional crafts and "survival skills" (Wong and Wiersum, 2019).

As noted in Section 2.14, no recent global figures for the value of NWFPs have been published and older published figures are recognized as being considerable underestimates. Regional figures for Europe (not including the Russian Federation) put the value of marketed NWFPs at EUR 1.6 billion for plant products (of which 47.2 percent came from ornamental plants, 29.0 percent from food, 20.9 percent from other plant products, 1.5 percent from raw material for medicine and aromatic products, 0.7 percent from exudates and 0.7 percent from raw materials for utensils handicrafts and construction) and EUR 0.62 billion for animal products (of which 51.10 percent came from wild meat, 45.68 percent from wild honey and beeswax, 2.90 percent from hides skins and trophies, 0.21 percent from other edible and non-edible animal products, 0.08 percent from living animals and 0.02 from raw materials for medicine) (Forest Europe, 2015). Thus, raw materials for uses other than food, ornament and medicines account for a relatively small proportion of the recorded value of NWFPs. Global figures for 2005 (FAO, 2010b) also show food and ornamental plants accounting for most of the recorded value of NWFPs.

Material obtained from forests and trees are crucial to the livelihoods of many people around the world. For example, as of 2011 an estimated 2.4 billion people (34 percent of the global population) relied on wood fuel (wood or charcoal) for cooking, including 63

---

[14] Roundwood comprises "all wood obtained from removals, i.e. the quantities removed from forests and from trees outside the forest, including wood recovered from natural, felling and logging losses during the period, calendar year or forest year. It includes all wood removed with or without bark, including wood removed in its round form, or split, roughly squared or in other form (e.g. branches, roots, stumps and burls (where these are harvested) and wood that is roughly shaped or pointed. It is an aggregate comprising wood fuel, including wood for charcoal and industrial roundwood (wood in the rough). It is reported in cubic metres solid volume underbark (i.e. excluding bark)." (Eurostat *et al*., 2018).

[15] FAOSTAT data accessed May 2020.

[16] According to FAO (1999), "non-wood forest products consist of goods of biological origin other than wood derived from forests, other wooded land and trees outside forests." However, a range of range of different terms and definitions are used to describe products of this kind (Sorrentini, 2017). In the case of the Global Forest Resources Assessment, countries are invited to report on "goods derived from forests that are tangible and physical objects of biological origin other than wood" (FAO, 2018b).

percent of the population of Africa, 38 percent of the population of Asia and Oceania and 15 percent of the population of Latin America and the Caribbean (FAO, 2014c). As of the period 2000 to 2010, at least 1.3 billion people were living in homes where the walls, roofs or floors were constructed mainly from forest products, including 1 billion in Asia and Oceania and 150 million in Africa (ibid.).

Natural rubber,[17] which is mainly produced from the latex of the rubber tree (*Hevea brasiliensis*), is a major industrial raw material that is used for a variety of purposes, including in the production of tyres. Rubber trees are grown exclusively in the developing regions of the world: FAOSTAT data for 2018 indicate that 90 percent of the 14.3 million tonnes produced globally came from Asia.[18]

The total number of tree species in the world is estimated to be about 60 000 (Beech *et al*., 2017). Country reports submitted to FAO as inputs to the preparation of *The State of the World's Forest Genetic Resources* (FAO, 2014a) refer to more than 1 000 species "actively managed" for timber, a similar number for non-wood forest products (including foods) and about 500 for fuel. Many more are used in one way or another as sources of raw materials of various kinds. However, planted forests, which make up about 7 percent of the global forest area and account for more than 50 percent of the world's industrial roundwood production, are largely based on about 30 tree species belonging to four genera (*Acacia, Eucalyptus, Pinus* and *Populus*) (ibid).

Genetic diversity within tree species enables them to grow across a range of environmental conditions and to provide products with a variety of specific characteristics. Genetic diversity also provides the basis for evolution in response to changes in environmental conditions and for genetic improvement programmes aimed at increasing yield or resistance to diseases or other stressors. Globally, more than 700 tree species are subject to genetic improvement activities of some kind (FAO, 2014b).

2.2.4 Aquatic species

Non-food products provided by aquatic plants and animals include natural sponges, fish-skin leathers, hides from alligators and other reptiles, jewellery (e.g. pearls and abalone and trochus shells) and cosmetic compounds. A number of aquatic plant species provide products that are essential for food processing and other industrial purposes. For instance, phyco-colloids derived from seaweeds (e.g. alginates and carrageenans) have a wide range of uses as binders and gelling agents in processed foods (Hurtado, 2017). Marine algae, especially seaweeds, are also harvested for use in biofuel production (Mata, Martins and Caetano, 2010; Milledge *et al*., 2014).

2.3 Freshwater

Ecosystems contribute in many ways to the supply of freshwater that can be used domestically, in food and agricultural production systems and in industry. For example, vegetation, particularly forest vegetation, is thought to influence rainfall levels.[19] Vegetation, as well as dead plant material that provides soil cover, also affects the balance between water infiltration into the soil and run-off into downstream areas. Infiltration and run-off rates are also affected by soil structure, soil texture and soil organic matter content, which are in turn affected by the actions of (among other components of biodiversity) soil micro-organisms and invertebrates (see Section 3.3 for further discussion). Increasing

---

[17] The explanatory notes to the terms and definitions used for the Global Forest Resources Assessment indicate that non-wood forest products specifically include rubber/latex whether from natural forests or plantations (FAO, 2018b).

[18] FAOSTAT data (http://www.fao.org/faostat/en/#home), accessed May 2020.

[19] For further discussion, see Section 3.1.2.

infiltration rates means that water is released more slowly and over a longer period, which may help to keep streams and rivers flowing during dry periods of the year (TEEB, 2010).

Ecosystems also contribute to water purification. A range of different physical, chemical and biological processes contribute to removing contaminants (harmful organic and inorganic substances, pathogenic microbes, etc.) from water supplies as they pass through soils or through water bodies such as rivers and lakes. Many different organisms contribute to the process of filtering pollutants before they can enter waterbodies, “pumping” them from the water (e.g. into bottom sediments or the atmosphere) or degrading them into benign or less harmful components (Ostroumov, 2010).

The precise relationships between the levels of biodiversity within ecosystems and their capacity to deliver services related to the regulation and purification of water flows are not well understood. Structural diversity within a stand of vegetation increases the range of mechanisms through which runoff can be reduced. There is also some evidence that greater species diversity within a particular type of plant community is associated with a greater capacity to prevent excess run-off (see Section 3.3.1). Invertebrate and micro-organism diversity plays a vital role in the formation and maintenance of healthy soils (see Section 3.3.2) and hence to the water-holding and water-purifying services provided by soil ecosystems. The significance of algal species diversity in water purification has been investigated experimentally using artificial streams. For example, Cardinale (2011) showed that a mixture of eight algal species could remove nitrate from the water 4.5 times faster than one species could, the effect arising because of the abilities of the different species to occupy different niches within the stream. Overall, however, there is limited evidence that, in practice, more-diverse ecosystems are more effective than less-diverse ones as providers of water-purification services. Cardinale *et al.* (2012) concluded that more studies had found no relationship between diversity and water-purification capacity than had found a positive relationship.

Many rivers, streams and lakes are bordered by crop, livestock, aquaculture or forest production systems. Riparian forest and grassland vegetation can play a significant role in reducing the flow of sediment, excess nutrients and other pollutants into waterbodies, and “buffer” strips are sometimes planted specifically to deliver this service (Klapproth and Johnson, 2000). The other side of the coin, however, is that crop, livestock, forest and aquaculture production systems are often major sources of pollutant flows into water bodies and thus, where water quality is concerned, are providers of ecosystem disservices rather than services. Various components of BFA can, however, contribute to reducing these disservices. For example, the use of inputs such as pesticides, fertilizers and veterinary drugs that may end up as pollutants of aquatic ecosystems can be reduced by using more disease-resistant or pest-resistant varieties or breeds of domesticated plants, fish or terrestrial livestock, or by taking advantage of the pest-control and soil fertility-enhancing services provided by associated biodiversity.

2.4 Medicinal and other biochemical resources

Many components of BFA are valued for their medicinal properties or as sources of biochemical substances that can be used in the manufacture of pharmaceuticals, cosmetics, crop protection agents and other biochemical products. For example, domesticated plant species contain a wide range of chemical compounds that can be used for such purposes (Harborne, Baxter and Moss, eds, 1999; Ranalli, ed, 2007). Many agricultural by-products can be used as substrates for microbial processes that generate substances such as organic acids, enzymes, surfactants and pigments (Chatzipavlidis *et al.*, 2013). Various marine species are sources of bio-active compounds that can be used as pharmaceuticals (e.g. haemocyanin from the keyhole limpet) (Donia and Hamann, 2003; Harnedy and FitzGerald, 2012; Harris and Markl, 1999; Kim and Mendis, 2006; Rocha *et al.*, 2011;

Sipkema *et al.*, 2005). Terrestrial livestock species are also sources of various pharmaceutical substances, including insulin, antibodies and hormones (Redwan, 2009).

Numerous medicinal plant species, especially aromatic herbs, are grown in home gardens around the world and some are cultivated as field crops (Schippmann, Cunningham and Leaman, 2002). Many people in developing countries rely heavily on medicinal species collected from the wild, for example from forest ecosystems. A range of different industries engage in bioprospecting for substances with valuable properties or for species with characteristics that can provide models or inspiration for new innovations (Beattie *et al.,* 2011). Many commercially traded medicinal plants are collected from the wild rather than being cultivated (Schippmann, Cunningham and Leaman, 2002; Chen *et al.*, 2016).

2.5 Ornamental resources

Ornamental products are a significant component of the provisioning services provided by BFA. For example, domesticated ornamental plants are of major cultural[20] and economic importance (Ciesla, 2002; van Tuyl *et al.*, 2014). Aesthetic objectives are often important in breeding strategies for pet or companion animals. Among many domesticated species raised for food and agricultural purposes, some breeds or varieties are valued primarily for their aesthetic characteristics. There are, for example, many "fancy" breeds of chicken, pigeon, rabbit and other species. Among crops and forages, there are ornamental varieties of species such as cabbage, capsicum, tall grass and pumpkin. Many tree species have long been used for ornamental purposes. Their aesthetic features (e.g. foliage colour and density, form, size and shape), their fragrances and the shade they provide help to create serene settings in gardens, city parks, along streets, etc., and natural and planted woodlands and trees are often key elements of visually appealing rural landscapes (Ciesla, 2002). Ornamental tree species are embedded in many rituals, celebrations and customs (Crews, 2003). Ornamental fish include species specifically bred for their appearance as well as species that are taken from the wild. It has been estimated that the freshwater aquarium trade relies on cultured animals for around 98 percent of its products and that only 2 percent are captured (Sugiyama, Staples and Funge-Smith, 2004). The reverse is true for the marine aquarium trade, which relies on capture for 98 percent of its production (ibid.). The global marine aquarium trade regularly transports large numbers of species. Wabnitz *et al.* (2003) provide a figure of 1 471 species of fish traded worldwide. Rhyne *et al.* (2012) report that over marine 1 800 aquarium species were imported into the United States of America alone in 2005.

---

[20] The cultural significance of BFA and the products it supplies is further discussed in Section 4.

3. Regulating, supporting and habitat services

It is increasingly recognized that crop, livestock, forest and aquatic production systems benefit directly or indirectly from a wide range of different regulating, supporting and habitat services. However, in the following descriptions the field is explored from the opposite perspective, i.e. the role of the biodiversity (both non-domesticated and domesticated) found in and around food and agricultural systems in the supply of ecosystem services, whether to food and agriculture or more widely.

3.1 Air-quality and climate regulation

3.1.1 Air-quality regulation

Gaseous and particulate air pollutants cause serious problems for human health and for ecosystem functions. Particulate matter can be removed from the air by becoming attached to the surfaces of trees and other plants and gaseous pollutants can be absorbed through leaf stomata. It has been estimated that trees and forests in the (conterminous) United States of America removed 17.4 million tonnes of air pollution in 2010, a service valued at USD 6.8 billion (Nowak *et al*., 2014). Although most of the pollution removal occurred in rural areas, most of the impact on human health and most of the value generated was provided by trees in urban areas. Thus, a major share of the value of this service comes from trees growing in parks, cemeteries, gardens and streets rather than in forest production systems *per se*.

Different tree species have different capacities to remove pollutants from the air, for example because of their size or because of the characteristics of their leaves (Smith, 2012). A downside of trees in urban locations is that biogenic volatile organic compounds emitted by trees can cause an increase in ozone pollution [21] (Donovan, Mackenzie and Hewitt, 2005; Karlik and Pittenger, 2012). Again, different types of tree will produce different quantities of these compounds (ibid.). Thus, if trees are to be used to improve air quality, appropriate species need to be chosen, taking these differences into account and also considering the need for the trees to be well adapted to local conditions (Smith, 2012).

3.1.2. Climate regulation

Ecosystems and their constituent biodiversity can affect the climate at both local and global scales. Local-scale effects on temperature have been demonstrated, for example, by Alkama and Cescatti (2016), who found that, in all climatic zones except at the most northern latitudes, forest clearance increased temperatures, particularly mean annual maximum air surface temperatures, with the most marked effect being observed in arid zones. Shading provided by trees can have a significant effect on the temperatures of rivers and streams and hence on river biodiversity and the ecosystem services it provides, including the supply of food from river fisheries (Bowler *et al.*, 2012; Lenane, 2012). Forests can also affect rainfall patterns. For example, Spracklen, Arnold and Taylor (2012) concluded, based on models of atmospheric transport of air masses and satellite observations of rainfall and vegetation cover, that, over more than 60 percent of the tropical land surface, air that had passed over extensive vegetation in the preceding few days produces at least twice as much rain as air that has passed over little vegetation. Trees in the Amazon rainforest have been found to generate rain via transpiration of water from leaf surfaces and thereby initiate the rainy season two or three months before moist air from the ocean arrives in the region (Wright *et al.*, 2017).

---

[21] The biogenic volatile organic compounds react with oxides of nitrogen (e.g. those emitted by motor vehicles) in the presence of sunlight to produce ozone, a pollutant that is harmful to plants and to human health..

Globally, the climate is affected by the absorption and release of greenhouse gases such as carbon dioxide and methane. In a world struggling to address climate change, the sequestration of carbon has become an increasingly significant ecosystem service. Forests, grasslands, wetlands and marine ecosystems – many of which serve as forest, livestock or fishery production systems – are key players in global carbon cycles.

Soils hold the largest terrestrial carbon pool (Scharlemann *et al.*, 2014) and play a crucial role in regulating the exchange of greenhouse gases with the atmosphere. Biodiversity, both above and below ground, is vital to soil health and hence to soil's contribution to carbon sequestration (Beed *et al.*, 2011; Cock *et al.,* 2011) (see Section 3.3 for further discussion of soil-related ecosystem services). However, much remains to be learned about the roles of different species and groups of species in the processes that lead to the accumulation and/or release of carbon from the soil and how these processes are influenced by environmental variables. The significance of diversity *per se*, for example whether and how the number of species present in the community or in a particular functional group influences the provision of carbon-sequestration services, also needs to be better understood. Some studies have shown that higher levels of diversity in grassland plant communities lead to more carbon sequestration in the soil (Fornara and Tilman, 2008; Steinbeiss *et al.*, 2008). Lange *et al.* (2015) found that the positive effect of plant diversity revealed by long-term data from a grassland biodiversity experiment in Europe operated via its influence on soil microbial communities.

The significance of diversity in terms of influencing forests' ability to sequestrate carbon is not well understood. As is generally the case with the contributions of biodiversity to the supply of ecosystem services, the capacity of forests to sequestrate carbon benefits from the presence of species and within-species populations that are well adapted to local conditions (Loo *et al.*, 2011). Where smallholder tree plantings in agroforestry systems are concerned, the supply of carbon sequestration services is also dependent on the trees' capacity to provide livelihood benefits to local people, as otherwise they are unlikely to be planted and maintained (ibid.). Globally, higher levels of biodiversity are associated with higher levels of carbon storage in forest ecosystems: tropical moist forests are both diverse and rich in carbon (Midgley *et al.*, 2010). However, no clear association between biodiversity levels and levels of carbon storage has been demonstrated within and among tropical forests, although some studies indicate such a link, and studies in other ecosystems have shown that high diversity is often associated with higher levels of primary production and stability and hence with higher carbon stocks (Hicks *et al.*, 2014). A large-scale experiment in subtropical China comparing forest plots planted with different numbers of tree species found that combining multiple species provided higher levels of productivity: after eight years, 16-species mixtures had accumulated more than twice as much carbon as had monocultures on average (Huang *et al*., 2018).

The ongoing role of forests, whether managed or unmanaged, as carbon stores is dependent on their ability to withstand the various shocks that they have to contend with – droughts, storms, fires, disease outbreaks, etc. – many of which are being exacerbated by climate change, changes to land use and other drivers related to human activities (FAO, 2014b). Resilience in the face of such shocks is therefore an important characteristic. It is relatively difficult to investigate links between biodiversity and resilience in forests (as compared to ecosystems such as grasslands) because they are more difficult to observe and manipulate (Miles *et al.*, 2010). However, theoretical considerations and findings from other ecosystems suggest that resilience may be promoted by higher levels of biodiversity (Hicks *et al.,* 2014).

Grasslands are another major global store of carbon. Aside from vegetation and soil biodiversity (see above), these ecosystems typically host a variety of above-ground animals, including grazing species and the predators that prey on them. Many grasslands have long been used to raise cattle, sheep and other domesticated species. Grazing animals affect plant and soil communities via their grazing, trampling and dunging and via the dispersal of seeds. The effects that grazing livestock have on carbon sequestration depend on a range of factors including stocking rates, how grazing is managed, the climate and the characteristics of the local soils and plant communities. For example, Mcsherry and Ritchie (2013) found that the negative effect of grassland grazing on soil organic carbon density was decreased by an increase in precipitation on finer textured soils. On sandy soils, however, the same increase in precipitation caused an increase in the negative effect of grazing on soil organic carbon (ibid). Choosing appropriate management strategies for grassland systems is therefore a complex matter, particularly given the need to account not only for impacts on the soil but also for the methane and other greenhouse- gas emissions associated with the animal-production process, and the various other environmental and livelihood costs and benefits involved (FAO, 2009; Pilling and Hoffmann, 2011; Gerber *et al.*, 2013).[22]

With regard to the deliberate use of grazing livestock to increase carbon sequestration in grasslands, a review by Garnett *et al.* (2017) concluded that, while in some circumstances appropriately managed grazing can have a positive impact, the overall potential of this approach is modest relative to the scale of greenhouse-gas emissions from the world's grazing systems. With regard to the significance of livestock diversity (as opposed to grazing livestock as an undifferentiated mass), different species and to some extent different breeds have different grazing behaviour (Hoffmann, From and Boerma, 2014) and thus potentially have different effects on carbon sequestration in, or loss from, grassland soils. Likewise, the impacts of mixed herds/flocks can be different from those of single species or breeds (e.g. Chang *et al.*, 2018). However, it remains unclear whether there is any significant potential to promote carbon sequestration or prevent carbon loss by optimizing the choice of grazing animals or how widely any such strategies could be applied.

The soil organic carbon pool in many cropland soils has been severely depleted (Lal, 2013a), especially in soils that are managed using extractive farming practices (Lal, 2013b). However, carbon sequestration in crop production systems can potentially be improved through changes in management practices, some of which involve utilizing the diversity of domesticated BFA and/or promoting the activity of associated biodiversity. Options include no-till farming with crop-residue mulch, integrated nutrient management, complex crop rotations, cover cropping and agroforestry (Corsi *et al.*, 2012; FAO, 2005, 2013b, 2016b; Lal, 2013b). The application of biofertilizers (fertilizers containing living micro-organisms) may contribute to maintaining or increasing the quantity of organic matter in the soil, especially when such fertilizers increase the formation of permanent humus compounds. However, more research is needed into their potential to increase carbon sequestration (Dębska *et al.*, 2016). More generally, soil-dwelling invertebrates and micro-organisms are vital to efforts to maintain and improve the health and the carbon-sequestering capacity of the soil (Gougoulias, Clark and Shaw, 2014).

Aquatic ecosystems and their biota account for the largest carbon and nitrogen fluxes on the planet and serve as its largest carbon sinks (Pullin and White, 2011). Oceans are the world's largest long-term carbon sinks – capturing about 30 percent of the carbon dioxide released annually, most of which is then stored for millennia – and account for 55 percent of all the planet's so-called green carbon (carbon captured through photosynthesis and stored in plants and soils) (Nellemann *et al.*, 2009). Carbon dioxide is removed from the atmosphere both by being dissolved in water and by being used by phytoplankton and other aquatic plants, including those in coastal ecosystems such as mangrove forests, salt marshes and seagrass meadows (Laffoley and Grimsditch, eds, 2009; Laffoley

---

[22] See Hristov *et al.* (2013) and Gerber *et al.* (2016) for more general discussions of mitigation opportunities in the livestock sector.

et al., 2014; Herr *et al.*, 2017). Some aquatic micro-organisms, such as foraminiferans and coccolithophores, incorporate carbon into their bodies in the form of calcium carbonate (Pullin and White, 2011). When they die and sink to the floor of the ocean, much of this carbon is buried in sediments, where it remains locked up indefinitely (ibid.). Calcium carbonate in the skeletal structures of marine invertebrates – particularly echinoderms (starfish, sea urchins, etc.) – and the carbonates precipitated in the intestines of marine fish also make huge contributions to global carbon storage. All these services are underpinned in numerous ways by the wider biodiversity of the respective aquatic ecosystems (Laffoley and Grimsditch, 2009; Laffoley *et al.*, 2014; Pullin and White, 2011). The significance of freshwater ecosystems and their biodiversity in carbon sequestration is also increasingly being recognized (e.g. Battin *et al.*, 2009; Downing, 2010; Mendonça *et al.*, 2017; Taylor *et al.*, 2019).

3.2 Natural-hazard regulation

Natural hazards include events such as droughts, floods, hurricanes, earthquakes, volcanic eruptions, avalanches, wild fires and landslides (FAO, 2015b; TEEB, 2010). The frequency of several kinds of extreme weather events is predicted to increase under climate change, and thus one way in which ecosystems and the biodiversity within them can help to reduce the threat posed by disasters is via their contributions to climate change mitigation (see Section 3.1.2). However, ecosystems also contribute in more immediate ways to the moderation of extreme events. For example, a number of coastal ecosystems (mangroves, coral reefs, seagrass meadows, kelp forests, etc.) provide protection against coastal storms and flooding (Barbier *et al.*, 2011; Barbier, 2014; UNEP-WCMC, 2014; Rao *et al.*, 2015). As noted in Section 2.3, terrestrial and freshwater vegetation influences water flows across the landscape. The relationship between forest cover and flooding in downstream areas is complex and has generated a lot of debate, but in some circumstances and at some scales forests and trees have been found to reduce flood risk (e.g. Marshall *et al.*, 2013; Rogger *et al.*, 2017; Sing *et al.*, 2017). It is unclear whether diversity *per se* at species or genetic level has any influence on the supply of this service other than presumably via general effects on forest resilience.[23]

Agriculture and food production may affect the capacities of ecosystems to moderate extreme events. These effects can be harmful, for example if poorly managed livestock grazing removes vegetation that regulates water flows and reduces the risk of flooding or if crop production practices diminish the water-holding capacity of farmland soils. However, it may be possible to adopt more favourable management methods that reduce such problems or to adapt production systems to improve their capacity to moderate extreme events, both internally and in the wider environment. For example, trees can be planted to provide shelter for crops and livestock against the effects of extreme winds, heat waves, snowfalls, etc. (Gregory, 1995) or to help reduce flooding. Grazing livestock can in some circumstances be used in fire or avalanche prevention (Fabre, Guérin and Bouquet, 2010; Lovreglio, Meddour-Sahar and Leone, 2014; Pecora *et al.*, 2015). Services of this kind can often be best delivered by locally adapted species and populations that can withstand harsh local environments. They may be enhanced by the use of a diverse range of species, varieties or breeds. For example, a shelterbelt consisting of a combination of trees, shrubs and grasses is likely to be more effective than a single-species block of trees, as the latter may create an impermeable barrier that generates turbulence or create a wind-tunnel effect at ground level (CPP, 1999).

---

[23] See DuVal, Mijatovic and Hodgkin (2019) for a discussion of the contributions of BFA to resilience in forest production systems.

3.3 Soil formation and protection and nutrient cycling

3.3.1 Erosion prevention

Soil erosion has severe negative effects on a range of ecosystem services, including food production, carbon sequestration and the regulation of water flows (Pimentel, 2006). The main factor in reducing soil erosion is vegetation cover, which protects the soil from rainfall and wind, acts as a barrier to runoff and – through the actions of plant roots – helps to bind the soil in place (Pimentel, 2006; Zuazo and Pleguezuelo, 2008). Some plants are more effective than others at preventing erosion because of their morphology or capacity to grow in particular locations. (Durán Zuazo and Rodríguez Pleguezuelo, 2008). In crop systems, the choice of crops and production methods, for example use of crop rotation, can reduce the amount of erosion that occurs. The same is true for the numbers and types of animals kept and the grazing strategies practised in grassland production systems (e.g. Bilota, Brazier and Haygarth, 2007).

As discussed above (Section 2.3), structural diversity in stands of vegetation tends to reduce surface runoff of water. However, the precise relationships between plant species diversity and erosion control have not been studied in great depth. A study undertaken by Allen, Cardinale and Wynn-Thompson (2016) found that, on average,[24] increasing the number of riparian herbaceous species in an experimental plot from one to eight significantly reduced soil erosion, although most of the effect was caused by the first few additional species – increases between four and eight had little further effect. The authors conclude that effect probably arises because an increase in diversity has a positive influence on root length and the number of root tips and that interactions between legumes and non-legumes are particularly important (ibid.).

More broadly, the maintenance of plant cover is dependent on a range of ecosystem processes – nutrient cycling, pollination, seed dispersal, etc. – that involve a wide variety of species within the local ecosystem. The risk of losing plant cover may be reduced through the delivery of ecosystem services related to the prevention of disastrous events such as fires (see Section 3.2).

While soil biodiversity is essential to the health of soil ecosystems (see below), the specific relationships between soil biodiversity and soil erosion remain poorly understood. The actions of individual components of soil biodiversity can increase or decrease erosion: for example, fungi hyphae bind soil particles together, while the actions of earthworms may on the one hand reduce erosion by promoting water infiltration and on the other increase erosion by creating cast material that can more easily be moved by water (Orgiazzi and Panagos, 2018). The influence of differences in species diversity within such taxonomic groups remains unclear (ibid.).

3.3.2 Maintenance of soil quality

Healthy soils are vital to food production and to the delivery of a range of other ecosystem services (FAO and ITPS, 2015; Orgiazzi *et al*., 2016). Soil formation and maintenance are inextricably linked to biodiversity. Micro-organisms contribute to soil formation by facilitating the breakdown of organic matter and of the parent mineral material of the soil, establishing nutrient-cycling pathways and creating habitats for other organisms (Beed *et al*., 2011; Schulz *et al*., 2013). They are key players in the carbon cycle, breaking down dead plant and animal matter and releasing carbon back into the atmosphere in the form of carbon dioxide or accumulating it within their bodies, with some of the latter ending up in long-term storage in soil organic matter (Gougoulias, Clark and Shaw, 2014). Micro-

---

[24] The overall best performer was a monoculture. However, species mixes performed better on average. The authors argue that the significance of the findings for conservation and restoration decisions relate to the likelihood of there being uncertainty about, *inter alia*, the erosion prevention capacities of individual species.

organisms secrete substances that act as binding agents and promote the aggregation of soil particles, which in turn affects the carbon-storing and water-storing properties of the soil (Beed *et al*., 2011). Micro-organisms are also vital to the nitrogen cycle. Different groups of bacteria fix nitrogen from the atmosphere in the form of ammonia, break down proteins in dead plant and animal material, convert ammonia, via nitrites, into nitrates that can be taken up and used by plants, and return nitrogen to the atmosphere via denitrification (ibid.). Certain types of bacteria and fungi enter into mutually beneficial symbiosis with plant roots: nodulating bacteria associated with the roots of many legume species fix nitrogen from the atmosphere, while mycorrhizal fungi increase plants' capacity to take up phosphorus and other nutrients (Barrios, 2007). Another important role played by micro-organisms is in breaking down soil pollutants; various bioremediation methods involve managing micro-organisms for this purpose, either *in situ* or *ex situ* (Beed *et al*., 2011; Chatzipavlidis *et al*., 2013).

Soil invertebrates include species such as earthworms and ants that act as so-called ecosystem engineers and modify the physical properties of the soil. Structures created by these organisms influence soil processes such as the infiltration, storage and release of water and the sequestration of organic matter. They provide habitats for communities of smaller invertebrates and micro-organisms (Cock *et al*., 2011). A wide range of invertebrates, including those that live in the leaf litter, contribute to the decomposition of plant material and the release of nutrients (ibid.).

Soil micro-organisms and invertebrates are bound together in complex food webs and via their effects on each other's habitats. The diversity of these communities is important to the effective functioning of the soil ecosystem. By experimentally manipulating soil communities, studies have shown that reducing diversity can impair various soil processes, including decomposition, nutrient retention and nutrient cycling (Wagg *et al*., 2014), and reduce resilience to shocks (Griffiths *et al*., 2000).

Plants play an important role in physically protecting the soil from erosion (see Section 3.3.1). They also contribute organic matter to the soil, and their roots affect soil structure through their physical action and by releasing substances that bind soil particles (Angers and Caron, 1998). As noted above, a range of practices and strategies can be used to maintain and improve soil quality in crop production systems, including composting, use of cover crops/green manure crops, crop rotation, zero or reduced tillage and agroforestry.

Above-ground animals, including domesticated livestock, drop dung and urine onto the ground, trample the soil with their feet and may spread plant seeds across the landscape (Hoffman, From and Boerma, 2014). These actions can have both positive and negative effects on soil quality. Excessive trampling can damage the soil structure and large doses of nutrients from dunging can harm vegetation, disrupt nutrient cycling and result in the loss of nutrients into watercourses where they act as pollutants (Bilota, Brazier and Haygarth, 2007). On the other hand, animal manure can serve as an important source of nutrients and be used to increase the organic matter content of depleted soils. It is widely used as a fertilizer in crop production. Appropriate use of animal manure can increase the abundance and diversity of soil micro-organisms and invertebrates and promote their contributions to soil quality (Graham, Grandy and Thelen, 2009; Sradnick *et al*., 2013).

3.3.3 Nutrient cycling in aquatic ecosystems and the wider environment

Nutrient cycling is also vital to the functioning of aquatic ecosystems, including those used for fisheries and aquaculture. Nitrogen is fixed from the atmosphere by micro-organisms (Arigo, 2005; Gruber, 2008; Paerl, 2017) and cycled through waterbodies and sediments by a wide range of organisms (Palmer *et al*., 2000; Hauer *et al*., 2016). As further discussed in Section 3.6, a range of species contribute to the removal of excess nutrients from aquatic ecosystems.

Biodiversity also contributes to the cycling of nutrients between terrestrial and aquatic ecosystems and across wider landscapes and seascapes. Large animals can enable the movement of nutrients over long distances. In parts of Africa, large quantities of nutrients are transferred from grasslands into watercourses in the bodies of wildebeest (gnus) (*Connochaetes* spp.) that drown during their migrations and in the dung and urine of hippopotamuses and other animals (Pringle, 2017; Subalusky *et al*., 2015, 2017). Another example is the transfer of nutrients from the ocean into North America's inland streams and lakes in the bodies of migrating salmon and then into terrestrial ecosystems via predators such as bears and wolves that feed on the salmon (Adams *et al*., 2010; Cederholm *et al*., 1999; Hilderbrand *et al*., 1999). In the oceans, whales and seals recycle nutrients into surface waters by feeding at depth and defecating closer to the surface (Roman and McCarthy, 2010).

3.4 Pollination

An estimated 87.5 percent of all flowering-plant species are pollinated by animals (Ollerton, Winfree and Tarrant, 2011). Crops at least partially pollinated by animals account for 35 percent of global human food production (Klein *et al*., 2007) and are particularly significant in the supply of micronutrients such as vitamin C, folate and vitamin A (Rose *et al*., 2016). Gallai *et al*. (2009) estimated the value of pollination services to be approximately EUR 153 billion per year globally. Bees – including both managed honey bees and wild species – are generally the main providers of pollination services. Other insects, birds, bats and some other animals also contribute.

While farmers in intensive systems often rent managed honey bees to pollinate their crops, many farmers are reliant on wild pollinators. Moreover, it has been shown that pollination services are enhanced by the presence of wild insects even where honey bees are abundant (Garibaldi *et al*., 2013). Both higher pollinator density and higher species diversity of pollinator visits to flowers have been found to be associated with higher crop yields (Garibaldi *et al*., 2016). Species diversity among pollinators can also be important in buffering the supply of pollination services against the effects of fluctuations in the populations of individual species (Kremen, Williams and Thorp, 2002).

Pollinator density and diversity depend, in turn, on the characteristics of the local environment, including the state of its biodiversity, and on management practices within agriculture, forestry, livestock keeping, etc. Declines in bee populations can occur, *inter alia*, as a result of pesticide use, loss of floral diversity in the local area or the effects of bee parasites (Goulson *et al*., 2015). Pollination services can potentially be promoted by managing crop fields and surrounding habitats to provide bees and other pollinators with resources such as food and water, nesting sites and nesting materials (Altieri *et al*., 2015; Frimpong *et al*., 2011; Pereira *et al*., 2015; Sheffield, Ngo and Azzu, 2016; Varah *et al*., 2013).

3.5 Pest and disease regulation

Pest- and disease-control services are provided by a range of species, including predators, parasitoids and herbivores. These species can include those that occur naturally in the local area (referred to as "natural biological control agents") and those that are deliberately introduced in order to control a problem species. In the latter case, the species may be introduced permanently into the local ecosystem or be applied, as and when required, to a specific target such as the plants growing in an individual greenhouse (Cock *et al*., 2011). Permanent introduction of a natural enemy is referred to as "classical biological control" and is typically used against exotic pests that cannot be effectively controlled by locally occurring natural enemies. The practice of introducing biological control agents directly onto a target crop during a specific cropping cycle is known as "augmentative biological control" (ibid.).

Advantages of biological control over other pest-control methods include the absence of toxic effects on humans and other species. Natural biological control agents and classical biological control agents (once they have become established) often provide services at little or no direct cost to producers. However, the effectiveness of these services depends on the capacity of the local ecosystem to maintain the relevant natural-enemy species in sufficient numbers. This capacity can be diminished by human actions, for example by habitat destruction (Letourneau, 1998) or inappropriate use of pesticides (Ruberson, Nemoto and Hirose, 1998), but it is also possible to manage habitats so as to increase their suitability for natural enemies (Ferro and McNeil, 1998; Gurr, van Emden and Wratten, 1998).

In addition to the contributions provided by associated-biodiversity species such as insects, bats and wild birds, biological control services can also be provided by domesticated crops and livestock and by species used in forestry, aquaculture and fisheries. Examples involving livestock include the use of ducks to control pests in rice fields (Men, Ogle and Lindberg, 2002; Teo, 2001) and chickens to control ticks (Dreyer, Fourie and Kok, 1997). Various kinds of grazing livestock are used to control invasive plant species (FAO, 2015a; Silliman *et al*., 2014). Livestock grazing can also contribute to the control of some crop pests (Hatfield *et al*., 2007; Umberger, 2009). Farmed and wild fish can also contribute to the control of crop pests, for example in rice–fish (Halwart and Gupta, 2004) and water chestnut–fish (Gosh *et al*., 2016) production systems. Other examples of the use of fish in biological control include the introduction of grass carp to feed on invasive aquatic plants and the introduction of gold fish, tilapia and mosquito fish to prey on mosquito eggs and larvae (Bartley and Casal, 1998; Gozlan, 2008). Cover crops can be used to control weeds via the effects of shading or competition for space and nutrients or through the release of chemicals (allelopathic substances) that are harmful to the weeds (Lemessa and Wakjira, 2015; Teasdale, 2003).

Diversity among biological control agents may increase the effectiveness of service provision. For example, Letourneau *et al*. (2009) report that in the majority of cases studied diverse natural-enemy communities provided more effective biological control than less-diverse ones. Reasons for this may include complementary effects. For example, different natural enemies may attack a pest at different stages in its life cycle or on different parts of the plant (Rocca and Messelink, 2017). In some cases, however, increased natural-enemy diversity does not benefit pest control. Effects can even be negative, for example if predators prey on each other as well as on the pests (Finke and Denno, 2004). From a longer-term perspective, the presence of a number of different natural enemy species may increase the likelihood that biological control services will be able to continue if individual species are lost or their populations decline (Cock *et al*., 2011). The diversity of natural enemies is, in turn, dependent on the presence of a range of other species within the local ecosystem – plants that provide shelter, prey species that provide alternative sources of food, and so on. In addition to providing ongoing biological control services, diverse ecosystems that support a range of natural-enemy species also provide a potential source of new species for use in future biological control strategies (ibid).

3.6 Water purification and wastewater treatment

The role of biodiversity in the supply of freshwater is discussed above in Section 2.3. Many of the same processes contribute to the decontamination of water after it has been used. To a large degree, these services can be regarded as two sides of the same coin: waste-water treatment often delivers freshwater that can potentially be used again by downstream users. However, water-treatment services can also benefit saltwater and brackishwater ecosystems and waters flowing into them. These ecosystems themselves, and the biodiversity within them, also provide water purification services.

Various components of marine biodiversity, including some that are raised in marine and coastal aquaculture, are recognized for their roles in water purification. In several parts of the world, efforts to restore oyster beds are partly motivated by their role in the supply of water-purification services via filter feeding (Coen *et al*., 2007; Grabowski and Peterson, 2007; Lindahl *et al*., 2005; NOAA, 2018; zu Ermgassen *et al*., 2013). Both wild and farmed shellfish can remove excess carbon, nitrogen and other nutrients from the water and incorporate them into their bodies or deposit them in sediments (Higgins, Stephenson and Brown, 2011; Rice, 2001). Marine seaweed also provides water purification services in coastal waters (Chopin *et al*., 2001; Neori *et al*., 2007; Smale *et al*., 2013). So-called integrated multitrophic level aquaculture takes advantage of the capacities of certain farmed species (e.g. seaweed) to utilize waste discharged by others (e.g. fish or shrimp) (Troell, 2009).

3.7. Habitat provisioning

A habitat is a location that can serve as the home of a particular species. It provides the species with conditions in which it can survive and reproduce, for example temperatures within a given range and access to appropriate feed resources in sufficient quantities. Without suitable habitat, a species cannot survive and can play no role in the supply of ecosystem services.

TEEB (2010) highlights two aspects of habitat ecosystem services that can be regarded as particularly significant. The first of these is the provision of habitats that enable migratory species to maintain their life cycles, for example feeding and roosting sites for migratory birds and free-flowing rivers for migratory fishes. Providing services of this kind means that an ecosystem helps to support the beneficial roles played by the respective species at distant locations. The second aspect is the maintenance of genetic diversity and in particular the maintenance of habitats that serve as “hotspots” of biodiversity.

Biodiversity is both a contributor to and a product of habitat services. All species rely on others, some directly as sources of food, shelter, pollination, etc., and others indirectly via ecosystem functions that help to keep the environment habitable, for example preventing soil erosion or regulating water flow and quality.

The habitat impacts of food and agricultural production systems are very diverse. In some cases, industrial livestock units for example, production environments are almost totally human-controlled and whatever natural or semi-natural habitats may previously have been present are more or less obliterated. In others, habitats are transformed to create favourable conditions for particular species (crops, livestock, trees, etc.) but still retain semi-natural elements – in the soil, at field margins, etc. In yet others, production takes place largely in semi-natural habitats – grazed rangelands or managed forests for example – altered to varying degrees by the introduction of domesticated species or other management interventions. Finally, there are systems in which products are extracted from essentially “wild” ecosystems such as oceans and unmanaged forests.

None of these production activities leaves natural habitats in their pristine forms, although well-managed fisheries may have negligible impact. Land-use and water-use changes associated with agriculture and food production are recognized as major drivers of habitat loss (CBD, 2010). It is therefore tempting to regard food and agricultural systems purely as purveyors of habitat disservices rather than habitat services. This is, however, not always the whole story. For example, some wild species are now heavily dependent on particular agricultural habitats, and can be threatened not only by the intensification of production systems but also by their abandonment (e.g. Dyulgerova *et al*., 2015; Zakkak *et al*., 2014). Domesticated species raised in semi-natural ecosystems sometimes provide particular habitat-maintenance services that are difficult to replace. Out of 231 habitat types listed in Annex 1 of the European Union's Habitats Directive (EU, 1992), 63 are considered either to be fully or partly dependent on agricultural practices (Halada *et al*., 2011). In most cases, the habitat-benefiting effects are a result of grazing or mowing (ibid.) and are therefore mainly associated with livestock production, although some wild species can be affected by the loss of arable fields (Brambilla, Guidali and Negri, 2009; Nikolov *et al*., 2011). In many places, the abandonment of agriculture leads to the spread of scrub and woodland – habitats that have their own value as wildlife habitats and in the supply of other ecosystem services. However, the process may threaten biodiverse habitat mosaics created by traditional agriculture, and species that rely on open habitats such as meadows.

So-called conservation grazing – the intentional use of grazing animals such as cattle, sheep and horses to maintain vegetation in a state that provides suitable habitat for particular kinds of wildlife – has become a widespread practice, particularly in Europe (Woodland Trust, 2012). These services depend on the availability of animals that are well adapted to local climates, terrains and forage resources. Moreover, different species – and to some extent different breeds – have different feeding habits that can be used to achieve different objectives in habitat management (GAP, 2009). Sometimes, a mixed herd or flock of animals will provide better habitat management services than a single species or breed (e.g. Loucougaray, Bonis and Bouzillé, 2004). Positive impacts of grazing on targeted components of biodiversity have been observed, for example, in sand-dune habitats (Plassmann, *et al*., 2010), marshlands (Mérő, Lontay and Lengyel, 2015) and grasslands (Faria, Rabaça and Morales, 2012). Care must, however, be taken to avoid a one-size-fits-all approach, as grazing has been found to be harmful to some types of wildlife (e.g. Jofre and Reading, 2012; Sharps *et al*., 2015). More generally, the relationship between livestock grazing and habitat services depends on the characteristics of the local ecosystem, the types of animals grazed and how the grazing is managed, with impacts on wild populations varying across taxonomic groups and species (Schieltz and Rubenstein, 2016). Effects can extend beyond the grazed area itself. For example, overgrazing has been shown to negatively affect riparian communities and to have caused the loss of important aquatic species in some areas (Armour, Duff and Elmore, 1991).

There have been some cases in which habitat services have only come to light after livestock are removed from a site. For example, when cattle and water buffalo were banned from the Keoladeo National Park in India, a wintering site for the rare Siberian crane (*Leucogeranus leucogeranus*), an aquatic grass species formerly fed upon by the livestock became more abundant and made it difficult for the cranes to dig up the plant tubers they use for food, leading to a decline in their population (Pirot, Meynell and Elder, 2000). The disappearance of the large blue butterfly in the United Kingdom provides another example. The large blue is a brood parasite, i.e. it operates a cuckoo-like strategy of tricking other species into raising its young. The hosts in this case are certain types of ants, the most suitable being *Murmica sabuleti*, a species that thrives on closely grazed pastures. A decline in sheep numbers led to a decline in the *M. sabuleti* population and in turn to the local extinction of the large blue (Thomas, 1980).

Some crop and mixed production systems include a diverse range of cultivated species, which in turn contributes to their roles as habitats. For example, in many parts of the tropics people maintain highly diverse home gardens that serve as sources of food, medicines, ornamentally and culturally important plants, fuel, fodder and other products. In places, these gardens serve as refuges for native wild plants that are threatened by habitat loss in the wider landscape (Hemp, 2006; Larios *et al*., 2013; Webb and Kabir, 2009). For example, coffee plants in home gardens in Ethiopia have been found to be important habitats for a range of rainforest epiphytic species (Hylander and Nemomissa, 2008). Home gardens can be important to the survival of crop wild relatives (Salako *et al*., 2014) and in the maintenance of within-species genetic diversity in wild species (Gao, He and Li, 2012). Relatively modest increases in the diversity of the crops grown within a production system – for example through practices such as crop rotation and intercropping – can also contribute to habitat diversity and to more wildlife-friendly agriculture (Mineau and Mclaughlin, 1996; Sokos *et al*., 2013; Vignesh, Maheswari and Doraisamy, 2014). Likewise, mixed-species forest plantations tend to provide greater habitat diversity than monocultures (e.g. Felton *et al*., 2016; Hartley, 2002). Many forest and aquatic production systems are highly diverse habitats. A number of forest and marine zones are recognized among the world's biodiversity hotspots (CEPF, 2020). Although fish farmers do not appreciate it, bird numbers sometimes increase around aquaculture ponds because of the extra supply of easily accessible food (Fleury and Sherry, 1995).

Associated biodiversity of all kinds contributes to the supply of habitat services. In the soil, for example, earthworms, ants, termites and small mammals create habitats for smaller organisms by creating durable soil aggregates and pores (Turbé *et al*., 2010). These structures can become hotspots for microbial activities that in turn underpin habitat services (and other ecosystem services) on a larger scale. The roles of plants in providing habitat for useful species such as pollinators and biological control agents are noted in the respective sections above. In the marine environment, kelp beds, although usually dominated by a few major keystone species, support a range of organisms that depend on the seaweeds for shelter, food and substrate (Graham, 2004). Coral reefs, mangroves and seagrass meadows play vital roles in providing habitats for an enormous variety of species (e.g. Barbier *et al*., 2011; Knowlton *et al*., 2010; Saenger, Gartside and Funge-Smith, 2013; UNEP-WCMC, 2014).

4. Cultural and amenity services

Cultural and amenity services are the aesthetic, recreational, inspirational, spiritual and educational benefits that people obtain from direct or indirect contact with ecosystems (TEEB, 2010). Biodiversity has a major influence on the aesthetic appearance of many ecosystems, their capacity to inspire, their suitability for various recreational activities and their educational significance. Some cultural activities depend directly on the presence of particular species or a significant level of species diversity, for example various wildlife-watching activities and recreational fishing. In other cases, characteristic species or biological communities add to the particular aesthetic and inspirational qualities of a local landscape.

Many cultural ecosystem services are associated with ecosystems that are not used for food and agriculture. However, agricultural production systems and their biodiversity – both non-domesticated and domesticated – also contribute to the supply of these services. This is the case, for example, for many culinary traditions, which are often linked to local products and may depend on particular local species, varieties or breeds of crops, livestock, trees or fish. The same is true for a variety of non-food products made from wood, plant and animal fibres, skins, feathers, shells, bones or horns.

Farming, livestock-keeping, forest and fishing communities are often guardians of a range of traditional knowledge related to the characteristics and management of their local BFA. In addition to its practical value, this knowledge is part of the cultural heritage of the local area (and of society more broadly). Many myths, legends and folktales, songs, dances, poems and artistic traditions are similarly linked to local BFA (MEA, 2005b). The survival of these cultural traditions is not necessarily dependent on the ongoing presence of respective components of biodiversity. However, if the biodiversity is lost, the traditions may lose much of their imaginative power and artists lose sources of ongoing inspiration.

Particular plants and animals or products obtained from them are important elements in many cultural and religious events and festivals. Gardening and raising small livestock such as pigeons and rabbits are widely pursued as leisure activities, and in some places larger-scale hobby farming is popular. Pets and companion animals of various kinds, including aquarium species, are also widely popular. Horses and other animals are used in various sports.

Agricultural, pastoral, wetland and forest landscapes are often valued for their aesthetic qualities, their cultural significance or as sites for recreational activities. Particular kinds of crops, fish, trees, livestock or components of associated biodiversity, such as wild farmland birds or flowers, may be vital to the "sense of place" associated with a given location (Hausmann *et al*., 2016). Grazing livestock can play a major role in shaping the local vegetation and hence the character of semi-natural landscapes.

There is clearly a degree of subjectivity in judgements about what kind of landscape is preferable. Arguments can be made in support of more naturalistic or "rewilded" landscapes. However, in many places landscapes shaped by agriculture are highly regarded by both local people and tourists (e.g. Ciaian and Gomez y Paloma, 2011; Zander *et al*., 2013). Landscape-related ecosystem services are often best provided by locally adapted plants and animals, both because of their ability to cope with local conditions and because of their links to local culture.

In addition to the intangible benefits that they provide in terms of recreation, aesthetic appreciation, inspiration and so forth, cultural ecosystem services can also have measurable positive effects on human health. Studies have shown that access to green spaces can produce benefits in terms of, *inter alia*, cardiovascular, reproductive and mental health (CBD and WHO, 2015; Dadvand *et al*., 2012; Mitchell and Popham, 2008; Tamosiunas *et al*., 2014).

The significance of biodiversity to these effects (e.g. whether more diverse ecosystems have a greater impact), and the mechanisms involved, however, remain quite poorly understood (CBD and WHO, 2015). As well as possible direct effects on psychological well-being, interaction with nature may promote a more physically active lifestyle and also bring people into contact with diverse environmental microbiota, a factor increasingly considered to have a significant influence on health (ibid.).[25] In some countries, there is growing interest in "care farming", the use of commercial farms, agricultural landscapes and everyday farming activities for therapeutic or health-promoting purposes, as well as in approaches such as animal and horticultural therapy (Elings, 2012; Hassink, 2003; Hine, Peacock and Pretty, 2008).

Cultural ecosystem services also create significant economic opportunities in fields such as tourism (including – in the context food and agriculture – farm holidays and visits to areas with historical or scenic farming or forest landscapes) and recreational fishing and hunting. Recreational fishing is a multibillion dollar industry largely practised in developed countries, but gaining popularity in developing countries as well (Arlinghaus and Cook, 2009). FAO has published guidelines to help ensure that it is done responsibly (FAO, 2012).

---

[25] Links between human microbiota and health are beyond the scope of this study. However, it is interesting to note that there is some evidence that exposure to farms and farmland, as well as to farm animals and dogs, can help people acquire microbial diversity that can protect against allergic disorders (CBD and WHO, 2015).

5. Conclusion

The examples presented in this short overview illustrate the wide range of ecosystem services provided by BFA. They also show that the benefits that a given food and agricultural production unit (i.e. farm, fish farm, forest stand, fishery or livestock holding) gains from biodiversity generally come both from within and from outside the production unit. These services are supplied, and made more resilient, by a diverse range of interacting components of biodiversity, often including those that are used in or associated with other production units (including those in other sectors of food and agriculture) and those found on land or in waters not used for food and agriculture. It follows, similarly, that flows of benefits to one production unit can be disrupted by events, including those associated with human management or mismanagement, in others and in the wider landscape or seascape. These interactions point to the need for a more integrated management of production units and their surroundings, at least at landscape (or seascape) scale. The examples also show that biodiversity present in and around food and agricultural production systems often provides ecosystem services whose benefits are felt far beyond the food and agriculture sector (and in some cases far away in geographical terms). While there are potential "win–win" scenarios in the management of BFA for ecosystem services, there will inevitably be cases where there are trade-offs in terms of who benefits or loses out. Efforts need to be made to develop equitable ways of addressing such issues, as well as to facilitate cooperation in the implementation of mutually beneficial actions.

Assessing the significance of diversity *per se* to the capacity of BFA to supply ecosystem services is often difficult. However, experimental evidence and theoretical considerations suggest that biological communities that are more diverse at species or within-species level will often be more effective or more resilient suppliers of ecosystem services. Diversity also provides the basis for adapting production systems to future challenges to the supply of ecosystem services.

References

Adams, L.G., Farley, S.D., Stricker, C.A., Demma, D.J., Gretchen, H., Miller, D.C., Rye, R.O., Stricker, A. & Miller, C. 2010. Are inland wolf — ungulate systems influenced by marine subsidies of Pacific salmon ? *Ecological Applications*, 20(1): 251–262.

Adarme-Vega, T.C., Lim, D.K.Y., Timmins, M., Vernen, F., Li, Y. & Schenk, P.M. 2012. Microalgal biofactories: a promising approach towards sustainable omega-3 fatty acid production. *Microbial Cell Factories*, 11: 96.

Alkama, R. & Cescatti, A. 2016. Biophysical climate impacts of recent changes in global forest cover. *Science*, 351(6273): 600–604.

Allen, D.C., Cardinale, B.J. & Wynn-Thompson, T. 2016. Plant biodiversity effects in reducing fluvial erosion are limited to low species richness. *Ecology*, 97(1): 17–24.

Altieri, M.A., Nicholson, C.I., Gillespie, M., Waterhouse, B., Wratten, S., Gbehounou, G. & Gemmill-Herren, B. 2015. *Crops, weeds and pollinators. Understanding ecological interaction for better management*. Rome, FAO. (available at http://www.fao.org/3/a-i3821e.pdf).

Angers, D.A. & Caron, J. 1998. Plant-induced changes in soil structure: Processes and feedbacks. *Biogeochemistry*, 42(1–2): 55–72.

Arigo, K.R. 2005. Marine microorganisms and global nutrient cycles. *Nature*, 437:349–355.

Arlinghaus, R. & Cook, S.J. 2009. Recreational fisheries: socioeconomic importance, conservation issues and management challenges. *Recreational hunting, conservation and rural livelihoods: science and practice*, pp. 39–58. Oxford, UK, Wiley-Blackwell.

Armour, C.L., Duff, D.A. & Elmore, W. 1991. The effects of livestock grazing on riparian and stream ecosystems. *Fisheries*, 16: 7–11.

Barbier, E.B. 2014. Valuing the storm protection service of estuarine and coastal ecosystems. *Ecosystem Services*, 11: 32–38

Barbier, E.B., Hacker, S.D., Kennedy, C., Koch, E.W., Stier, A.C. & Silliman, B.R. 2011. The value of estuarine and coastal ecosystem services. *Ecological Monographs*, 81(2): 169–193.

Barrios, E. 2007. Soil biota, ecosystem services and land productivity. *Ecological Economics*, 64(2): 269–285.

Bartley, D. & Casal, C.V. 1998. Impacts of introductions on the conservation and sustainable use of aquatic biodiversity. *FAO Aquaculture Newsletter*, 20: 15–19. (available at http://www.fao.org/tempref/FI/CDrom/aquaculture/a0805e/documents/article_Bartley_Casal.pdf).

Battin, T.J., Luyssaert, S., Kaplan, L.A., Aufdenkampe, A.K. Richter, A. & Tranvik, L.J. 2009. The boundless carbon cycle. *Nature Geoscience*, 2: 598–600.

Beattie, A., Hay, M., Magnusson, B., de Nys, R., Smeathers, J. & Vincent, J.F.V. 2011. Ecology and bioprospecting. *Austral Ecology*, 36(3): 341–356.

Beech, E., Rivers, M., Oldfield, S. & Smith, P.P. 2017. Global Tree Search: the first complete global database of tree species and country distributions. *Journal of Sustainable Forestry*, 36(5): 454–489.

Beed, F., Benedetti, A., Cardinali, G., Chakraborty, S., Dubois, T., Garrett, K. & Halewood, M. 2011. *Climate change and micro-organism genetic resources for food and agriculture: state of knowledge, risks and opportunities*. Rome, FAO. (available at http://www.fao.org/3/mb392e/mb392e.pdf).

Bell, J., Ganachaud, A., Gehrke, P., Hobday, A., Hoegh-Guldberg, O., Johnson, J., Le Borgne, R., *et al.* 2013. Vulnerability of fisheries and aquaculture to climate change in Pacific Island countries and territories. *In* J. Johnson, J. Bell & C. De Young, eds. *Priority adaptations to climate change for Pacific fisheries and aquaculture: reducing risks and capitalizing on opportunities*, pp. 25–100. FAO/Secretariat of the Pacific Community Workshop, Noumea, New Caledonia, 5–8 June 2012. FAO Fisheries and Aquaculture Proceedings No. 28. Rome. (available at http://www.fao.org/3/i3159e/i3159e.pdf).

Bharucha, Z. & Pretty, J. 2010. The roles and values of wild foods in agricultural systems. *Philosophical Transactions of the Royal Society B: Biological Sciences*, 365(1554): 2913–2926.

Bilota, G.S., Brazier, R.E. & Haygarth, P.M. 2007. The impacts of grazing animals on the quality of soils, vegetation, and surface waters in intensively managed grasslands. *Advances in Agronomy*, 94: 237–280.

Bowler, D.E., Mant, R., Orr, H., Hannah, D.M. & Pullin, A.S. 2012. What are the effects of wooded riparian zones on stream temperature? *Environmental Evidence*, 1(1): 1–9.

Brambilla, M., Guidali, F. & Negri, I. 2009. Breeding-season habitat associations of the declining Corn Bunting Emberiza calandra – A potential indicator of the overall bunting richness. *Ornis Fennica,* 86(2): 41–50.

Burlingame, B., Charrondiere, R. & Mouille, B. 2009. Food composition is fundamental to the cross-cutting initiative on biodiversity for food and nutrition. *Journal of Food Composition and Analysis*, 22: 361–365.

Capstaff, N.M. & Miller, A.J. 2018. Improving the yield and nutritional quality of forage crops. *Frontiers in Plant Science*, 9: 535.

Cardinale, B. 2011. Biodiversity improves water quality through niche partitioning. *Nature*, 472: 86–89.

Cardinale, B.J., Duffy, J.E., Gonzalez, A., Hooper, D.U., Perrings, C., Venail, P., Narwani, A., *et al.* 2012.Biodiversity loss and its impact on humanity. *Nature*, 486(7401): 59–67.

CBD (Convention on Biological Diversity). 2010. *Global Biodiversity Outlook 3.* Montreal, Canada, Secretariat of the Convention on Biological Diversity.

CBD & WHO. 2015. *Connecting global priorities: biodiversity and human health, a state of knowledge review*. C.D. Romanelli, C. Cooper, D. Campbell-Lendrum, D. Maiero, M. Karesh, W.B. Hunter, & D. Golden, eds. Geneva, Switzerland, WHO, and Montreal, Canada, Secretariat of the Convention on Biological Diversity.

Cederholm, C.J., Kunze, M.D., Murota, T. & Sibitan, A. 1999. Criteria to define extinction risk in marine fishes. *Fisheries*, 24(10): 6–15.

Chang, Q., Wang, L., Ding, S., Xu, T., Li, Z., Song, X., Zhao, X., Wang, D. & Pan, D. 2018. Grazer effects on soil carbon storage vary by herbivore assemblage in a semi-arid grassland. *Journal of Applied Ecology*, 55(5): 2517–2526.

Chatzipavlidis, I., Kefalogianni, I., Venieraki, A. & Holzapfel, W. 2013. *Status and trends of the conservation and sustainable use of microorganisms in agroindustrial processes*. Commission on Genetic Resource for Food and Agriculture. Background Study Paper No. 64. Rome, FAO. (available at http:// www.fao.org/docrep/meeting/028/mg339e.pdf).

Chen, S-L., Yu, H., Luo, H-M., Wu, Q., Li, C-F. & Steinmetz, A. 2016. Conservation and sustainable use of medicinal plants: problems, progress, and prospects. *Chinese Medicine*, 11: 37.

Chopin, T., Yarish, C., Neefus, C., Kraemer, G., Zertuche-Gonzalez, J., Belyea, E. & Carmona, R. 2001. Aquaculture from a different angle: the seaweed perspective, and the rationale for promoting integrated aquaculture. *In* M. Tlusty, D. Bengtson, H.O. Halvorson, S. Oktay, J. Pearce & R.B.J. Rheault, eds. *Marine aquaculture and the environment,* pp. 69–72. Falmouth, USA, Cape Cod Press.

Ciaian, P. & Gomez y Paloma, S. 2011. *The value of EU agricultural landscape*. Seville, Spain, European Commission Joint Research Centre Institute for Prospective Technological Studies.

Ciesla, W.M. 2002. *Non-wood forest products from temperate broad-leaved trees*. Non-Wood Forest Products No. 15. Rome, FAO. (available at http://www.fao.org/3/y4351e/y4351e00.htm).

Cock, M.J.W., Biesmeijer, J.C., Cannon, R.J.C., Gerard, P.J., Gillespie, D., Jiménez, J.J., Lavelle, P.M. & Raina, S.K. 2011. *Climate change and invertebrate genetic resources for food and agriculture: state of knowledge, risks and opportunities*. Commission on Genetic Resources for Food and Agriculure. Background Study Paper No. 54. Rome, FAO. (available at http://www.fao.org/docrep/meeting/022/mb390e.pdf).

Coen, L.D., Brumbaugh, R.D., Bushek, D., Grizzle, R., Luckenbach, M.W., Posey, M.H., Powers, S.P. & Tolley, S.G. 2007. Ecosystem services related to oyster restoration. *Marine Ecology Progress Series*, 341: 303–307.

Colfer, C.J.P., Sheil, D. & Kishi, M. 2006. *Forests and human health: assessing the evidence*. CIFOR Occasional Paper No. 45. Bogor, Indonesia, Center for International Forestry Research

Corsi, S., Friedrich, T., Kassam, A., Pisante, M. & Sà, J. de M. 2012. *Soil organic carbon accumulation and greenhouse gas emission reductions from conservation agriculture: a literature review*. Rome, FAO. (available at http://www.fao.org/fileadmin/user_upload/agp/icm16.pdf).

Coutteau, P., Geurden, I., Camara, M.R., Bergot, P. & Sorgeloos, P. 1997. Review on the dietary effects of phospholipids in fish and crustacean larviculture. *Aquaculture*, 155(1–4): 149–164.

CPP. 1999. *Native shelterbelts. Benefits for wildlife.* Voluntary Conservation on Private and Public Land. Note 7. Huntsville, Australia, Conservation Partners Programme, NSW National Parks and Wildlife Service.

Crews, J. 2003. Forest and tree symbolism in folklore. *Unasylva*, 54(213): 37–44.

CEPF (Critical Ecosystem Partnership Fund). 2020. *The biodiversity hotspots* [Cited 20 July 2020]. https://www.cepf.net/our-work/biodiversity-hotspots

Dadvand, P., de Nazelle, A., Figueras, F., Basagaña, X., Su, J., Amoly, E., Jerrett, M., Vrijheid, M., Sunyer, J. & Nieuwenhuijsen, M.J. 2012. Green space, health inequality and pregnancy. *Environment International*, 40(1): 110–115.

Dawson, I.K., Attwood, S.J., Park, S.E., Jamnadass, R., Powell, W., Sunderland, T., Kindt, R., *et al.* 2019. *Contributions of biodiversity to the sustainable intensification of food production*. Thematic study prepared for The State of the World's Biodiversity for Food and Agriculture. Rome.

Dębska, B., Długosz, J., Piotrowska-Długosz, A. & Banach-Szott, M. 2016. The impact of a bio-fertilizer on the soil organic matter status and carbon sequestration—results from a field-scale study. *Journal of Soils and Sediments*, 16(10): 2335–2343.

de Deckere, E.A. 2001. Health aspects of fish and n-3 polyunsaturated fatty acids from plant and marine origin. *In* T. Wilson & N.J. Temple, eds. *Nutritional health*, pp. 195–206. New York City, USA, Springer Science+Business Media.

Donia, M. & Hamann, M.T. 2003. Marine natural products and their potential applications as anti-infective agents. *The Lancet*, 3(June): 338–348.

Donovan, R.G., Mackenzie, A.R. & Hewitt, C.N. 2005. Development and application of an urban tree air quality score for photochemical pollution episodes using the Birmingham, United Kingdom, area as a case study. *Environmental Science and Technology*, 39(17): 6730–6738.

Downing, J.A. 2010. Emerging global role of small lakes and ponds: little things mean a lot. *Limnetica*, 29(1): 9–24.

Dreyer, K., Fourie, L.J. & Kok, D.J. 1997. Predation of livestock ticks by chickens as a tick-control method in a resource-poor urban environment. Onderstepoort *Journal of Veterinary Medicine,* 63(4): 273–276.

Durán Zuazo, V.H. & Rodríguez Pleguezuelo, C.R. 2008. Soil-erosion and runoff prevention by plant covers. A review. *Agronomy for Sustainable Development*, 28(1) 65–86.

DuVal, A., Mijatovic, D. & Hodgkin, T. 2019. *The contribution of biodiversity for food and agriculture to resilience of production systems to environmental change and uncertainty*. Thematic study prepared for *The State of the World's Biodiversity for Food and Agriculture*. Rome, FAO. (avaliable at http://www.fao.org/3/ca5008en/ca5008en.pdf).

Dyulgerova, S., Gramatikov, M., Pedashenko, H., Vassilev, K., Kati, V. & Nikolov, S.C. 2015. Farmland birds and agricultural land abandonment: Evidences from Bulgaria. *Acta Zoologica Bulgarica,* 67(2): 223–234.

Edwards, P. 2015. Aquaculture environment interactions: Past, present and likely future trends. *Aquaculture*, 447: 2–14.

Elings, M. 2012. *Effects of Care Farms. Scientific research on the benefits of care farms for clients.* Wageningen, the Netherlands, Plant Research International.

EU (European Union). 1992. Council Directive 92/43/EEC of 21 May 1992 on the conservation of natural habitats and of wild fauna and flora. *Official Journal of the European Communities*, L206: 7–50.

Eurostat, FAO, ITTO & UNECE. 2018. *Joint forest sector questionnaire definitions 2018.* (available at http://www.fao.org/forestry/7800-0aded052ed8904ee31f045d5a3f79ae1d.pdf).

Fabre, P., Guérin, G. & Bouquet, P.M. 2010. *The management of natural areas by pastoralism. A French mediterranean example: the Merinos d'Arles in transhumance*. Paper presented at the Eighth World Merino Conference, May 3–5, 2010, Rambouillet, France.

FAO. 1997. *The State of the World's Plant Genetic Resources for Food and Agriculture*. Rome. (available at http://www.fao.org/3/a-w7324e.pdf).

FAO. 1999. FAO forestry. Towards a harmonized definition of non-wood forest products. *Unasylva*, 50(198): 63–64. (available at http://www.fao.org/3/x2450e/x2450e0d.htm).

FAO. 2003. *Word agriculture towards 2015/2030. An FAO perspective*. J. Bruinsma, ed. London, Earthscan. (available at http://www.fao.org/3/a-y4252e.pdf).

FAO. 2005. *The importance of soil organic matter. Key to drought-resistant soil and sustained food production*. A. Bot & J. Benites, eds. Rome. (available at http://www.fao.org/3/a0100e/a0100e00.htm).

FAO. 2008a. *Aquaculture development 3. Genetic resource management*. FAO Technical Guidelines for Responsible Fisheries No. 5, Suppl. 3. Rome. (available at http://www.fao.org/3/a-i0283e.pdf).

FAO. 2008b. *The State of Food and Agriculture 2008. Biofuels: prospects, risks and opportunities*. Rome. (available at http://www.fao.org/catalog/inter-e.htm).

FAO. 2009. T*he State of Food and Agriculture 2009. Livestock in the balance*. Rome. (available at http://www.fao.org/3/a-i0680e.pdf).

FAO. 2010a. *The Second Report on the State of the World's Plant Genetic Resources for Food and Agriculture*. Rome (available at http://www.fao.org/3/i1500e/i1500e00.htm).

FAO. 2010b. *Global Forest Resources Assessment 2010. Main report*. Rome (available at http://www.fao.org/3/a-i1757e.pdf).

FAO. 2012. *Recreational fisheries*. FAO Technical Guidelines for Responsible Fisheries No. 13. Rome. (available at http://www.fao.org/3/i2708e/i2708e00.htm).

FAO. 2013a. *Guidelines for the preparation of the country reports for The State of the World's Biodiversity for Food and Agriculture*. Rome. (available at http://www.fao.org/3/a-as644e.pdf).

FAO. 2013b. *Climate-smart agriculture sourceboo*k. Rome (available at http://www.fao.org/docrep/018/i3325e/i3325e00.htm).

FAO. 2014a. *The State of Food and Agriculture. Innovation in family farming*. Rome (available at http://www.fao.org/3/a-i4040e.pdf).

FAO. 2014b. *The State of the World's Forest Genetic Resources.* Rome, Commission on Genetic Resources for Food and Agriculture, Food and Agriculture Organization of the United Nations. (available at http://www.fao.org/forestry/fgr/64582/en/).

FAO. 2014c. S*tate of the World's Forests 2014. Enhancing the socioeconomic benefits from forests.* Rome. (available at http://www.fao.org/3/a-i3710e.pdf).

FAO. 2015a. *The Second Report on the State of World's Animal Genetic Resources for Food and Agriculture*. B.D. Scherf & D. Pilling, eds. Rome. (available at http://www.fao.org/3/a-i4787e.pdf).

FAO. 2015b. *The impact of natural hazards and disasters on agriculture and food security and nutrition. A call for action to build resilient livelihoods*. Rome. (available at http://www.fao.org/3/a-i4434e.pdf).

FAO. 2016a. *State of the World's Forests. 2016.* Rome. (available at http://www.fao.org/publications/sofo/2016/en).

FAO. 2016b. *The State of Food and Agriculture 2016. Climate change, agriculture and food systems.* Rome. (available at http://www.fao.org/3/a-i6030e.pdf).

FAO. 2018a. *The State of World Fisheries and Aquaculture 2018*. Meeting the Sustainable Development Goals. Rome. (available at http://www.fao.org/3/i9540en/i9540en.pdf).

FAO. 2018b. *Terms and definitions. FRA 220*. Forest Resources Assessment Working Paper 188. Rome. (available at http://www.fao.org/3/I8661EN/i8661en.pdf).

FAO. 2019. *The State of the World's Aquatic Genetic Resources for Food and Agriculture*. Rome. (available at https://doi.org/10.4060/CA5256EN).

FAO & ITPS. 2015. *Status of the World's Soil Resources (SWSR) – Main Report.* Rome, FAO and ITPS. (available at http://www.fao.org/3/a-i5199e.pdf).

Faria, N., Rabaça, J.E. & Morales, M.B. 2012. The importance of grazing regime in the provision of breeding habitat for grassland birds: The case of the endangered little bustard (*Tetrax tetrax*). *Journal for Nature Conservation*, 20(4): 211–218.

Felton, A., Nilsson, U., Sonesson, J., Felton, A.M., Roberge, J.M., Ranius, T., Ahlström, M., *et al.* 2016. Replacing monocultures with mixed-species stands: Ecosystem service implications of two production forest alternatives in Sweden. *Ambio*, 45: 124–139.

Ferro, D.N. & McNeil, J.N. 1998. Habitat enhancement and conservation of natural enemies of insects. *In* P. Barbosa, ed. *Conservation biological control*, pp. 123–132. London, Academic Press.

Finke, D.L. & Denno, R.F. 2004. Predator diversity dampens trophic cascades. *Nature*, 429(6990): 407–410.

Fleury, B.E. & Sherry, T.W. 1995. Long-term population trends of colonial wading birds in the southern United States: the impact of crayfish aquaculture on Louisiana populations. *Auk*, 112(3): 613–632.

Forest Europe. 2015. *State of Europe's Forests 2015*. Madrid.

Fornara, D.A. & Tilman, D. 2008. Plant functional composition influences rates of soil carbon and nitrogen accumulation. *Journal of Ecology*, 96(2): 314–322.

Frimpong, E.A., Gemmill-herren, B., Gordon, I. & Kwapong, P.K. 2011. Dynamics of insect pollinators as influenced by cocoa production systems in Ghana. *Journal of Pollination Ecology*, 5(10): 74–80.

Gallai, N., Salles, J.M., Settele, J. & Vaissière, B.E. 2009. Economic valuation of the vulnerability of world agriculture confronted with pollinator decline. *Ecological Economics*, 68(3): 810–821.

Gao, J., He, T. & Li, Q.M. 2012. Traditional home-garden conserving genetic diversity: A case study of *Acacia pennata* in southwest China. *Conservation Genetics*, 13(4): 891–898.

GAP (Grazing Animals Project). 2009. *Breed profiles handbook*. Stoneleigh Park, UK, Grazing Animals Project.

Garibaldi, L.A., Steffan-Dewenter, I., Winfree, R., Aizen, M.A., Bommarco, R., Cunningham, S.A., Kremen, C., *et al.* 2013. Wild pollinators enhance fruit set of crops regardless of honey bee abundance. *Science*, 340(6127): 1608–1611.

Garibaldi, L.A., Carvalheiro, L.G., Vaissière, B.E., Gemmill-Herren, B., Hipólito, J., Freitas, B.M., Ngo, H.T., *et al.* 2016. Mutually beneficial pollinator diversity and crop yield outcomes in small and large farms. *Science*, 351(6271): 388–391.

Garnett, T., Godde, C., Muller, A., Röös, E., Smith, P., de Boer, I., zu Ermgassen, E., Herrero, M., van Middelaar, C., Schader, C. & van Zanten, H. 2017. *Grazed and confused. Ruminating on cattle, grazing systems, methane, nitrous oxide, the soil carbon sequestration question – and what it all means for greenhouse gas emissions*. Oxford, UK, Food Climate Research Network, University of Oxford.

Gerber, P.J., Steinfeld, H., Henderson, B., Mottet, A., Opio, C., Dijkman, J., Falcucci, A. & Tempio, G. 2013. *Tackling climate change through livestock – A global assessment of emissions and mitigation opportunities*. Rome, FAO (available at http://www.fao.org/3/a-i3437e.pdf).

Ghosh, S., Chowdhury, S.R., Mohanty, R.K. & Brahmanand, P.S. 2016. Impact of integrated farming of water chestnut and cat fish on livelihood of farmers in seasonal waterlogged areas of Odisha. *Agricultural Economics Research Review*, 29(1): 159–165

Gougoulias, C., Clark, J.M. & Shaw, L.J. 2014. The role of soil microbes in the global carbon cycle: Tracking the below-ground microbial processing of plant-derived carbon for manipulating carbon dynamics in agricultural systems. *Journal of the Science of Food and Agriculture*, 94(12): 2362–2371.

Goulson, D., Nicholls, E., Botías, C. & Rotheray, E.L. 2015. Bee declines driven by combined stress from parasites, pesticides, and lack of flowers. *Science*, 347(6229): 1255957.

Gozlan, R.E. 2008. Introduction of non-native freshwater fish: is it all bad? *Fish and Fisheries*, 9: 106–115.

Grabowski, J.H. & Peterson, C.H. 2007. Restoring oyster reefs to recover ecosystem services. *Theoretical Ecology Series*, 4(C): 281–298.

Graham, M.H. 2004. Effects of local deforestation on the diversity and structure of southern California giant kelp forest food webs. *Ecosystems*, 7(4): 341–357.

Graham, E., Grandy, S. & Thelen, M. 2009. *Manure effects on soil organisms and soil quality.* Emerging Issues in Animal Agriculture. East Lansing, USA, Michigan State University.

Gregory, N.G. 1995. The role of shelterbelts in protecting livestock: a review. *New Zealand Journal of Agricultural Research*, 38(4): 423–450.

Griffiths, B.S., Ritz, K., Bardgett, R.D., Cook, R., Christensen, S., Ekelund, F., Sorensen, S.J., *et al.* 2000. Ecosystem response of pasture soil communities to fumigation-induced microbial diversity reductions: an examination of the biodiversity-ecosystem function relationship. *Oikos*, 90(2): 279–294.

Grivetti, L.E. & Ogle, B.M. 2000. Value of traditional foods in meeting macro- and micronutrient needs:the wild plant connection. *Nutrition Research Reviews*, 13(01): 31.

Gruber, N. 2008. The marine nitrogen cycle: overview and challenges. *In* D.G. Capone, D.A. Bronk, M.R. Mulholland & E.J. Carpenter, eds. *Nitrogen in the marine environment*, Second edition. pp. 1–50. Amsterdam, Elsevier.

Gurr, G.M., van Emden, H.F. & Wratten, S.D. 1998. Habitat manipulation and natural enemy efficiency. Implications for the control of pests. *In* P. Barbosa, ed. *Conservation biological control*, pp. 155–183. London, Academic Press.

Habib, M.A.B., Parvin, M., Huntington, T.C. & Hasan, M.R. 2008. *A review on culture, production and use of spirulina as food for humans and feeds for domestic animals and fish*. FAO Fisheries and Aquaculture Circular No. 1034. Rome, FAO. (available at http://www.fao.org/3/a-i0424e.pdf).

Halada, L., Evans, D., Romão, C. & Petersen, J.E. 2011. Which habitats of European importance depend on agricultural practices? *Biodiversity and Conservation*, 20(11): 2365–2378.

Halwart, M. & Gupta, M. V. 2004. *Culture of fish in rice fields*. Rome, FAO, and Penang, Malaysia, WorldFish Center. (available at http://www.fao.org/3/a-a0823e.pdf).

Harborne, J.B., Baxter, H & Moss, P. eds. 1999. *Phytochemical dictionary: a handbook of bioactive compounds from plants*. Second Edition. Boca Raton, USA, Taylor and Francis.

Harnedy, P.A. & FitzGerald, R.J. 2012. Bioactive peptides from marine processing waste and shellfish: A review. *Journal of Functional Foods*, 4(1): 6–24.

Harris, D.R. 1989. An evolutionary continuum of people– plant interaction. *In* D.R. Harris & G.C. Hillman, eds. *Foraging and farming: the evolution of plant exploitation*, pp. 11–26. London, Unwin Hyman.

Harris, J.R. & Markl, J. 1999. Keyhole limpet hemocyanin (KLH): A biomedical review. *Micron*, 30(6): 597–623.

Hartley, M.J. 2002. Rationale and methods for conserving biodiversity in plantation forests. *Forest Ecology and Management*, 155(1–3): 81–95.

Hassink, J. 2003. Combining agricultural production and care for persons with disabilities: a new role of agriculture and farm animals. *In* A. Cirstovao & L.O. Zorini, eds. *Farming and rural systems research and extension. Local identities and globalisation. Fifth IFSA European Symposium, 8–11 April 2002, Florence,* pp. 332–341. Florence, Italy, Agenzia Regionale per lo Sviluppo e l'Innovazione nel Settore Agro-forestale della Regione Toscane (ARSIA).

Hatfield, P., Goosey, H., Lenssen, A. & Blodgett, S. 2007. *Sheep grazing to manage crop residues, insects and weeds in northern plains grain and alfalfa systems.* Agricultural Innovations Fact Sheet. College Park, USA, Sustainable Agriculture Research and Extension.

Hauer, F.R., Locke, H., Dreitz, V.J., Hebblewhite, M., Lowe, W.H., Muhlfeld, C.C. & Rood, S.B. 2016. Gravel-bed river floodplains are the ecological nexus of glaciated mountain landscapes. *Science Advances,* 2(6): e1600026.

Hausmann, A, Slotow, R. Burns, J.K. & di Minin. 2016. The ecosystem service of sense of place: benefits for human well-being and biodiversity conservation. *Environmental Conservation*, 43(2): 117–127.

Hemp, A. 2006. The banana forests of Kilimanjaro: Biodiversity and conservation of the Chagga homegardens. *Biodiversity and Conservation*, 15(4): 1193–1217.

Herr, D., von Unger, M., Laffoley, D. & McGivern, A. 2017. Pathways for implementation of blue carbon initiatives. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 7(S1): 116–129.

Herrero, M., Wirsenius, S., Henderson, B., Rigolot, C., Thornton, P., Havlík, P., de Boer, I. & Gerber, P.J. 2015. Livestock and the environment: what have we learned in the past decade? *Annual Review of Environment and Resources,* 40: 177–202.

Hickey, G.M., Pouliot, M., Smith-Hall, C., Wunder, S. & Nielsen, M.R. 2016. Quantifying the economic contribution of wild food harvests to rural livelihoods: A global-comparative analysis. *Food Policy*, 62: 122–132.

Hicks, C., Woroniecki, S., Fancourt, M., Bieri, M., Garcia Robles, H., Trumper, K. & Mant, R. 2014. *The relationship between biodiversity, carbon storage and the provision of other ecosystem services: critical review for the forestry component of the International Climate Fund*. Cambridge, UK, UNEP.

Higgins, C.B., Stephenson, K. & Brown, B.L. 2011. Nutrient bioassimilation capacity of aquacultured oysters: quantification of an ecosystem service. *Journal of Environment Quality*, 40(1): 271.

Hilderbrand, G. V., Hanley, T.A., Robbins, C.T. & Schwartz, C.C. 1999. Role of brown bears (*Ursus arctos*) in the flow of marine nitrogen into a terrestrial ecosystem. *Oecologia*, 121(4): 546–550.

Hine, R., Peacock, J. & Pretty, J. 2008. *Care farming in the UK: evidence an opportunities*. Report for the National Care Farming Initiative (UK). Colchester, UK, University of Essex.

Hoffmann, I., From, T. & Boerma, D. 2014. *Ecosystem services provided by livestock species and breeds, with special consideration to the contributions of small-scale livestock keepers and pastoralists*. CGRFA Background Study Paper No. 66. Rome, FAO. (available at http://www.fao.org/3/a-at598e.pdf).

Hristov, A.N., Oh, J., Lee, C., Meinen, R., Montes, F., Ott, T., Firkins, J., *et al.* 2013. M*itigation of greenhouse gas emissions in livestock production – A review of technical options for non-CO_2 emissions*. P.J. Gerber, B. Henderson & H.P.S. Makkar, eds. FAO Animal Production and Health Paper No. 177. Rome, FAO. (available at http://www.fao.org/3/i3288e/i3288e.pdf).

H**uang, Y., Chen, Y., Castro-Izaguirre, N., Baruffol, M., Brezzi, M., Lang, A., Li, Y., *et al.*** 2018. Impacts of species richness on productivity in a large-scale subtropical forest experiment. *Science*, 362(6410): 80–83.

Hurtado, A.Q. 2017. *Thematic Background Study No. 4: Genetic resources for farmed seaweeds.* Subcommittee on Aquaculture. Ninth session, Rome, 24–27 October 2017. Rome, FAO. COFI:AQ/IX/2017/SBD.6. (available at http://www.fao.org/cofi/46056-0e272e19f4b0051d1e1c3b679e5ca8ada.pdf).

Hylander, K. & Nemomissa, S. 2008. Home garden coffee as a repository of epiphyte biodiversity in Ethiopia. *Frontiers in Ecology and the Environment*, 6(10): 524–528.

IPK (Leibniz Institute of Plant Genetics and Crop Plant Research). 2018. *Mansfeld's World Database of Agriculture and Horticultural Crops.* [Cited 25 June 2018]. http://mansfeld.ipk-gatersleben.de/apex/f?p=185:3.

Jofre, G.M. & Reading, C.J. 2012. *An assessment of the impact of conservation grazing on reptile populations.* ARC Research Report 12/01. Bournemouth, UK, Amphibian and Reptile Conservation Trust.

Johns, T. & Sthapit, B.R. 2004. Biocultural diversity in the sustainability of developing country food systems. *Food and Nutrition Bulletin,* 25(2): 13.

Johnson, R. 2018. *Hemp as an agricultural commodity*. (available at www.crs.gov).

Karlik, J.F. & Pittenger, D.R. 2012. *Urban trees and ozone formation: a consideration for large-scale plantings*. Agriculture and Natural Resources Publication 8484. Richmond, USA, University of California.

Kennedy, G.L., Pedro, M.R., Seghieri, C., Nantel, G. & Brouwer, I. 2007. Dietary diversity score is a useful indicator of micronutrient intake in non-breastfeeding Filipino children. *The Journal of Nutrition*, 137(2): 472–477.

Kim, S.K. & Mendis, E. 2006. Bioactive compounds from marine processing byproducts - A review. *Food Research International,* 39(4): 383–393.

Klapproth, J.C. & Johnson, J.E. 2000. *Understanding the science behind riparian forest buffers: effects on water quality*. Virgina Cooperative Extension. Publication 420–151. Blacksburg, USA, Virginia Polytechnic Institute and State University.

Klein, A.-M., Vaissiere, B.E., Cane, J.H., Steffan-Dewenter, I., Cunningham, S.A., Kremen, C. & Tscharntke, T. 2007. Importance of pollinators in changing landscapes for world crops. *Proceedings of the Royal Society B: Biological Sciences*, 274(1608): 303–313.

Knowlton, N., Brainard, R.E., Fisher, R., Moews, M., Plaisance, L. & Caley, M.J. 2010. Coral reef biodiversity. *In* A.D. McIntyre, ed. *Life in the world's oceans: diversity, distribution, and abundance,* pp. 65–74. Oxford, UK, Wiley-Blackwell.

Kremen, C., Williams, N.M. & Thorp, R.W. 2002. Crop pollination from wild bees at risk from agricultural intensification. *PNAS*, 99: 16812–16816.

Laffoley, D. & Grimsditch, G., eds. 2009. T*he management of natural coastal carbon sinks*. Cambridge, UK and Gland, Switzerland, IUCN.

Laffoley, D., Baxter, J. M., Thevenon, F. & Oliver, J., eds. 2014. *The significance and management of natural carbon stores in the open ocean*. Full report. Gland, Switzerland, IUCN (available at https://portals.iucn.org/library/sites/library/files/documents/2014-049.pdf).

Lal, R. 2013a. Intensive agriculture and the soil carbon pool. *Journal of Crop Improvement*, 27(6): 735–751.

Lal, R. 2013b. Soil carbon management and climate change. *Soil Carbon*, 4: 339–361.

Lange, M., Eisenhauer, N., Sierra, C.A., Bessler, H., Engels, C., Griffiths, R.I., Mellado-Vázquez, P.G., *et al.* 2015. Plant diversity increases soil microbial activity and soil carbon storage. *Nature Communications*, 6.

Larios, C., Casas, A., Vallejo, M., Moreno-Calles, A.I. & Blancas, J. 2013. Plant management and biodiversity conservation in Náhuatl homegardens of the Tehuacán Valley, Mexico. *Journal of Ethnobiology and Ethnomedicine*, 9(1).

Lemessa, F. & Wakjira, M. 2015. Cover crops as a means of ecological weed management in agroecosystems. *Journal of Crop Science and Biotechnology*, 18(2): 133–145.

Lenane, R. 2012. *Keeping rivers cool. Getting ready for climate change by creating riparian shade.* Bristol, UK, Environment Agency.

Letourneau, D. 1998. Conservation biology: lessons for conserving natural enemies. *In* P. Barbosa, ed. *Conservation biological control*, pp. 9–38. London, Academic Press.

Letourneau, D.K., Jedlicka, J.A., Bothwell, S.G. & Moreno, C.R. 2009. Effects of natural enemy biodiversity on the suppression of arthropod herbivores in terrestrial ecosystems. *Annual Review of Ecology, Evolution, and Systematic*s, 40(1): 573–592.

Lindahl, O., Hart, R., Hernroth, B., Kollberg, S., Loo, L.-O., Olrog, L., Rehnstam-Holm, A.-S., Svensson, J., Svensson, S. & Syversen, U. 2005. Improving marine water quality by mussel farming: a profitable solution for Swedish society. *AMBIO: A Journal of the Human Environment*, 34(2): 131–138.

Loo, J., Fady, B., Dawson, I., Vinceti, B. & Baldinelli, G. 2011. *Climate change and forest genetic resources: state of knowledge, risks and opportunities.* Commission on Genetic Resources for Food and Agriculture. Background Study Paper No. 56. Rome, FAO. (available at http://www.fao.org/docrep/meeting/023/mb696e.pdf).

Loucougaray, G., Bonis, A. & Bouzillé, J.B. 2004. Effects of grazing by horses and/or cattle on the diversity of coastal grasslands in western France. Bi*ological Conservation*, 116(1): 59–71.

Lovreglio, R., Meddour-Sahar, O. & Leone, V. 2014. Goat grazing as a wildfire prevention tool: A basic review. *IForest*, 7(4): 260–268.

MacLeod, M., Gerber, P., Mottet, A., Tempio, G., Falcucci, A., Opio, C., Vellinga, T., Henderson, B. & Steinfeld, H. 2013. *Greenhouse gas emissions from pig and chicken supply chains – a global life cycle assessment*. Rome, FAO. (available at http://www.fao.org/3/i3460e/i3460e.pdf).

Marshall, M.R., Ballard, C.E., Frogbrook, Z.L., Solloway, I., McIntyre, N., Reynolds, B. & Wheater, H.S. 2013. The impact of rural land management changes on soil hydraulic properties and runoff processes: results from experimental plots in upland UK. *Hydrological Processes,* 28: 2617–2629.

Mata, T.M., Martins, A.A. & Caetano, N.S. 2010. Microalgae for biodiesel production and other applications: A review. *Renewable and Sustainable Energy Reviews,* 14(1): 217–232.

Mateo-Sagasta, J. Marjani Zadeh, S. & Turral, H., eds. 2018. M*ore people, more food, worse water? A global review of water pollution from agriculture.* Rome, FAO. (available at http://www.fao.org/3/CA0146EN/ca0146en.pdf).

Mcsherry, M.E. & Ritchie, M.E. 2013. Effects of grazing on grassland soil carbon: A global review. *Global Change Biology*, 19(5): 1347–1357.

MEA (Millennium Ecosystem Assessment). 2005a. *Ecosystem and human well-being: synthesis.* Washington DC, Island Press.

MEA. 2005b. *Ecosystems and human well-being: current state and trends,* volume 1. Washington DC, Island Press.

Mendonça, R., Müller, R.A., Clow, D. Verpoorter, C.. Raymond, P., Tranvik, L.J. & Sobek, S. 2017. Organic carbon burial in global lakes and reservoirs. *Nature Communictions*, 8: 1694.

Men, B.X., Ogle, R.B. & Lindberg, J.E. 2002. Studies on integrated duck-rice systems in the Mekong Delta of Vietnam. J*ournal of Sustainable Agriculture*, 20(July 2012): 27–40.

Mérő, T.O., Lontay, L. & Lengyel, S. 2015. Habitat management varying in space and time: the effects of grazing and fire management on marshland birds. *Journal of Ornithology*, 156(3): 579–590.

Midgley, G.F., Bond, W.J., Kapos, V., Ravilious, C., Scharlemann, J.P.W. & Woodward, F.I. 2010. Terrestrial carbon stocks and biodiversity: Key knowledge gaps and some policy implications. *Current Opinion in Environmental Sustainability*, 2(4): 264–270.

Miles, L., Dunning, E., Doswald, N. & Osti, M. 2010. *A safer bet for REDD+: review of the evidence on the relationship between biodiversity and the resilience of forest carbon stocks*. Working Paper v2. Multiple Benefits Series No. 10. Prepared on behalf of the UN-REDD Programme. Cambridge, UK, UNEP-WCMC.

Milledge, J.J., Smith, B., Dyer, P.W. & Harvey, P. 2014. Macroalgae-derived biofuel: A review of methods of energy extraction from seaweed biomass. *Energies*, 7(11): 7194–7222.

Mineau, P. & Mclaughlin, A. 1996. Conservation of biodiversity within Canadian agricultural landscapes: Integrating habitat for wildlife. *Journal of Agricultural and Environmental Ethics*, 9(2): 93–113.

Mitchell, R. & Popham, F. 2008. Effect of exposure to natural environment on health inequalities: an observational population study. *The Lancet*, 372(9650): 1655–1660.

Mottet, A., de Haan, C., Falcucci, A, Tempio, B. Opio, C & Gerber, P. 2017. Livestock: on our plates or eating at our tables? A new analysis of the feed/food debate. *Global Food Security*, 14: 1–8.

Moursi, M.M., Arimond, M., Dewey, K.G., Treche, S., Ruel, M.T. & Delpeuch, F. 2008. Dietary diversity is a good predictor of the micronutrient density of the diet of 6- to 23-month-old children in Madagascar. *Journal of Nutrition*, 138(12): 2448–2453.

Nellemann, C., Corcoran, E., Duarte, C.M., Valdés, L., De Young, C., Fonseca, L. & Grimsditch, G. 2009. *Blue carbon. A rapid response assessment.* Arendal, Norway, GRID-Arendal, and Nairobi, UNEP.

Neori, A., Troell, M., Chopin, T., Yarish, C., Critchley, A. & Buschmann, A.H. 2007. The need for a balanced ecosystem approach to blue aquaculture. *Environment*, 49(3): 36–43.

Nielsen, M.R., Meilby, H., Smith-Hall, C., Pouliot, M. & Treue, T. 2018. The importance of wild meat in the global south. *Ecological Econonomics*, 146: 696–705.

Nikolov, S.C., Demerdzhiev, D.A., Popgeorgiev, G.S. & Plachiyski, D.G. 2011. Bird community patterns in sub-Mediterranean pastures: the effects of shrub cover and grazing intensity. *Animal Biodiversity and Conservation*, 34(1): 11–21.

NOAA (United States National Oceanic and Atmospheric Administration. Chesapeake Bay Office). 2018. *Oyster reefs* [online] [Cited 22 June 2018]. https://chesapeakebay.noaa.gov/oysters/oyster-reefs

Nowak, D.J., Hirabayashi, S., Bodine, A. & Greenfield, E. 2014. Tree and forest effects on air quality and human health in the United States. *Environmental Pollution*, 193: 119–129.

OECD/FAO. 2016. *OECD-FAO Agricultural Outlook 2016–2025.* Paris, OECD Publishing. (available at https://www.oecd-ilibrary.org/agriculture-and-food/oecd-fao-agricultural-outlook-2016_agr_outlook-2016-en).

Ollerton, J., Winfree, R. & Tarrant, S. 2011. How many flowering plants are pollinated by animals? *Oikos*, 120(3): 321–326.

Orgiazzi, A. & Panagos, P. 2018. Soil biodiversity and soil erosio: it is time to get married. *Global Ecology and Biogeography*, 27(10): 1155–1167.

Orgiazzi, A., Bardgett, R.D., Barrios, E., Behan-Pelletier, V., Briones, M.J.I., Chotte, J.-L., De Deyn, G.B., *et al.*, eds. 2016. *Global soil biodiversity atlas.* Luxembourg, European Commission, Publications Office of the European Union.

Ostroumov, S.A. 2010. Biocontrol of water quality: multifunctional role of biota in water self-purification. *Russian Journal of General Chemistry*, 80(13): 2754–2761.

Paerl, H. 2017. The cyanobacterial nitrogen fixation paradox in natural waters. *F1000Res.* 6: 244.

Palmer, M.A., Covich, A.P., Lake, S., Biro, P., Brooks, J.J., Cole, J. & Van De Bund, W.J. 2000. Linkages between aquatic sediment biota and life above sediments as potential drivers of biodiversity and ecological processes. *BioScience*, 60: 820–832.

Parlesak, A., Geelhoed, D. & Robertson, A. 2014. Toward the prevention of childhood undernutrition: diet diversity strategies using locally produced food can overcome gaps in nutrient supply. *Food and Nutrition Bulletin*, 35(2): 191–199.

Pecora, G., Musto, M., D'adamo, C., Adduci, F., Freschi, P. & Cosentino, C. 2015. GIS system and livestock field survey as tools to manage the potential reducing of fuel load for fire prevention. *Journal of Agricultural Informatics*, 6(3): 57–66.

Pereira, A.L.C., Taques, T.C., Valim, J.O.S., Madureira, A.P. & Campos, W.G. 2015. The management of bee communities by intercropping with flowering basil (*Ocimum basilicum*) enhances pollination and yield of bell pepper (*Capsicum annuum*). *Journal of Insect Conservation*, 19(3): 479–486.

Pilling, D. & Hoffmann, I. 2011. *Climate change and animal genetic resources for food and agriculture: state of knowledge, risks and opportunities*. CGRFA Background Study Paper No. 53. Rome, FAO. (available at http://www.fao.org/3/mb386e/mb386e.pdf).

Pimentel, D. 2006. Soil erosion: A food and environmental threat. *Environment, Development and Sustainability*, 8(1): 119–137.

Pirot, J.Y., Meynell, P.J. & Elder, D. 2000. *Ecosystem management: lessons from around the world.* A guide for development and conservation practitioners. Gland, Switzerland and Cambridge, UK, IUCN.

Plassmann, K., Laurence, M., Jones, M. & Edwards-Jones, G. 2010. Effects of long-term grazing management on sand dune vegetation of high conservation interest. *Applied Vegetation Science,* 13(1): 100–112.

Powell, B., Thilsted, S.H., Ickowitz, A., Termote, C., Sunderland, T. & Herforth, A. 2015. Improving diets with wild and cultivated biodiversity from across the landscape. *Food Security*, 7(3): 535–554.

Pringle, R.M. 2017. How large herbivores subsidize aquatic food webs in African savannas. *Proceedings of the National Academy of Sciences*, 114(29): 7489–7491.

Pullin, R. & White, P. 2011. *Climate change and aquatic genetic resources for food and agriculture: state of knowledge, risks and opportunities.* Commission on Genetic Resource for Food and Agriculture. Background Study Paper No. 55. Rome, FAO. (available at http://www.fao.org/docrep/meeting/022/mb507e.pdf).

Rah, J.H., Akhter, N., Semba, R.D., Pee, S.D., Bloem, M.W., Campbell, A.A., Moench-Pfanner, R., Sun, K., Badham, J. & Kraemer, K. 2010. Low dietary diversity is a predictor of child stunting in rural Bangladesh. *European Journal of Clinical Nutrition*, 64(12): 1393–1398.

Ranalli, P., ed. 2007. *Improvement of crop plants for industrial end uses*. Dordrecht, Netherlands, Springer.

Rao, N.S., Ghermandi, A., Portela, R & Wang, X. 2015. Global values of coastal ecosystem services: A spatial economic analysis of shoreline protection values. *Ecosystem Serivces*, 11: 95–105.

RBG Kew (Royal Botanic Gardens Kew). 2016. *The State of the World's Plants 2016*. Kew, UK.

Redwan, E.-R.M. 2009. Animal-derived pharmaceutical proteins. *Journal of Immunoassay and Immunochemistry*, 30(3): 262–290.

Rhyne, A.L., Tlusty, M.F., Schofield, P.J., Kaufman, L., Morris, J.A. & Bruckner, A.W. 2012. Revealing the appetite of the marine aquarium fish trade: the volume and biodiversity of fish imported into the United States. *PLoS ONE*, 7(5).

Rice, M.A. 2001. Environmental impacts of shellfish aquaculture: filter feeding to control eutrophication. *In* M.F. Tlusty, D.A. Bengston, H.O. Halvorson, S.D. Oktay, J.B. Pearce & R.B. Rheault, eds. *Marine aquaculture and the marine environment*, pp. 76–86. Falmouth, USA, Cape Cod Press.

Robb, D.H.F., MacLeod, M. Hasan, M.R. & Soto, D. 2017. *Greenhouse gas emissions from aquaculture. A life cycle assessment of three Asian systems.* FAO Fisheries and Aquaculture Technical Paper 609. Rome, FAO. (available at http://www.fao.org/3/a-i7558e.pdf).

Rocca, M. & Messelink, G.J. 2017. Combining lacewings and parasitoids for biological control of foxglove aphids in sweet pepper. *Journal of Applied Entomology*, 141(5): 402–410.

Rocha, J., Peixe, L., Gomes, N.C.M. & Calado, R. 2011. Cnidarians as a source of new marine bioactive compounds - An overview of the last decade and future steps for bioprospecting. *Marine Drugs*, 9(10): 1860–1886.

Rogger, M., Agnoletti, M., Alaoui, A., Bathurst, J.C., Bodner, G., Borga, M., Chaplot, V., *et al.* 2017. Land use change impacts on floods at the catchment scale: challenges and opportunities for future research. *Water Resources Research*, 53(7): 5209–5219.

Roman, J. & McCarthy, J.J. 2010. The whale pump: Marine mammals enhance primary productivity in a coastal basin. PLoS *ONE*, 5(10).

Rose, T., Kremen, C., Thrupp, A., Gemmill-Herren, B., Graub, B. & Azzu, N. 2016. *Policy Analysis Paper: Mainstreaming of biodiversity and ecosystem services with a focus on pollination*. Rome, FAO. (available at http://www.fao.org/3/a-i4242e.pdf).

Rowland, D., Ickowitz, A., Powell, B., Nasi, R. &Sunderland, T. 2017. Forest foods and healthy diets: quantifying the contributions. *Environmental Conservation*, 44(2): 102–114.

Ruberson, J.R., Nemoto, H. & Hirose, Y. 1998. Pesticides and conservation of natural enemies in pest management. *In* P. Barbosa, ed. *Conservation biological control*, pp. 207–220. London, Academic Press.

Saenger, P., Gartside, D. & Funge-Smith, S. 2013. *A review of mangrove and seagrass ecosystems and their linkage to fisheries and fisheries management.* RAP Publication 2013/09. Bangkok, FAO Regional Office for Asia and the Pacific. (available at http://www.fao.org/docrep/018/i3355e/i3355e.pdf).

Salako, V.K., Fandohan, B., Kassa, B., Assogbadjo, A.E., Idohou, A.F.R., Gbedomon, R.C., Chakeredza, S., Dulloo, M.E. & Kakaï, R.G. 2014. Home gardens: An assessment of their biodiversity and potential contribution to conservation of threatened species and crop wild relatives in Benin. *Genetic Resources and Crop Evolution*, 61(2): 313–330.

Sargent, J.R. 1997. Fish oils and human diet. *British Journal of Nutrition*, 78(01): S5–S13.

Scharlemann, J.P.W., Tanner, E.V.J., Hiederer, R. & Kapos, V. 2014. Global soil carbon: Understanding and managing the largest terrestrial carbon pool. *Carbon Management*, 5(1): 81–91.

Schippmann, U., Cunningham, A.B. & Leaman, D.J. 2002. Impact of cultivation and gathering of medicinal plants on biodiversity: global trends and issues. In *Biodiversity and the Ecosystem Approach in Agriculture, Forestry and Fisheries, satellite event on the occasion of the Ninth Regular Session of the Commission on Genetic Resources for Food and Agriculture, Rome 12–13 October 2002. Proceedings*. Rome, FAO. (available at http://www.fao.org/3/y4586e/y4586e00.htm).

Schulp, C.J.E., Thuiller, W. & Verburg, P.H. 2014. Wild food in Europe: A synthesis of knowledge and data of terrestrial wild food as an ecosystem service. *Ecological Economics*, 105: 292–305.

Schulz, S., Brankatschk, R., Dumig, A., Kogel-Knabner, I., Schloter, M. & Zeyer, J. 2013. The role of microorganisms at different stages of ecosystem development for soil formation. *Biogeosciences*, 10(6): 3983–3996.

Shackleton, C.M., Shackleton, S.E., Gambiza, J., Nel, E., Rowntree, K., Urquhart, P., Fabricus, C. & Ainslie, A. 2010. *Livelihoods and vulnerability in the arid and semi-arid lands of southern Africa: exploring the links between ecosystem services and poverty alleviation.* New York City, USA, Nova Publishers.

Sharps, E., Smart, J., Skov, M.W., Garbutt, A. & Hiddink, J.G. 2015. Light grazing of saltmarshes is a direct and indirect cause of nest failure in Common Redshank *Tringa totanus*. *Ibis*, 157(2): 239–249.

Sheffield, C.S., Ngo, H.T. & Azzu, N. 2016. *A manual on apple pollination*. Rome, FAO. (available at http://www.fao.org/3/a-i5527e.pdf).

Schieltz, J.M. & Rubenstein, D.I. 2016. Evidence based review: positive versus negative effects of livestock grazing on wildlife. What do we really know? *Environmental Research Letters*, 11: 113003.

Silliman, B.R., Mozdzer, T., Angelini, C., Brundage, J.E., Esselink, P., Bakker, J.P., Gedan, K.B., van de Koppel, J. & Baldwin, A.H. 2014. Livestock as a potential biological control agent for an invasive wetland plant. *PeerJ*, 2: e567.

Simopoulos, A.P. 1991. Omega-3 fatty acids in health and disease and in growth and development. *The American Journal of Clinical Nutrition*, 54(3): 438–463.

Sing, L., Metzger, M.J., Paterson, J.S. & Ray, D. 2017. A review of the effects of forest management intensity on ecosystem services for northern European temperate forests with a focus on the UK. *Forestry*, 91: 151–164,

Sipkema, D., Franssen, M.C.R., Osinga, R., Tramper, J. & Wijffels, R.H. 2005. Marine sponges as pharmacy. *Marine Biotechnology, 7*(3): 142–162.

Smale, D.A., Burrows, M.T., Moore, P., O'Connor, N. & Hawkins, S.J. 2013. Threats and knowledge gaps for ecosystem services provided by kelp forests: a northeast Atlantic perspective. *Ecology and Evolution*, 3(11): 4016–4038.

Smith, J. 2012. *Urban air quality.* Grantham, UK, The Woodland Trust.

Smith, P., Bustamante, M., Ahammad, H., Clark, H., Dong, H., Elsiddig, E.A., Haberl, H., *et al.* 2014. Agriculture, forestry and other land use (AFOLU). *In* O. Edenhofer, R. Pichs-Madruga, Y. Sokona, E. Farahani, S. Kadner, K. Seyboth, A. Adler, *et al.*, eds. *Climate change 2014: Mitigation of climate change.* Contribution of Working Group III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge, UK and New York, USA, Cambridge University Press.

Sokos, C.K., Mamolos, A.P., Kalburtji, K.L. & Birtsas, P.K. 2013. Farming and wildlife in Mediterranean agroecosystems. *Journal for Nature Conservation*, 21(2): 81–92.

Sorrenti, S. 2017. *Non-wood forest products in international statistical systems.* Non-wood Forest Products Series No. 22. Rome, FAO. (available at http://www.fao.org/3/a-i6731e.pdf).

Spracklen, D.V., Arnold, S.R. & Taylor, C.M. 2012. Observations of increased tropical rainfall preceded by air passage over forests. *Nature*, 489(7415): 282–285.

Sradnick, A., Murugan, R., Oltmanns, M., Raupp, J. & Joergensen, R.G. 2013. Changes in functional diversity of the soil microbial community in a heterogeneous sandy soil after long-term fertilization with cattle manure and mineral fertilizer. *Applied Soil Ecology*, 63: 23–28.

Stadlmayr, B., Charrondiere, U.R., Eisenwagen, S., Jamnadass, R. & Kehlenbeck, K. 2013. Nutrient composition of selected indigenous fruits from sub-Saharan Africa. *Journal of the Science of Food and Agriculture*, 93(11): 2627–2636.

Steinbeiss, S., Bessler, H., Engels, C., Temperton, V.M., Buchmann, N., Roscher, C., Kreutziger, Y., Baade, J., Habekost, M. & Gleixner, G. 2008. Plant diversity positively affects short-term soil carbon storage in experimental grasslands. *Global Change Biology*, 14(12): 2937–2949.

Steinfeld, H., Gerber, P., Wassenaar, T., Castel, V., Rosales, M. & de Haan, C. 2006. *Livestock's long shadow*. Rome, FAO. (available at http://www.fao.org/3/a0701e/a0701e00.htm).

Subalusky, A.L., Dutton, C.L., Rosi-Marshall, E.J. & Post, D.M. 2015. The hippopotamus conveyor belt: Vectors of carbon and nutrients from terrestrial grasslands to aquatic systems in sub-Saharan Africa. *Freshwater Biology,* 60(3): 512–525.

Subalusky, A.L., Dutton, C.L., Rosi, E.J. & Post, D.M. 2017. Annual mass drownings of the Serengeti wildebeest migration influence nutrient cycling and storage in the Mara River. *Proceedings of the National Academy of Sciences*, 114(29): 7647–7652.

Sugiyama, S., Staples, D. & Funge-Smith, S.J. 2004. *Status and potential of fisheries and aquaculture in Asia and the Pacific.* RAP Publication 2004/25. Bangkok, FAO Regional Office for Asia and the Pacific. (available at http://www.fao.org/3/a-ad514e.pdf).

Sunderland, T.C.H. 2011. Food security: why is biodiversity important? *International Forestry Review*, 13(3): 265–274.

Tamosiunas, A., Grazuleviciene, R., Luksiene, D., Dedele, A., Reklaitiene, R., Baceviciene, M., Vencloviene, J., *et al.* 2014. Accessibility and use of urban green spaces, and cardiovascular health: Findings from a Kaunas cohort study. *Environmental Health: A Global Access Science Source,* 13(1): 1–11.

Taylor, S., Gilbert, P.J. Cooke, D.A., Deary, M.E. & Jeffries, M.J. 2019. High carbon burial rates by small ponds in the landscape. *Frontiers in Ecology and the Environment,* 7(1): 25–31.

Teasdale, J.R. 2003. Principles and practices of using cover crops in weed management systems. *In* R. Labrada, ed. W*eed management in developing countries*, pp. 169–178. Rome, FAO. (available at http://www.fao.org/3/y5031e/y5031e00.htm).

TEEB (The Economics of Ecosystems and Biodiversity). 2010. *The economics of ecosystems and biodiversity: ecological and economic foundations.* P. Kumar, ed. London, Earthscan.

Teo, S.S. 2001. Evaluation of different duck varieties for the control of the golden apple snail (*Pomacea canaliculata*) in transplanted and direct seeded rice. *Crop Protection,* 20(7): 599–604.

Thomas, J. 1980. Why did the large blue become extinct in Britain? *Oryx*, 15(3): 243–247.

Toledo, A. & Burlingame, B. 2006. Biodiversity and nutrition: a common path toward global food security and sustainable development. *Journal of Food Composition and Analysis*, 19(6–7): 477–483.

Troell, M. 2009. Integrated marine and brackishwater aquaculture in tropical regions: research, implementation and prospects. *In* D. Soto, ed. *Integrated mariculture: a global review*, pp. 47–131. FAO Fisheries and Aquaculture Technical Paper No. 529. Rome, FAO. (available at http://www.fao.org/3/i1092e/i1092e00.htm).

Turbé, A., De Toni, A., Benito, P., Lavelle, P., Lavelle, P., Ruiz, N., Van der Putten, W.H., Labouze, E. & Mudgal, S. 2010. *Soil biodiversity: functions, threats and tools for policy makers.* Technical Report - 2010 - 049. European Commission DG ENV. Luxembourg, European Commission, Publications Office of the European Union.

Umberger, S.H. 2009. *Sheep grazing management.* Blacksburg, USA, Virginia Polytechnic Institute and State University.

UNEP-WCMC. 2014. T*he importance of mangroves to people: a call to action*. J. van Bochove, E. Sullivan & T. Nakamura, eds. Cambridge, UK

van Dam, J.E.G. 2008. Environmental benefits of natural fibre production and use. *Proceedings of the Symposium on Natural Fibres* (available at http://www.fao.org/docrep/pdf/011/i0709e/i0709e03.pdf).

van Tuyl, J.M., Arens, P., Miller, W.. & Anderson, N.O. 2014. The role of ornamentals in human life. *In* G.R. Dixon & D.E. Aldous, eds. *Horticulture: plants for people and places. Volume 1: Production horticulture,* pp. 407–433. Dordecht, Netherlands, Springer.

Varah, A., Jones, H., Smith, J. & Potts, S.G. 2013. Enhanced biodiversity and pollination in UK agroforestry systems. *Journal of the Science of Food and Agriculture*, 93(9): 2073–2075.

Vignesh, N.S., Maheswari, M. & Doraisamy, P. 2014. Biodiversity assessment under cocoa (Theobroma cacao L.) plantations of Tamil Nadu. *Asian Journal of Environmental Science*, 9(1): 1–5.

Vinceti, B., Ickowitz, A., Powell, B., Kehlenbeck, K., Termote, C., Cogill, B. & Hunter, D. 2013. The contributions of forest foods to sustainable diets. *Unasylva*, 64(241): 54–64.

Wabnitz, C., Taylor, M., Green, E. & Razak, T. 2003. *From ocean to aquarium*. Cambridge, UK, UNEP-WCMC.

Wagg, C., Bender, S.F., Widmer, F. & van der Heijden, M.G.A. 2014. Soil biodiversity and soil community composition determine ecosystem multifunctionality. *Proceedings of the National Academy of Sciences,* 111(14): 5266–5270.

Webb, E.L. & Kabir, M.E. 2009. Home gardening for tropical biodiversity conservation. *Conservation Biology*, 23(6): 1641–1644.

Welch, R.M. 2002. The impact of mineral nutrients in food crops on global human health. *Plant and Soil*, 247: 83–90.

Wong, J.L.G. & Wiersum, F.K. 2019. A spotlight on NWFPs in Europe. *In* B. Wolfslehner, I. Prokofieva & R. Mavsar, eds. *Non-wood forest products in Europe: seeing the forest around the trees. What science can tell us* 10, pp. 11–30. Joensuu, Finland, European Forest Institute.

Woodland Trust. 2012. *Woodwise: Conservation grazing in woodland management.* Grantham, UK, The Woodland Trust.

Wright, J.S., Fu, R., Worden, J.R., Chakraborty, S., Clinton, N.E., Risi, C., Sun, Y. & Yin, L. 2017. Rainforest-initiated wet season onset over the southern Amazon. *Proceedings of the National Academy of Sciences,* 114(32): 8481–8486.

Wunder, S., Angelsen, A. & Belcher, B. 2014. Forests, livelihoods, and conservation: broadening the empirical base. *World Development,* 64(S1): S1–S11.

Zakkak, S., Kakalis, E., Radović, A., Halley, J.M. & Kati, V. 2014. The impact of forest encroachment after agricultural land abandonment on passerine bird communities: the case of Greece. *Journal for Nature Conservation*, 22(2): 157–165.

Zander, K.K., Signorello, G., De Salvo, M., Gandini, G. & Drucker, A.G. 2013. Assessing the total economic value of threatened livestock breeds in Italy: implications for conservation policy. *Ecological Economics,* 93: 219–229.

zu Ermgassen, P.S.E., Spalding, M.D., Grizzle, R.E. & Brumbaugh, R.D. 2013. Quantifying the loss of a marine ecosystem service: filtration by the eastern oyster in US estuaries. *Estuaries and Coasts,* 36(1): 36–43.

ISBN 978-92-5-133168-2

9 789251 331682
CB0649EN/1/08.20