RAINFOREST EQUIVALENCE–A NEW APPROACH FOR COMPARATIVE ASSESSMENT OF ECOSYSTEM SERVICES OF HABITATS

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(Received 15th Aug 2024; accepted 22nd Nov 2024)

Abstract. To establish a framework of global agreements aimed at fostering a sustainable global community, it becomes imperative to possess the capability to measure the ecosystem services rendered by individual nations (or regions) to the entire planet in a manner that allows for meaningful comparisons. An effective valuation system should be capable of verification and recalculation by individuals at any time. It should rely solely on data that are consistently accessible in public databases, traceable, applicable across various spatial and temporal dimensions, and cumulative in nature. The lowland tropical rainforest is often regarded as the epitome of an ideal state of existence among the various biomes found on our planet. This particular biome stands out because of its exceptional levels of biomass, productivity, and biodiversity. The presence of liquid water diminishes considerably as individuals travel towards the northern or southern poles from the equator, and as they ascend higher altitudes in mountainous areas. Similarly, as individuals approach the Earth's poles, the amount of solar radiation suitable for utilization decreases, resulting in a decrease in annual heat input and a subsequent decrease in temperature. It is deemed suitable to quantify the worth of various communities in different habitats by using rainforest equivalents, taking into account their biomass, productivity, and the biodiversity they sustain. This approach to valuation enables quick and effective comparisons at a regional level, thus serving a crucial function in measuring adherence to multilateral international agreements, informing political and economic decisions, and evaluating changes in land use through monitoring.

Keywords: *biomes, biodiversity, climate, biomass, carbon*

Introduction and objective

To develop a system of international conventions for a sustainable global society, it is crucial to quantify the ecosystem services (*Table 1*). provided by individual countries or regions This quantification should enable meaningful comparisons at a global level. The valuation system must be verifiable and recalculable by anyone, at any time. It should be based only on data that are continuously available in public databases. The data must be traceable, applicable at different spatial and temporal scales, and cumulative.

The ecological conditions necessary for human survival include a climate regulated by biological processes, appropriate chemical composition of the atmosphere, and natural waters and soils. These conditions also involve biological materials used by humanity. Ecosystem services provide these conditions (Daily et al., 1997; Tallis and Kareiva, 2005; Costanza et al., 2017). The quantity and quality of these services depend on the condition, integrity, and health of the ecosystems that produce them (Carpenter et al., 2009). Different habitat types contribute differently to the quantity of services provided by the biosphere. This contribution per unit area (service value) is proportional to the total biomass of the biotic community. This can be expressed in terms of carbon stock as a

DOI: http://dx.doi.org/10.15666/aeer/2206_56175667

quantitative indicator. It also depends on the biological activity of the biotic community, which reflects annual productivity and can be expressed in terms of carbon sequestration. Additionally, it depends on the efficiency and reliability of the processes resulting from functional redundancy. This is an indicator of biodiversity and, on land, can be characterised by the number of vascular plant species.

Table 1. *Ecosystem services and functions Based on Newell and Roberts (2017)*

	Ecosystem service	Ecosystem function	Example
1.	Gas regulation	Regulating the chemical composition of the atmosphere	O_2/CO_2 balance, O_3 for UVB protection, SOx levels
2.	Climate regulation	Temperature, rainfall regulation, and other biologically influenced processes	Dimethyl sulfoxide affects cloud formation
3.	Disruption and disaster recovery	Maintaining ecosystem integrity, response to fluctuations	Storm protection, flood mitigation
4.	Water control	Regulation of hydrological processes	Agriculture, industry, and transport
5.	Water supply	Water storage and retention	River basins, reservoirs, and catchment areas
6.	Erosion reduction and sediment retention	Maintaining the soil that forms the basis of the ecosystem	Preventing wind and flood erosion
7.	Soil development and maintenance	Regulation of biological processes that lead to soil formation	Weathering of rocks, accumulation of organic matter
8.	Regulation of nutrients	Regulation of C, N, and other cycles, processing, storage, and accumulation of nutrients	Nitrogen fixation, supply of nutrients to agriculture
9.	Contamination treatment	Removal or breakdown of toxic xenobiotic substances	Detoxification
10.	Pollination	Transport of pollen	Production of agricultural crops
11.	Biological control	Regulating populations	Pest control by predators
12.	Refugium	Habitat for local and migratory populations	Providing a protected site for species of economic importance
13.	Food production	A percentage of the ecosystem's production can be used as food	Fish, game, cereals and fruit
14.	Raw materials	A percentage of the ecosystem's production can be extracted as raw material	Timber, fuel and feed
15.	Genetic pool	Source of biological production	Genes to build resistance
16.	Recreation	Opportunity for recreational activities	Ecotourism, fishing
17.	Cultural	Providing opportunities for cultural activities, spiritual and scientific values	Art, aesthetics, education

Among the biomes of our planet, the lowland tropical rainforest is considered an "ideal state of existence" in terms of biomass, productivity, and biodiversity (Alexandrov and Matsunaga, 2008; Keith et al., 2009; Mics et al., 2013; Eiserhardt et al., 2017). In comparison, all other habitats are 'struggle zones' with much lower values (Hufnagel et al., 2018). This distinctive feature of tropical rainforests is due to high, stable temperatures, which are favourable for biological processes. It is also due to high solar radiation, high humidity, and good availability of liquid water (Park, 2003). Moving north or south of the equator, or increasing altitude in mountains, reduces the availability of liquid water significantly. Moving towards the poles reduces the amount of usable solar radiation, decreases annual heat input, and lowers temperatures (Woodward et al., 2004).

It seems appropriate to express the value of communities in different habitats in terms of rainforest equivalents. This should be done in proportion to their biomass, productivity, and the biodiversity they support. This method of valuation allows for rapid and efficient regional comparisons. It can play an important role in quantifying commitments to international agreements. It can also assist in preparing political and economic decisions and in assessing land-use changes.

The aim of this paper is:

1. To develop a calculation procedure for a valuation system based on rainforest equivalency. This includes collecting the necessary baseline data and developing a global valuation of ecosystem services by biome and continent.
2. To conduct a global comparative valuation case study at the continental level. This serves as an example application and provides a conceptual basis for potential international agreements.

Review of literature

Ecosytem services valuation methodologies

Professionals are increasingly facing the challenge of estimating the value of specific areas, ecosystems, or species. Decisions are needed on how to manage an area and what should happen to its wildlife. For example, should a forest remain in its natural state or be converted for forestry? To answer this, the value of the forest must be assessed.

Two approaches to valuing natural ecosystems are found in the literature: anthropocentric and biocentric. The anthropocentric approach values nature based on its usefulness to humanity. The biocentric approach, however, considers everything in nature to have intrinsic value, regardless of its utility to humans (Daily, 1997). Proponents of the anthropocentric approach argue that since humans are the dominant species, they have the right to determine what is valuable for them (Daily, 1997).

Another approach views nature as having both direct (use value) and indirect (non-use value) value (de Groot et al., 2010). According to the Millennium Ecosystem Assessment, there are four categories of benefits from nature: provisioning (e.g., fisheries, timber), regulating (e.g., climate and river flow regulation), supporting (e.g., pollination, pest control), and cultural (e.g. tranquillity, inspiration) (Chan et al., 2006).

Since the 1960s, there has been increasing attention in the literature on assessing the value of ecosystems (Hein et al., 2006). Since the term “ecosystem services” was first mentioned in 1983, the number of related articles and citations has risen steeply (Costanza and Kubiszewski, 2012). Ecosystems provide a wide range of goods and services essential for human well-being (Nelson et al., 2009).

To protect ecosystems and their services, policymakers need to ensure that human activities are sustainable and that resources are allocated equitably and efficiently (Costanza and Folke, 1997). It must also be recognised that the value of nature and its contribution to quality of life cannot be separated from cultural and institutional contexts (Pascual et al., 2017). Political decisions and public opinion greatly influence the perceived value of services, making their valuation controversial (Loomis et al., 2000).

Some believe it is not possible or wise to value certain aspects of nature, as intangibles like aesthetics or long-term ecological benefits are hard to quantify (Costanza, 2000). As a result, there can be significant contradictions between economic and ecological valuations (Farber et al., 2002). Particularly in Western countries, it is important to value natural ecosystems, where high productivity is prioritised in economic decisions (Gómez-Baggethun et al., 2009).

Expressing ecosystem services in monetary terms does not mean they can be treated as marketable products or private property (Costanza et al., 2014). For example, pollination and water control benefit everyone but cannot be owned (Wilson and Howarth, 2002). Policy decisions should consider this, though translating ecosystem service valuation into financial mechanisms is still unresolved (Daily et al., 2009). Ecosystem services are hard to compare to factory goods, which results in them being given little weight in policy decisions (Chee, 2004). However, economic valuation is crucial for monitoring services (Kumar and Kumar, 2008).

The attitude towards valuing services is illustrated by the water and diamond paradox. Water, essential to life, is valued little, while diamonds, which are not essential, have a high monetary value (Heal, 1999). Humanity benefits from natural ecosystems but also alters them. It is vital to monitor changes in ecosystem conditions, as their degradation affects human quality of life (Howarth and Farber, 2000). Ecological processes are under threat due to human activities. Habitat destruction, conversion, and pollution lead to the disappearance of natural ecosystems globally (Barbier, 2007). Despite environmental regulations, agricultural, industrial, and residential developments continue to degrade natural vegetation (National Research Council, 2005). These threats are expected to grow as demand for energy and raw materials increases (de Groot et al., 2012).

Most people today live disconnected from nature, and conservation is often seen as an obstacle to development. However, ecosystem services can shift this perspective, making conservation a driver of development (Gómez-Baggethun and Pérez, 2011). Valuing ecosystem services helps decision-makers choose between management options to achieve multiple objectives (Liu et al., 2010). This system links ecology with economics, so economic methods should be used to value ecosystem components (Chan et al., 2012). Many methods exist to determine the monetary value of services, though missing data make this challenging (Sherrouse et al., 2011).

Direct market valuation methods

Revealed preference methods

Market price method

In some cases, the value of a service can be measured directly from the market price of its products, which can be sold in the market. In these cases, the value is based on the transaction price. This means no complex methods are needed. Examples include timber, firewood, fish, and other foodstuffs. The value of these commodities also reflects the value of the ecosystem service.

The advantage of this method is its simplicity. It uses available information on prices, quantities, and costs, and requires only simple assumptions. However, many services are not directly tradable, which can lead to false or distorted information that does not reflect the true value of the service. Additionally, it is difficult to use for large-scale changes that affect the supply and demand of the service (Koetse et al., 2015).

Production function method

This approach is used when a good or service is partly the result of human labour and partly due to ecosystem contributions. For example, many agricultural crops depend on insect pollination, and the value of pollination can be estimated from the value and quality of these crops. It is a method developed to estimate indirect use values.

The difficulty lies in determining the relationship between ecosystem services and human contributions, making this method less commonly used. However, it has been applied to measure water quality or changes in water quality based on reduced treatment costs, improved crop production due to better pollination, or improved soil quality. In other words, the quality of a marketable commodity improves due to an ecosystem service.

A problem with this method is that researchers must consider both human and mechanical inputs, which can lead to overestimating the value of the ecosystem service. Despite this, it has the advantage of being theoretically suitable for valuing ecosystem services, as it assumes a close relationship between the service and its economic benefit (Pascual et al., 2009).

Cost-based methods

This method measures the value of an ecosystem service by estimating the damage that would occur if the service were not provided. It also considers the costs of replacing the service. It is commonly used to assess water quality, water purification costs, and protection against soil erosion, storms, and other natural disasters. It also measures the protection of natural habitats. In this case, the service is a non-marketable product and reflects the cost of producing the benefit, rather than the benefit itself. This method aligns with how the economy values and creates value. However, a disadvantage is that the cost of repairing the damage may not always reflect the actual benefits received (Daly, 2016).

Random utility and travel-cost methods

The travel-cost method and the random utility method are based on the assumption that people know their preferences, although researchers do not always know these preferences. Certain aspects of preferences can be determined using statistical methods. These methods are mostly used to evaluate recreational fishing in lakes, rivers, and the sea. They assess the value of a non-marketable ecosystem service by measuring how much money and time people spend to reach a fishing or swimming site. Time, money, and the number of visits represent the value of the site, fishing, and swimming (National Research Council, 2005).

Hedonic pricing method

This method measures the indirect value of an ecosystem service that cannot be sold but can be estimated through the value of a related good. To determine the value, two products are needed that are identical except for some environmental factors, such as traffic noise or distance from a park. The difference in their monetary value reflects people’s willingness to pay for an ecosystem service. This method is often used to estimate the benefits or costs related to the quality of the natural environment, such as air pollution, water pollution, or noise. For example, the value of houses can reflect the quality of the environment. A house in a better environment is worth more, while a similar house in an area with higher air pollution may be worth less. This analysis helps determine whether a change in environmental factors affects the value of a marketable good (Bouma and van Beukering, 2015).

Stated preference methods

Contingent valuation

This method uses questionnaires to assess the value of ecosystem services. The responses show how much people are willing to pay for certain services. In other words, it tries to understand how people would behave in specific situations. Since these services are not sold on the market, the questions ask how much the respondent would pay in a given situation. Options may include a new tax, an entrance fee to a national park, or an annual, monthly, or one-off maintenance fee. This method is widely used for assessing the value of community property. However, respondents often struggle to determine how much they would pay for the service. Many value the ecosystem highly but cannot assign it a monetary value, and responses also depend on their income level (Carson and Bergstrom, 2003).

Conjoint analysis

This is another popular method based on questionnaires. Respondents answer questions about the characteristics of a product or service. For example, they may choose between two options describing possible features of a park, such as distance from their home, size, vegetation, and accessibility. Statistical analysis then reveals the relative importance of these characteristics to the respondents. It also shows how far people are willing to travel to visit the park. Responses can be compared to those for other recreational opportunities (Bergkamp and Goldsmith, 2013).

Non-monetary valuation

There are cases where monetary valuation methods cannot be used or are not suitable. This may be due to the nature of the ecosystem service, uncertainty about changes in natural factors, or the intentions of the participants. In such cases, other options are available (Hadley et al., 2011).

Interviews and focus groups

Led by a moderator, 7-10 people who do not know each other discuss a topic in a structured manner. Although they are strangers, they share common characteristics or interests. During the discussion, participants express their opinions, and the moderator observes trends and patterns in their thinking. They are not pressured to vote, plan, or reach a consensus. In-depth interviews are similar to focus groups but aim to provide a detailed understanding of the research subject. In these interviews, experts are also involved (Erdoğdu et al., 2016).

Citizen's jury

This method involves a test with 12-25 people, guided by an independent moderator. The aim is to obtain an informed opinion from the community on a specific issue or when choosing between different decision options. Experts and stakeholders present their opinions and evidence. The jury considers this information carefully and forms an opinion, which is only a recommendation and not binding. Jury members are selected from all sections of society to represent the whole population and increase public participation in the democratic process (Wittmer and Gundimeda, 2012).

Health-based valuation approaches

This approach is based on the impact on health, considering both the quality and length of life of the people studied. It measures how a person's health has improved or deteriorated over time, considering factors in the natural environment (Ten Brink, 2011).

Q-methodology

This is a thorough and statistically reliable method for investigating people's subjective opinions. Participants complete a test called a Q-sort, where they rank various statements based on different criteria. This helps reveal the values and preferences that the community considers important. Participants associate negative, neutral, and positive indicators with different statements, and a ranking of importance is established. The results are then analysed using factor analysis (Armatas et al., 2014).

Delphi method

The aim of this method is to collect and process the comments and opinions of experts in a particular field. By continuously reviewing and refining opinions, a consensus is eventually reached in the investigation. The collective input of several experts helps solve the problem more effectively. Originally developed for military purposes in the 1950s during the Cold War, this method is now used successfully in many disciplines (Boberg and Morris-Khoo, 1992).

Summary of literature

Plants and plant ecosystems are essential in assessing land management practices, as they are significant natural and cultural assets (Wang et al., 2018). An important factor in this research is improving the objectivity and ease of conducting biological assessments. This can be done by developing formulas to quantify the environmental factors involved. Quantification is crucial, especially when working with planning authorities, who prefer numerical approaches.

Ecosystem valuation involves assigning a value–monetary, biophysical, or otherwise–to an ecosystem and its services. Quantifying vegetation's benefits, such as flood and erosion control, carbon storage, support for endangered species, and absorption of harmful chemicals, allows for monetizing these advantages. This approach helps policymakers and conservationists assess the impact of management strategies and conduct cost-benefit analyses to compare policies. Assigning monetary value makes evaluating management impacts more objective and aids decision-making. However, these valuations are only approximations, involving quantitative uncertainty and philosophical debate over non-market costs and benefits.

The current global challenge is to conserve natural resources, address climate change through mitigation and adaptation, and prevent the degradation of ecosystem services. Despite the growing importance of ecosystem services, their value remains poorly understood in economic markets and government policies (Hancock, 2010). The importance of biodiversity in supporting ecosystem services and the potential impacts of biodiversity decline are widely recognised (Feld et al., 2009). Preserving natural resources, addressing climate change, and preventing ecosystem degradation are the main challenges facing the global community today.

Estimating the monetary value of ecosystem services can be useful in various contexts. One such context is the use of decision-making tools like cost-benefit analysis and cost-effectiveness analysis, which rely heavily on monetary values. By incorporating these values, these tools provide a comprehensive evaluation of the significance of all aspects of ecosystem services, including both utilitarian and non-utilitarian aspects. This facilitates an objective assessment.

Despite many methodologies for classifying, quantifying, and valuing ecosystem services, choosing the most suitable one remains unclear and often depends on researchers' preferences. Valuing ecosystem services in different ecosystems, such as forests, grasslands, deserts, and wetlands, is challenging due to disputes over the methods used, the specific services valued, and the lack of comprehensive global coverage (Chan and Satterfield, 2020).

No monetary value is currently assigned to the rainforest equivalent, but it could be. The market value of available biomass (carbon stock) and biomass produced (carbon sequestration) is relatively easy to calculate. The value of biodiversity is more difficult to determine, but it is possible. Economic value can be defined as the monetary amount an individual is willing to pay to obtain an ecosystem service (willingness to pay, WTP). Alternatively, it can be the monetary compensation needed to persuade an individual to forgo the service (willingness to accept, WTA) (Loomis et al., 2014).

Market data consists of prices determined by supply and demand, which can include environmental goods. Restoration expenses or compensation payments are part of this category. The hedonic price approach and travel cost method can measure people's preferences for exploring natural landscapes. These techniques are indirect and use proxies to represent the value of the object being evaluated.

When market prices are not available, values can be obtained directly from individuals using stated-preference methods. The contingent valuation and choice modelling methods are used to determine prices for environmental goods based on their characteristics. Other methods include participatory models, which often start with social WTP, and system-dynamic approaches, which focus on the stability of the investigated ecosystem services (ESS).

In the above methods, the perspective of the people using the ecosystem service is relevant. However, it is also important to consider the characteristics of the vegetation, which are independent of users' subjective views. The chosen attributes (carbon stock, carbon sequestration, and biodiversity) influence the ecosystem services provided. Therefore, it is crucial to assess vegetation's intrinsic attributes. These two approaches–user perspectives and vegetation attributes–are complementary, not mutually exclusive. People's assessments can be subjective, so it is essential to also consider vegetation characteristics. Both approaches are important to accurately determine the environmental value of an area.

The rainforest equivalent is a proposed assessment based on vegetation characteristics. It considers both the basic values and their multiplication and sum. This allows for a combined estimation of the value of three types of ecosystem services and introduces a new perspective in valuing natural resources.

Materials and methods

Basic data and their sources

The extent of biomes on each continent is characterised by potential (pre-human activity) and current (turn of the millennium) values. For anthropogenic land use types, different period-specific values are provided, with data in *Table 2* and sources in *Table 3*.

The first number in each cell of *Table 2* shows the theoretically possible extent of vegetation in a biome without human activity. The second number shows the current extent by the turn of the millennium, reduced due to human activity.

For North America, Oceania, South America, and Asia, rainforest data are combined, as the original author treated these areas together for simplicity. Anthropogenic biomes have no potential value; their extent increases with human population growth (Ellis and Ramankutty, 2008) and the demand for new territories, which accelerated during the industrial revolution. For anthropogenic biomes of artificial origin, the first number is smaller, and the second is larger. The first value is from the 1700s and provides only approximate information.

Differences in measurements by various authors can arise from differences in measurement procedures. Sometimes, certain formations are omitted. For example, some articles mention only savanna, while others distinguish between woody and grassy savannas. A biome may also lack a clear boundary, as with natural steppes and human-created pastures, where grazing can alter and degrade vegetation.

We tried to identify areas with untouched vegetation and used data from those areas. However, this is not possible everywhere. In Europe, for example, temperate forests have long been managed for forestry, which may alter their characteristics compared to pristine forests. Foresters often plant trees in areas with suboptimal conditions or introduce non-native species (Gilliam, 2016). Many grassy areas today are used for livestock grazing, which also changes vegetation characteristics.

Antarctica (14 million km^2) is excluded from this study because it is covered in ice and has minimal vegetation. Much of Greenland (2,166,086 km^2) is also covered in ice, but there are areas with tundra vegetation. Since Greenland is administratively part of Denmark, we include it in Europe. The total extent of potential vegetation is estimated at $1.36 * 10^8$ km^2. According to the United Nations, the ice-free land surface (excluding Antarctica and Greenland) is $1.34 * 10^8$ km^2. The data we used are approximations and may be slightly affected by uncertainties, but this does not significantly impact the applicability of the procedure.

Table 4 shows the carbon stock, carbon sequestration per unit area, and the number of vascular plant species per unit area. These values were used to characterise the vegetation, except for mangroves, which have very few species adapted to this specialised environment. *Table 5* summarises the sources of the data in *Table 4*.

For the number of vascular plant species per unit area (columns 3 and the last column), the data cannot simply be multiplied by the area. Instead, we used the Arrhenius equation to estimate species richness. This equation is a common tool for estimating species richness in ecology and conservation. According to Arrhenius (1921) and Kier et al. (2005), it follows the power model of the species–area relationship:

$$S_e = S_u * (\frac{A_e}{A_u})^z \quad \text{(Eq.1)}$$

S_e = biome estimated number of species,
S_u = number of species per unit area,
A_e = area of biome,
A_u = unit area (1 km^2),
z = parameter defining the slope.

The z values determining the slope of the curve and their literature sources are given in *Table 6*.

Table 2. *Potential and actual area values (the first figure is potential, the second is actual)*

	Africa km^2	**South America km^2**	**North America km^2**	**Asia km^2**	**Europe km^2**	**Pacific km^2**	**Total km^2**
Lowland rainforest	$4.0177*10^6$ – $8.7000*10^5$	$7.0721*10^6$ – $4.5400*10^6$	0	$3.4907*10^6$ – $1.7700*10^6$	0	0	$1.4581*10^7$ – $7.8000*10^6$
Montane rainforest	$5.4466*10^5$ – $1.5860*10^5$	$1.1506*10^6$ – $5.7340*10^5$	0	$1.5620*10^6$ – $4.8800*10^5$	0	0	$3.2573*10^6$ – $1.2200*10^6$
Cloud forest	$5.7190*10^4$ – $3.4328*10^4$	$9.6394*10^4$ – $8.7626*10^4$	0	$2.2758*10^5$ – $9.2676*10^4$	0	0	$3.8116*10^5$ – $2.1463*10^5$
Tropical seasonal forest	$3.6695*10^6$ – $3.4328*10^4$	$1.6816*10^6$ – $5.6840*10^5$	$2.2600*10^5$ – $1.4682*10^5$	$1.4266*10^6$ – $1.4682*10^5$	0	$4.6800*10^5$ – $3.9850*10^4$	$7.4717*10^6$ – $9.3622*10^5$
Mangrove	$5.3719*10^4$ – $3.6529*10^4$	$6.9614*10^4$ – $4.3161*10^4$	0	$1.2058*10^5$ – $7.7169*10^4$	0	$1.1961*10^4$ – $1.0287*10^4$	$2.5673*10^5$ – $1.6688*10^5$
Temperate deciduous forest	$1.0000*10^5$ – $1.0000*10^4$	$7.0000*10^5$ – $4.0000*10^5$	$4.2600*10^6$ – $2.1300*10^6$	$4.4200*10^6$ – $8.7000*10^5$	$4.4200*10^6$ – $1.1100*10^6$	$6.8000*10^5$ – $7.0000*10^4$	$1.4600*10^7$ – $4.5900*10^6$
Temperate conifer forest	0	0	$1.9717*10^6$ – $6.4619*10^5$	$1.4403*10^6$ – $2.8439*10^5$	$3.13*10^5$ – $9.4451*10^4$	0	$3.7250*10^6$ – $1.0250*10^6$
Boreal forest	0	0	$4.8247*10^6$ – $3.7923*10^6$	$7.1202*10^6$ – $2.2188*10^6$	$2.4992*10^6$ – $2.3829*10^5$	0	$1.4444*10^7$ – $6.2494*10^6$
Savannas	$1.40*10^7$ – $3.07*10^6$	$3.99*10^6$ – $1.37*10^6$	$7.69*10^4$ – $9.96*10^3$	$6.13*10^4$ – $2.05*10^4$	0	$2.14*10^6$ – $1.31*10^6$	$2.0268*10^7$ – $5.7805*10^6$
Deserts	$9.77*10^6$ – $7.30*10^6$	$1.14*10^6$ – $2.93*10^5$	$2.34*10^6$ – $4.88*10^5$	$1.11*10^7$ – $4.26*10^6$	0	$3.58*10^6$ – $2.60*10^6$	$2.7930*10^7$ – $1.4941*10^7$
Temperate grasslands	$3.60*10^5$ – $2.40*10^5$	$1.63*10^6$ – $3.46*10^5$	$3.10*10^6$ – $1.18*10^5$	$4.03*10^6$ – $2.83*10^6$	$7.00*10^5$ – $3.78*10^5$	$5.76*10^5$ – $1.73*10^3$	$1.0396*10^7$ – $3.9137*10^6$
Montane grassland	$3.10*10^4$ – $1.72*10^4$	$8.71*10^5$ – $2.24*10^5$	$3.19*10^5$ – $8.61*10^4$	$3.40*10^6$ – $8.55*10^5$	$1.95*10^5$ – $1.76*10^5$	$5.41*10^4$ – $1.62*10^4$	$4.8701*10^6$ – $1.3745*10^6$
Tundra	0	0	$3.94*10^6$ – $3.78*10^6$	$3.30*10^6$ – $3.08*10^6$	$1.06*10^6$ – $8.53*10^5$	0	$8.3000*10^6$ – $7.7130*10^6$
Mediterranean vegetation	$9.4695*10^5$ – $1.2509*10^5$	$1.4841*10^5$ – $1.1333*10^5$	$1.7643*10^5$ – $1.3649*10^5$	$5.6000*10^5$ – $1.2680*10^4$	$8.8900*10^5$ – $6.4800*10^5$	$8.0282*10^5$ – $5.1195*10^5$	$3.5236*10^6$ – $1.5475*10^6$
Total	$3.3551*10^7$ – $1.1896*10^7$	$1.8550*10^7$ – $8.5589*10^6$	$2.1235*10^7$ – $1.1334*10^7$	$4.2259*10^7$ – $1.7006*10^7$	$1.0076*10^7$ – $3.4977*10^6$	$8.3129*10^6$ – $4.5600*10^6$	$1.3398*10^8$ – $5.6853*10^7$

	Africa km^2	South America km^2	North America km^2	Asia km^2	Europe km^2	Pacific km^2	Total km^2
	1700-2000						
Human areas	$1.4140*10^3$ – $4.1679*10^4$	$7.0490*10^3$ – $1.4854*10^5$	$2.5700*10^2$ – $1.8185*10^5$	$4.3900*10^2$ – $3.8400*10^4$	$4.4390*10^3$ – $1.1406*10^5$	$4.9000*10^1$ – $1.3871*10^4$	$1.3647*10^4$ – $5.3840*10^5$
	1700-2000						
Cropland	$5.60*10^3$ – $1.43*10^6$	$1.29*10^4$ – $7.54*10^5$	$5.60*10^3$ – $1.97*10^6$	$1.03*10^5$ – $3.98*10^6$	$1.02*10^5$ – $2.35*10^6$	$2.10*10^3$ – $4.28*10^5$	$2.6560*10^5$ – $1.0902*10^7$
	1980-2000						
Pasture	$9.1111*10^6$ – $8.6988*10^6$	$1.0161*10^7$ – $1.1061*10^7$	$3.4300*10^6$ – $3.4860*10^6$	$4.2400*10^6$ – $4.5810*10^6$	$8.5578*10^5$ – $1.8234*10^6$	$4.5347*10^6$ – $4.1946*10^6$	$3.2333*10^7$ – 3.3844*107
	1990-2010						
Agroforest	$1.1663*10^5$ – $1.5409*10^5$	$7.8840*10^5$ – $1.2688*10^6$	$2.0481*10^5$ – $3.8660*10^5$	$8.2760*10^4$ – $1.3821*10^5$	$5.9046*10^5$ – $6.9318*10^5$		$1.7831*10^6$ – $2.6408*10^6$

Table 3. *Sources of territorial values (FRA- Forest Resource Assessment)*

	Africa	South America	North America	Asia	Europe	Pacific
Lowland rainforest	FRA, 2000	FRA, 2000	See South America	FRA, 2000	0	See Asia
Montane rainforest	Kapos et al., 2000; Iremonger et al., 1997; Bruijnzeel et al., 2011; Spracklen and Righelato, 2014	Kapos et al., 2000; Iremonger et al., 1997; Bruijnzeel et al., 2011; Spracklen and Righelato, 2014	See South America	Kapos et al., 2000; Iremonger et al., 1997; Bruijnzeel et al., 2011; Spracklen and Righelato, 2014	0	See Asia
Cloud forest	Bubb et al., 2004; Bruijnzeel et al., 2011; Spracklen and Righelato, 2014	Bubb et al., 2004; Bruijnzeel et al., 2011; Spracklen and Righelato, 2014	See South America	Bubb et al., 2004; Bruijnzeel et al., 2011; Spracklen and Righelato, 2014	0	See Asia
Tropical seasonal forest	FRA, 2000; Miles et al., 2006	FRA, 2000; Miles et al., 2006	FRA, 2000; Miles et al., 2006	FRA, 2000; Miles et al., 2006	0	FRA, 2000; Miles et al., 2006
Mangrove	Valiela, 2006	Valiela, 2006	See at South America	Valiela, 2006	0	Valiela, 2006

DOI: http://dx.doi.org/10.15666/aeer/2206_56175667

	Africa	**South America**	**North America**	**Asia**	**Europe**	**Pacific**
Temperate deciduous forest	Silander, 2001	Silander, 2001	Silander, 2001	Silander, 2001	Silander, 2001	Silander, 2001
Temperate conifer forest		Dinerstein et al., 2019	Dinerstein et al., 2019	Dinerstein et al., 2019		
Boreal forest	Burton et al., 2003	Burton et al., 2003	Burton et al., 2003	Burton et al., 2003	Burton et al., 2003	Burton et al., 2003
Savannas	Olson et al., 2001; Williams et al., 2020	Olson et al., 2001; Williams et al., 2020	Olson et al., 2001; Williams et al., 2020	Olson et al., 2001; Williams et al., 2020	0	Olson et al., 2001; Williams et al., 2020
Deserts	Olson et al., 2001; Williams et al., 2020	Olson et al., 2001-; Williams et al., 2020	Olson et al., 2001; Williams et al., 2020	Olson et al., 2001; Williams et al., 2020	0	Olson et al., 2001; Williams et al., 2020
Temperate grasslands	Carbutt et al., 2011	. Olson et al., 2001-; Williams et al., 2020	Olson et al., 2001; Williams et al., 2020	Gibson, 2009	Gibson, 2009; European Environmental Agency, 2002	Mark and McLennan, 2005
Montane grassland	Peart, 2008; Olson and Dineerstein,, 2002	Dinerstein et al., 2019	Diaz and Eischeid, 2007; Testolin et al., 2020	Olson and Dineerstein, 2002; Dinerstein et al., 2019	European Environmental Agency, 2002	Dinerstein et al., 2019; Olson and Dineerstein, 2002
Tundra	0	0	Olson et al., 2001; Williams et al., 2020	Olson et al., 2001; Williams et al., 2020	Olson et al., 2001; Williams et al., 2020	0
Mediterranean vegetation	Cox and Underwood, 2011; European Environmental Agency, 2002 Zahran and Gilbert, 2010; Efe et al., 2008; Blondel and Aronson, 1999; Blondel, 2010	Cox and Underwood, 2011; Pinborg, 2002; Zahran and Gilbert, 2010; Efe et al., 2008; Blondel and Aronson, 1999; Blondel, 2010	Cox and Underwood, 2011; Pinborg, 2002; Zahran and Gilbert, 2010; Efe et al., 2008; Blondel and Aronson, 1999; Blondel, 2010	Cox and Underwood, 2011; European Environmental Agency, 2002; Zahran and Gilbert, 2010; Efe et al., 2008; Blondel and Aronson, 1999; Blondel, 2010	Cox and Underwood, 2011; Pinborg, 2002; Zahran and Gilbert, 2010; Efe et al., 2008; Blondel and Aronson, 1999; Blondel, 2010	Cox and Underwood, 2011; European Environmental Agency, 2002; Zahran and Gilbert, 2010; Efe et al., 2008; Blondel and Aronson, 1999; Blondel, 2010

	Africa	**South America**	**North America**	**Asia**	**Europe**	**Pacific**
Human areas	Goldewijk et al., 2010a	Goldewijk et al., 2010a	Goldewijk et al., 2010a	Goldewijk et al., 2010a	Goldewijk et al., 2010a	Goldewijk et al., 2010a
Cropland	Goldewijk, 2001; Potapov et al., 2022	Goldewijk, 2001; Potapov et al., 2022	Goldewijk, 2001; Potapov et al., 2022	Goldewijk, 2001; Potapov et al., 2022	Goldewijk, 2001; Potapov et al., 2022	Goldewijk, 2001; Potapov et al., 2022
Rangeland; Pastureland	FAO; Goldewijk et al., 2010b	FAO; Goldewijk et al., 2010b	FAO; Goldewijk et al., 2010b	FAO; Goldewijk et al., 2010b	FAO; Goldewijk et al., 2010b	FAO; Goldewijk et al., 2010b
Planation	Kroeger, 2012	Kroeger, 2012	Kroeger, 2012	Kroeger, 2012	Kroeger, 2012	Kroeger, 2012

Table 4. *Carbon stock, carbon sequestration and vascular plant species number per unit area*

	Carbon stock C t/ha	**Carbon sequestration C t/ha/year**	**Species number/10,000 km^2**
Lowland rainforest	210	13-17	2750
Montane rainforest	130	7	4500
Cloud forest	69	4	5000
Tropical seasonal forest	82	7.6	1862
Mangrove	79	8	70 species worldwide
Temperate deciduous broadleaf forest	109	4.6	1087
Temperate conifer forest	62.1	5.7	1111
Boreal forest	114.5	4.1	545
Savannas	23.15	6.7	1041
Deserts	3.5	1.15-2.69	457
Temperate grasslands	4	3.75	756
Montane grassland	3.35	1.96	927
Tundra	3.5	1.94	227
Mediterranean vegetation	67	5	1220
Human areas	5	2.2	1684
Cropland	1.875	1.1-5.6	1038
Pastureland	5	4.3	980
Agroforest	162.5	7.2	1671

***Table 5.** Sources of carbon stock, carbon sequestration and species number values per unit area*

	Carbon stock C t/ha	**Carbon sequestration C t/ha/year**	**Species number**
Lowland rainforest	http://www.esd.ornl.gov	Girardin et al., 2010	Barthlott et al., 2007
Montane rainforest	http://www.esd.ornl.gov	Girardin et al., 2010	Barthlott et al., 2007
Cloud forest	Román-Cuesta et al., 2011	Girardin et al., 2010	Barthlott et al., 2007
Tropical seasonal forest	Becknell et al., 2012	Ito and Oikawa, 2004	Ellis et al., 2012
Mangrove	Yee, 2010	Yee, 2010	Polidoro et al., 2010
Temperate deciduous forest	Lal and Lorentz, 2012	Ito and Oikawa, 2004	Ellis et al., 2012
Temperate conifer forest	Thurner et al., 2014	Luyssaert et al., 2007	Ellis et al., 2012
Boreal forest	Thurner et al., 2014	Ito and Oikawa, 2004	Ellis et al., 2012
Savannas	Bouvet et al., 2018	Ito and Oikawa, 2004	Ellis et al., 2012
Deserts	Houghton et al., 2009	Ito and Oikawa, 2004	Ellis et al., 2012
Temperate grasslands	Houghton et al., 2009	Saugier et al., 2001	Ellis et al., 2012
Montane grassland	Oliveras et al., 2013	Sun et al., 2021	Ellis et al., 2012
Tundra	Houghton et al., 2009	Ito and Oikawa, 2004	Ellis et al., 2012
Mediterranean vegetation	Pan et al., 2013	Flexas et al., 2012	Ellis et al., 2012
Human areas	Melillo. et al., 1990	Haberl et al., 2007	Ellis et al., 2012
Cropland	Geist, 2006	Ito and Oikawa, 2004	Ellis et al., 2012
Pastureland	Gibbs et al., 2014	Haberl et al., 2007	Ellis et al., 2012
Agroforest	Nabuurs et al., 2003	Haberl et al., 2007	Ellis et al., 2012

***Table 6.** The z values for the Arrhenius equation*

Vegetation	**z Value**	**Source**
Rainforests	0.2875	Kier et al., 2005
Tropical dry forest	0.21	Kier et al., 2005
Temperate deciduous forest	0.17	Kier et al., 2005
Temperate conifer forest	0.14	Kier et al., 2005
Boreal forest	0.16	Kier et al., 2005
Savanna	0.18	Kier et al., 2005
Temperate grassland	0.12	Kier et al., 2005
Alpine	0.17	Kier et al., 2005
Tundra	0.13	Kier et al., 2005
Mediterranean	0.2	Kier et al., 2005
Deserts	0.11	Kier et al., 2005
Mangrove	-	
Cropland	0.269	Gerstner et al., 2014
Pasture	0.177	Gerstner et al., 2014
Plantation	0.13	Proenca et al., 2010
Urban	0.078	Ceschin et al., 2012

Calculation and evaluation methods

We calculated indices (ERR - Equivalent Rainforest Rate) for tropical lowland rainforests by dividing the carbon stock (as a measure of biomass), carbon sequestration (as a measure of production), and the number of vascular plant species per unit area (as an approximation of biodiversity) based on the values from *Table 4*. These indices express the proportion of equivalent rainforest area for each variable. If multiple literature values were available for a characteristic, we used their averages. For mangroves, we used the global species richness value instead of the per unit area value. Although this is an overestimate, its low value means it does not significantly affect the comparison.

These three characteristics are not entirely independent but provide complementary information. If they were completely independent, we would use the product of the three indices (Product ERR) to compare habitats. If they were closely related, we would consider them as different estimates of the same characteristic and use their average (Average ERR). The reality lies between these two extremes, so it is sensible to consider them together.

We multiplied the relative ratios for equivalent rainforest values by the area (in km²) of each habitat type being compared (biomes, continents). We then summed these values for each object to determine the size of the equivalent lowland rainforest (ERA - Equivalent Rainforest Area) for that object. This provides a suitable non-monetary characterisation of the natural capital (or expected ecosystem services) of the objects on a global scale. Assigning a monetary value would require pricing a unit area of rainforest, which is beyond the scope of this article.

Results

Values of ERR- Equivalent Rainforest Rate

Comparing the ERR values for biomes (*Table 7*) shows that both Average ERR and Product ERR have similar biome rankings. In both cases, the highest values are for lowland rainforest, followed by montane rainforest. Next are the cloud forest, tropical seasonal forest, and agroforest groups, though with slight differences in ranking. The lowest values are consistently found in the tundra, followed by deserts and montane grasslands, with other biomes in between.

Interestingly, the species richness of montane rainforests exceeds that of lowland rainforests in all other aspects. This can be explained by species from both lower and higher elevations thriving in these areas. These data may relate to the Intermediate Disturbance Hypothesis (IDH) in biodiversity (Wilkinson, 1999), but its analysis is beyond the scope of this article.

Worldwide aggregated Equivalent Rainforest Area (ERA) values for biomass and land use types

Table 8 shows the ERA data calculated from the Average and Product ERRs of the three attributes, by multiplying these ERRs by the area of the biomes globally. *Table 9* shows the same data aggregated by continent.

Application case study

To develop the use case study, additional data on human population, environmental pressure (carbon dioxide emissions), and economy (GDP) of the continents are needed. These data are presented in *Table 10*. A comparative assessment can be made using their percentage shares, as shown in *Table 11*.

Table 7. *ERR - Equivalent Rainforest Rate data for each habitat type (biome)*

Biome	ERR Carbon stock	ERR Carbon sequestration	ERR Species number	Average ERR	Product ERR
Lowland rainforest	1.00	1.00	1.00	1.0000	1.0000
Montane rainforest	0.62	0.47	1.64	0.9074	0.4727
Cloud (moss) forest	0.33	0.27	1.82	0.8045	0.1593
Tropical seasonal (dry) forest	0.74	0.51	0.68	0.6422	0.2548
Mangrove	0.38	0.53	0.03	0.3117	0.0051
Temperate broadleaf forest	0.69	0.31	0.40	0.4641	0.0837
Temperate conifer forest	0.29	0.34	0.40	0.34	0.0407
Boreal forest	0.43	0.27	0.20	0.3000	0.0232
Tropical savanna	0.09	0.45	0.38	0.3036	0.0145
Deserts	0.01	0.13	0.17	0.1028	0.0003
Temperate grassland	0.03	0.25	0.27	0.1854	0.0022
Montane grassland	0.02	0.09	0.34	0.1488	0.0005
Tundra	0.01	0.13	0.08	0.0746	0.0001
Mediterranean vegetation	0.32	0.33	0.44	0.3653	0.0472
Human areas (urban and villages)	0.02	0.15	0.61	0.2609	0.0021
Cropland	0.01	0.22	0.38	0.2032	0.0008
Pastureland	0.02	0.29	0.36	0.2223	0.0024
Agroforest	0.77	0.48	0.61	0.6205	0.2257

Table 8. *Biomass Equivalent Rainforest Area (ERA) values on our planet*

	ERA by Average ERR [km^2]		ERA by Product ERR [km^2]	
	Potential	**Actual**	**Potential**	**Actual**
Lowland rainforest	$1.46*10^7$	$7.18*10^6$	$1.46*10^7$	$7.18*10^6$
Montane rainforest	$2.96*10^6$	$1.11*10^6$	$1.54*10^6$	$5.77*10^5$
Cloud (moss) forest	$2.45*10^5$	$1.38*10^5$	$9.71*10^4$	$5.47*10^4$
Tropical seasonal (dry) forest	$1.59*10^5$	$1.04*10^5$	$5.77*10^4$	$3.77*10^4$
Mangrove	$6.01*10^6$	$1.46*10^6$	$1.19*10^6$	$2.90*10^5$
Temperate broadleaf forest	$7.51*10^6$	$4.75*10^6$	$1.35*10^6$	$8.57*10^5$
Temperate conifer forest	$1.29*10^6$	$3.56*10^5$	$1.52*10^5$	$4.18*10^4$
Boreal forest	$1.44*10^7$	$7.19*10^6$	$1.86*10^6$	$9.29*10^5$
Tropical savanna	$1.06*10^6$	$3.32*10^5$	$8.18*10^4$	$2.57*10^4$
Deserts	4.43*106	$1.37*10^6$	$2.11*10^5$	$6.55*10^4$
Temperate grassland	$2.96*10^6$	$1.11*10^6$	$4.85*10^4$	$1.82*10^4$
Montane grassland	$7.07*10^5$	$2.18*10^5$	$2.37*10^3$	$7.33*10^2$
Tundra	$7.51*10^5$	$7.51*10^5$	$8.75*10^3$	$8.75*10^3$
Mediterranean vegetation	$1.11*10^6$	$5.62*10^5$	$9.12*10^3$	$4.60*10^3$
Human areas (urban and villages)		$2.08*10^5$		$7.71*10^2$
Cropland		$3.21*10^6$		$1.08*10^4$
Pastureland		$3.48*10^6$		$1.03*10^4$
Agroforest		$1.97*10^5$		$3.36*10^2$
Total	**5.94*107**	**$3.45*10^7$**	**$2.11*10^7$**	**$1.01*10^7$**

Table 9. *Continents' Equivalent Rainforest Area (ERA) values as their contribution to global ecosystem services*

	ERA by Average ERR [km^2]		**ERA by Product ERR [km^2]**	
	Potential	**Actual**	**Potential**	**Actual**
Europe	2.50*10^6	1.81*10^6	9.55*10^4	3.78*10^4
Asia	1.65*10^7	9.22*10^6	5.30*10^6	2.43*10^6
Africa	1.95*10^7	8.29*10^6	6.67*10^6	1.83*10^6
North America	4.87*10^6	3.35*10^6	2.38*10^5	1.03*10^5
South America	1.28*10^7	8.12*10^6	8.35*10^6	5.22*10^6
Australia and Oceania	3.21*10^6	3.76*10^6	4.26*10^5	4.70*10^5
Antarctica				
Total	**5.94*10^7**	**3.45*10^7**	**2.11*10^7**	**1.01*10^7**

Table 10. *Data from continents on the characteristics of their human societies (as a measure of their use of and pressure on ecosystem services)*

	Population	**CO_2-emission (t, 2021)**	**GDP (nominal, billions of USD, 2021)**
Europe	7.48*10^8	5.31*10^9	2.35*10^4
Asia	4.68*10^9	2.17*10^{10}	3.68*10^4
Africa	1.37*10^9	1.45*10^9	2.69*10^3
North America	5.97*10^8	6.14*10^9	2.68*10^4
South America	4.34*10^8	1.07*10^9	3.25*10^3
Australia and Oceania	4.32*10^7	8.34*10^8	1.89*10^3
Antarctica	0	0	0
Total	7.87*10^9	3.65*10^{10}	5.49*10^4
Source	statisticstimes.com	https://ourworldindata.org	https://statisticstimes.com

Table 11. *The number and distribution of humanity by continent according to www.worldometer.info. Data download date 01.01.2023*

Continent	**Population 2020**	**Area (km²)**	**Density (P/km²)**	**World population share**
Europe	7.48*10^8	2.21*10^7	34	9.59%
Asia	4.64*10^9	3.10*10^7	150	59.54%
Africa	1.43*10^9	2.96*10^7	45	17.20%
North America	5.92*10^8	2.13*10^7	28	7.60%
South America	4.31*10^8	1.75*10^7	25	5.53%
Australia/Oceania	4.31*10^7	8.49*10^6	5	0.55%
Antarctica	0	1.37*10^7	0	0.00%

By examining the first four columns of *Table 12*, it is evident that Africa has the largest relative deficit between its potential and actual shares of providing ecosystem services. Africa, with its large equatorial regions, provides substantial ecosystem services but may have suffered the most damage, as its actual share is only one-third of its potential. In contrast, Australia and Oceania show the largest relative surplus, with an actual share

DOI: http://dx.doi.org/10.15666/aeer/2206_56175667

twice their potential, as they were among the last regions to be extensively exploited by human civilization. However, this is mainly of natural historical interest.

Table 12. *Comparing continents by their contribution to and percentage of use of global ecosystem services*

	Contribution to ecosystem services				**Use of ecosystem services**		
	Potential ERA by Avg EER [km^2]	**Potential ERA by Prod EER [km^2]**	**Actual ERA by Avg EER [km^2]**	**Actual ERA by Prod EER [km^2]**	**Population**	**CO_2 emission (t, 2017)**	**GDP (nominal, billions of USD, 2021)**
Europe	4.2%	0.5%	5.2%	0.4%	11.7%	17.7%	42.8%
Asia	27.8%	25.1%	26.7%	24.1%	50.8%	55.1%	67.1%
Africa	32.8%	31.6%	24.0%	18.1%	20.9%	3.8%	4.9%
North America	8.2%	1.1%	9.7%	1.0%	9.2%	18.8%	48.8%
South America	21.5%	39.6%	23.5%	51.7%	6.7%	3.2%	5.9%
Australia and Oceania	5.4%	2.0%	10.9%	4.7%	0.7%	1.4%	3.5%
Antarctica							
Total	100%	100%	100%	100%	100%	100%	100%

The key information comes from comparing the first four columns with the last three: Europe, Asia, and North America use global ecosystem services more than they contribute to them. In contrast, Africa, South America, Australia, and Oceania contribute significantly more than they use. Therefore, as long as this imbalance continues, it would be advisable for Europe, Asia, and North America, with larger GDPs and greater environmental impacts, to provide financial contributions. This would help Africa, South America, and Oceania continue providing ecosystem services through their conservation efforts, with lower per capita burdens. These comparisons can be broken down to countries, states, or smaller administrative units and could form the basis for detailed international agreements.

Discussion

In this article, we reviewed habitat types from the equator to the Arctic tundra. Due to human activities, natural vegetation-covered areas on Earth have decreased almost everywhere (Hoekstra et al., 2005). In contrast, human-transformed areas, such as agricultural lands, urban areas, and infrastructure, are increasing (Steffen et al., 2007). With the rapid growth of the human population, the land needed for food production and raw materials has also increased (Bongaarts, 2009).

The tropical rainforest is one of the largest, most productive habitats with high biodiversity, which is why we chose it as a reference for comparison. The highest obtained ratios were for the rainforest, so we used it as the unit. Montane rainforests had even higher species richness per unit area. Their high biodiversity in relatively small areas makes them especially valuable for conservation (Gradstein et al., 2008; Bendix et al., 2010). This high biodiversity can be explained by the island effect (Triantis et al., 2006) and possibly by their transitional nature or the Intermediate Disturbance Hypothesis (IDH).

Generally, the calculated ERR value decreases as we move away from the equator. Temperate and boreal forests have a relatively high carbon stock per unit area but do not reach rainforest levels. Significant habitat loss has occurred in temperate forests and grasslands, leaving little intact vegetation due to dense agricultural activity and human population in these regions. The least affected habitats are tundra, montane grasslands, and deserts, which have low carbon stock and primary production due to harsh climates. These biomes also have limited agricultural significance. Agricultural areas sequester less carbon compared to intact vegetation, and their ERR values are low, approaching desert levels. Plantation forests have higher values but still do not reach the maximum, as trees are harvested before achieving the carbon stock typical of old-growth forests (Thornley and Cannell, 2000).

Natural habitat areas have not only decreased in extent but also suffer from degradation and fragmentation, reducing their ability to reach their maximum potential (Sloan et al., 2014). As a result, the ability of terrestrial vegetation to sequester atmospheric carbon dioxide and produce biomass has declined. This leads to more carbon dioxide in the atmosphere, increasing the greenhouse effect and global temperature. Vegetation plays an essential role as an ecosystem service provider, and without it, carbon dioxide accumulates in the atmosphere. Human-transformed vegetation has a low ERR value and cannot fulfil this role. Comprehensive protection of habitats is needed to preserve the largest portion of all three levels of biodiversity: genetic, taxonomic, and ecological (Olson and Dinerstein, 1998).

Asia and Africa represent the largest share of the average rainforest equivalent. These continents have the most vegetation in terms of rainforest equivalent, providing valuable ecosystem services despite human activities. For the product of rainforest equivalent, South America stands out with the highest rainforest equivalent area, indicating that its remaining natural vegetation provides the most and highest-quality ecosystem services.

Asia has the largest population and highest carbon dioxide emissions among all continents. Many developing countries in Asia have a small ecological footprint per capita. High population density is mostly in the southern part of the continent, while central and northern regions have fewer people and lower emissions. Siberia has a large area of natural vegetation, contributing significantly to Asia’s ERA.

Africa has the lowest GDP among continents and ranks in the middle for carbon dioxide emissions. It has the second-largest population after Asia, but income levels are low, and the natural environment is less transformed. In terms of the product of ERA, South America has the most rainforest, which forms the basis of the new measurement method. South America has many valuable areas due to its large remaining natural vegetation, which is crucial for providing ecosystem services that support human life and quality of life.

Protecting the natural environment, optimising resource use, and improving environmental security are vital for the sustainability of developing countries. However, developing countries face ongoing environmental degradation through agriculture, water consumption, deforestation, pollution, and poor-quality food products (Glantz, 1999; Tamazian and Rao, 2010; Adebayo et al., 2021). The relationship between environmental degradation and economic growth is a key area in ecological economics (Keho, 2017). Sustainable economic development can only be achieved alongside sustainable environmental development (Tiwari et al., 2013).

The link between economic growth and CO_2 emissions has been extensively studied in recent decades. Many countries face the challenge of maintaining economic growth while protecting the environment. Economic growth often leads to higher energy use and CO_2 emissions, contributing to climate change. On the other hand, growth can also lead to the adoption of energy-efficient and low-carbon technologies. Understanding the relationship between environmental quality and economic growth helps policymakers balance these factors. Economic growth impacts the environment, and the main goal of any economy is to maximise growth (Živanović et al., 2016).

The sources of economic growth and differences in economic performance across regions are significant and challenging aspects of modern development, making them a major focus of scientific study (Greiner et al., 2005). In industrialising countries with low per capita income, emissions of certain pollutants may increase with economic growth. However, once per capita income reaches a certain level, further growth can lead to more environmental damage. Improvements may come from lower population growth, a shift to less polluting goods, increased imports of pollution-intensive goods, environmental policies, government agencies focused on environmental control, and the development of emission-reducing technologies (Komen et al., 1997; Mushafiq and Prusak, 2023).

Developed countries usually prioritise environmental protection, but differences in protection levels exist even among countries with similar economic performance. Individual factors can also influence environmental policy decisions. Recent social science research shows a growing interest in understanding what affects public support for environmental projects. It suggests that socio-demographic factors, such as age, education, gender, race, ideology, party affiliation, and urbanisation, as well as economic variables like employment and income, shape public opinion (Marsiliani and Renström, 2000).

It is important to note some limitations (and areas for further development) in our work. Our research did not include Antarctica or the world's oceans. These areas are nominally linked to states and alliances, but they have no substantial population or independent economy, making them subjects of international agreements rather than active participants.

A more significant criticism may be that we limited the assessment of biodiversity to vascular plants for data reliability and simplicity. Therefore, we did not consider mosses, lichens, microorganisms, or animal biomass, production, or biodiversity. A discussion of these issues and relevant supplementary data can be found in the appendix of our article.

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APPENDIX

Limitations, ecological aspects not considered, additional data, and suggestions for further development of the assessment system.

Content of appendices

A.1. The comparative assessment of continents omitted Antarctica (Additional information and references).

A.2. The role of mosses and lichens was not taken into account in the vegetation assessment (Additional information and references).

A.3. Animal biomass, production, and biodiversity not taken into account.

A.4. We have not investigated the role of microorganisms.

A.5. We have not taken into account the seas and oceans, and to a limited extent the mangrove habitat.

A.1. The comparative assessment of continents omitted Antarctica

The ice-free area of Antarctica represents only 0.5% of the continent, approximately 332,000 km² (Claridge et al., 2000). This area allows for the development of flora. Unlike other continents, vascular plants do not dominate here, making one of our indicators unsuitable. Due to the cold climate, mosses and lichens are the primary vegetation, with only two vascular plant species present (Deschampsia antarctica, Colobanthus quitensis) (Convey, 2001; Øvstedal and Lewis Smith, 2001). These species are mainly found on the peninsula, South Shetland Islands, and South Orkney Islands, where climate conditions are milder. Their biomass and production vary greatly depending on microclimatic, soil, and topographic conditions (Holdgate, 1964; Holtom and Greene, 1967; Edwards, 1972; Vera et al., 2013).

Microflora, including blue-green algae, eukaryotic algae, and fungi, are also significant. The fauna mainly consists of invertebrates. The continent can be divided into three biogeographical zones: surrounding islands, coastal areas, and the continental interior (Smith, 1984; Longton, 1988; Chown and Convey, 2007). Climatic conditions vary slightly, with the surrounding islands having the mildest climate, resulting in richer flora and fauna. As the climate becomes more extreme, species richness and biomass decrease. Overall, diversity is lower than in similar latitudes of northern tundras (Rønning, 1996; Nielsen and Wall, 2013). *Table A.1.1* shows the species richness of these three regions.

The average phytomass on continental land ranges from 5-200 g/m², while in oceanic areas it is between 300-1000 g/m² for living, photosynthesizing tissues. When considering the accumulation of dead tissues, it can reach up to 20-30 kg/m². Peat formation can reach 1 mm per year, and accumulating peat can be used to study climate change over thousands of years (Longton, 1992; Royles and Griffiths, 2015).

Table A.1.1. *Antarctic biodiversity based on Convey (2006)*

OTU	Sub-Antarctic Islands	Maritime Antarctica	Continental Antarctica
Protozoa	83		33
Rotifera	59	50	13
Tardigrada	34	26	19
Nematoda	22	28	14
Platyhelminthes	4	2	0
Gastrotricha	5	2	0
Annelida	23	3	0
Mollusca	4	0	0
Crustacea	48	10	14
Insecta	210	35	49
Collembola	30	10	10
Arachnida	167	36	29
Myriopoda	3	0	0
Flowering plants/Angiosperms	60	2	0
Ferns/Pteridophyta	16	0	0
Bryophyta	250	100	25
Marchantiophyta	85	25	1
Lichenophyta	250	250	150
Fungi	70	30	0
Total	**1423**	**609**	**357**

For mosses, Net Primary Productivity (NPP) ranges from 5-100 g/m²/year in continental climates, and from 300-650 g/m²/year in more favourable oceanic climates (Kennedy, 1995). Lichen biomass in continental areas ranges from 46-177 g/m² (Friedmann, 1982), with an NPP of only 5 mg/m²/year (Vestal, 1988). Near Birthday Ridge (70°8' 48°0' S), combined moss and lichen biomass ranges from 50-950 g/m² (Allan Green et al., 2007). In oceanic climates, biomass values are 800-1750 g/m², with an NPP of 250 g/m²/year (Smith, 1984). Near the Spanish base (Juan Carlos I), Beltrán-Sanz et al. (2022) found an average of 92.2 mg CO_2 gDW^{-1} per year from 2009-2014. At McMurdo Dry Valleys, Geyer et al. (2017) measured 217 g C/m²/year at a site with 60 days available for growth. Growth rates for lichens vary significantly, from 0.47–0.50 mm/year on Signy and Livingston Islands to less than 0.01 mm/year in the Dry Valleys (Sancho et al., 2019).

In the more northern islands, several vascular plants occur. On South Georgia Island, vegetation includes dry grasslands (Festuca contracta), moist grasslands (Deschampsia antarctica), oligotrophic peatlands (Rostkovia magellani), eutrophic peatlands (Tortula robusta and Juncus scheuchzerioides), cushion plants (Acaena magellanica), and tussock grasses (Poa flabellata) (Holdgate, 1977). In Festuca contracta grasslands, above-ground biomass averages 425 g/m², and NPP can reach 340 g/m²/year. In Poa flabellata grasslands, values reach 7525 g/m² and 5020 g/m²/year. On Signy Island, Deschampsia antarctica stands reach 327 g/m² for biomass and 390 g/m²/year for NPP (Holdgate, 1977).

On Marion Island, Smith (2008) measured biomass between 173-1245 g/m², with the lower value for mire vegetation and the higher for tussock grassland. NPP ranged from 266-949 g/m²/year, with the lower value for fellfield and the higher for tussock grassland. Climate change may lead to an expansion of ice-free areas, allowing invasive species to threaten existing ecosystems (Frenot et al., 2005).

Due to the climate, biodiversity is much lower. Convey (2006) summarised the species found on the continent, showing that there are more species on the milder northern islands. There are only 1423 species on the subantarctic islands, including both animals and plants. This is because the area is much smaller compared to the Arctic tundras. On the continent, the limited extent of ice-free areas restricts the biodiversity of flora and fauna.

Mosses, lichens, and areas with vascular plants provide habitat for invertebrates. On Signy Island, invertebrate biomass can reach 8.5 g/m², and on South Georgia Island, it can reach 17 g/m², with production of 10 g/m²/year (Heal and MacLean, 1975). Bacterial biomass in soil ranges from 0.04-0.63 µg/g on South Georgia Island to 2-12 µg/g on Signy Island (Rosswall and Heal, 1975). Díaz-Puente et al. (2021) measured soil microbial carbon on Deception Island, King George Island, and Cierva Point, finding values of 33.55 mg/kg, 75.69 mg/kg, and 134.96 mg/kg, respectively. Areas covered with moss and lichen had higher microbial biomass and carbon content.

The metabolic quotient [q(CO2)] was 7.16, 3.91, and 12.64, highest in vegetated areas. For comparison, in tropical rainforests in the Amazon, the microbial carbon content was 1287 µg/g in the upper 5 cm of soil, which decreased after deforestation (Luizao et al., 1992). In Venezuela, López-Hernández (2017) found 98-218 mg/g of microbial carbon in primary forest soils. In desert soil, 6.3 mmol C per kg was measured, equivalent to 0.0756 g/kg of soil (Xu et al., 2013).

Studying microbial diversity is challenging because many microorganisms in extreme environments are not cultivable in labs (Merino et al., 2019). Molecular technologies like pyrosequencing are helpful in such studies. In McMurdo Valley, Thompson et al. (2020) found 71 families and 90 genera of phagotrophic protists, and 32 genera and 23 families of ciliates using molecular techniques. Archer et al. (2017) found 501 Operational Taxonomic Units (OTUs) in sandstone and granite using the 16S rRNA method. Sandstone showed higher diversity due to its porous structure, which favours microbial communities. The orientation of the rock also affects which groups dominate, as it influences radiation exposure.

In the Mitchell Peninsula, Ji et al. (2016) collected 93 soil samples and identified 6928 OTUs across 40 phyla, with higher diversity due to better climate and nutrient content. In contrast, Lopatina et al. (2013) found fewer OTUs in snow samples near the Russian bases Druzhnaya and Leningradskaya. Surprisingly, a Pseudomonas species grew at 37°C, indicating it is not a psychrophile. It likely arrived in Antarctica in an inactive form.

Kim et al. (2007) compared Antarctic soils to tropical soils and found 396 OTUs in terra preta soil in the Jamari National Forest, Amazon, using the oligonucleotide fingerprint grouping (OFRG) method.

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A.2. The roles of mosses and lichens was not taken into account in the vegetation assessment

Vascular plants have the highest biomass, but non-vascular plants also contribute significantly to biomass and carbon storage. They provide essential ecosystem services and play a crucial role in mitigating climate change (Blok et al., 2011). These groups include Bryophyta, Marchantiophyta, Anthocerotophyta, and Lichenophyta. They occur in most terrestrial ecosystems and less often in aquatic environments. The more extreme the habitat, the more important their role and contribution to vegetation biomass. In extreme areas, higher-order plants cannot survive due to harsh conditions.

Lichens can photosynthesize and assimilate carbon, playing a role similar to plants. They are symbiotic organisms composed of algae and fungi. Lichens are highly tolerant of changing environmental conditions. The fungal partner is 98% Ascomycota and 2% Basidiomycota. In 10% of lichens, cyanobacteria are the partner, and in 90%, eukaryotic algae are the partner (Honegger, 2009). Lichens can survive in extremely low temperatures in Antarctica and tolerate desiccation for long periods.

Lichens lack specialized organs and absorb water and nutrients over their entire surface, but they also lose water through the surface. This limits their growth conditions. Without protection, they dry out quickly, and their physiological activity stops, leading to slow growth. Lichens also absorb toxic substances from the environment if unprotected. Due to their stress tolerance, they can survive on rock surfaces with few competitors. Their sensitivity to pollution makes them good ecological indicators.

Mosses have leaf- and root-like structures but lack vascular tissues. They do have cells for transport, though these are not like the tracheae of higher plants. Like lichens, mosses are highly resistant to desiccation and are poikilohydric. They have a haploid-diploid life cycle with a dominant gametophyte phase. Mosses are the closest living relatives of the first land plants. They bridge modern vascular plants and their algal ancestors, representing a transition to land plants (Kenrick and Crane, 1997).

In dry and cold areas, carbon uptake is about 16 g/m^2/year, equivalent to 16 t/km^2/year. Using the value of 170 t/km^2/year from Girardin et al. (2010), this results in a rainforest equivalent of 0.0941. In areas covered with herbaceous plants and grasses, non-vascular plants have a carbon uptake of 23 g/m^2/year, which equals 0.1352 rainforest equivalent. In areas with no soil and only rock surfaces, carbon uptake is 8 g/m^2/year (Elbert et al., 2009), corresponding to 0.0471 rainforest equivalent. The epiphytic layer provides 28 g/m^2/year of carbon uptake, resulting in a rainforest equivalent of 0.1647.

Biomass ranges from 1 to 1200 g in dry weight. The median value is 260 g/m^2 for soil and 130 g/m^2 for epiphytes. Converted to per square kilometre, these values are 260 and 130 tons, respectively. A conversion factor of 2 was used based on the methodology by Petersson et al. (2012). Therefore, carbon content ranges from 130 to 75 t/km^2. The rainforest value of 2100 t/ha corresponds to rainforest equivalents of 0.0619 and 0.0357.

In tropical rainforests, moss biomass on the forest floor varies from 10-12 g/m^2 in lowlands and increases with altitude. In mountainous areas, it can reach 200-500 kg/ha, and in cloud forests, it can reach up to 10,300 kg/ha. In cloud forests, mosses can retain up to 30,000 L of water per hectare when it rains, playing an important role in water management and nutrient supply (Pócs, 1982; Frahm et al., 2003). The lower biomass in lowland areas may be due to high temperatures causing respiration that cannot be balanced by photosynthesis under low light. In cloud forests, lower temperatures favour mosses and lichens (Richards, 1984; Wagner et al., 2013).

There are about 13,000 species of mosses (Goffinet et al., 2008) and 13,500 species of lichens (Hawksworth et al., 1995). The Phylum Bryophyta consists of approximately 13,000 moss species, divided into eight classes. These include pleurocarpous mosses (4–7 classes, or 42% of species) and acrocarpous mosses (23 orders), which are not linked to a single evolutionary group. Ninety percent of existing moss species belong to two subclasses: Dicranidae (acrocarpous haplolepidae) and Bryidae (diplolepideous-alternate mosses with both acrocarpous and pleurocarpous members) (Newton et al., 2009).

Schmit and Mueller (2007) estimate that there are at least 20,000 species of lichens, which is more than the previous estimate by Hawksworth et al. (1995). Most lichen species are found in temperate zones rather than tropical regions. An exception is Ecuador, which has a very high species count. The Andes Mountain range runs through Ecuador, and despite being near the equator, the climate is relatively moderate, which may explain the high diversity.

Many known species come from the Alps, the British Isles, and the Scandinavian Peninsula, but there may still be many undescribed species in tropical countries (Feuerer and Hawksworth, 2007). In Austria, Köckinger et al. (2011) report 1138 moss species, while Switzerland has 1100 species (Hofmann and Schnyder, 2010). The Alps, covering an area of 170,000 km², are one of the most thoroughly studied regions. Nimis et al. (2018) list 3009 species, excluding some uncertain findings. The Alps have 4450 vascular plant species, equivalent to 2200 species per 10,000 km² (Aeschimann et al., 2011).

Using the Arrhenius equation, the estimated number of rainforest species for 170,000 km² is 2977, resulting in a rainforest equivalent of 1.04. The z-value is set at 0.15, as recommended by Geffert et al. (2013). *Figure A.2.1* shows the geographic distribution of species richness.

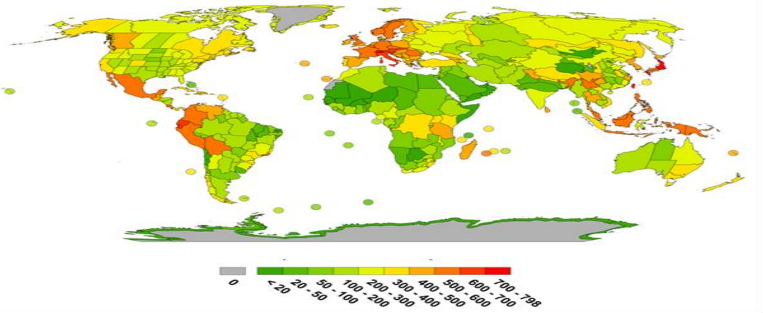

Figure A.2.1. *Number of moss species per 10,000 km^2 based on Geffert et al. (2013)*

In general, species richness increases closer to the equator. This pattern is often seen across different scales, habitats, and taxa (Mittelbach et al., 2007; Cox et al., 2016). However, there are exceptions. Taxa that evolved during warmer periods have a steeper gradient, while those from colder periods show a less pronounced gradient due to their lower affinity for tropical climates (Romdal et al., 2013). They have lower temperature optima than higher-order plants (Glime, 2007). Additionally, there are fewer habitats with poikilohydric conditions in tropical climates (Mateo et al., 2016). Lichens do not show a linear gradient in species richness. Instead, they show a U-shaped pattern in the relationship between species richness and latitude (Holt et al., 2015).

Mosses and lichens have traits that suit cold climates. Many species have a net assimilation rate at 10-15°C, and even below freezing, assimilation and respiration still occur (Longton, 1992).

Due to global climate change, rising temperatures and decreased precipitation may reduce moss-covered areas and their biomass. Mosses will also spend more time in an inactive state, without absorbing carbon dioxide (Li et al., 2021).

DOI: http://dx.doi.org/10.15666/aeer/2206_56175667

An important ecosystem service provided by mosses and lichens is soil formation and protection against degradation. Due to human activities, their role in this service may decrease, which will negatively affect other organisms (Allen and Lendemer, 2016).

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A.3. Animal biomass, production and biodiversity were not taken into account

Heterotrophic organisms occupy the next trophic levels, and their biomass and production depend on how much primary production they consume and transfer to the next level. Whittaker and Likens (1972) estimated consumption rates as follows: 1% for cultivated lands, 2-3% for deserts and tundras, 4-7% for forests, and 10-15% for steppes and savannas. In aquatic ecosystems, estimates are 40% in the open ocean, 40% in upwelling zones, 30% on continental shelves, and 15% in kelp forests and river mouths. On land, the average is around 7%, while in the ocean, it is 37%. These values are estimates based on limited data and species studied, and we lack complete data for all species. *Table A.3.1* shows animal biomass values for different habitat types.

In rainforests, the carbon content of plant biomass is 210 tons per hectare, while average animal biomass carbon is 9 g/m², which equals 0.09 tons per hectare. The ratio of 0.09/210 equals 0.00043, representing the rainforest equivalent. Production is 0.064 tons of carbon per square kilometre. Using a value of 17 t/ha for rainforests, the ratio of 0.064/17 equals 0.0038, representing the rainforest equivalent. This shows that only a small amount of carbon reaches heterotrophic animals at higher trophic levels.

The increase in species richness towards the equator, known as the latitudinal gradient, has long been recognised in ecology (Pianka, 1966; Mittelbach et al., 2007). The tropics have higher species richness, particularly for vertebrates. In contrast, the Nearctic and Palearctic regions have temperate or cold climates with fewer plant species, lower biomass, and lower production. This is also reflected in animal diversity, as adapting to cold climates is stressful, and plant food sources are less diverse. *Table A.3.2* shows vertebrate species richness in different zoogeographical realms, reflecting climate conditions and vegetation diversity, which forms the basis of the food chain.

Table A.3.1. *Average biomass and production of animals by ecosystem based on Whittaker and Likens (1972)*

Ecosystem	Animal biomass t C/ha	production t C/ha
Tropical rainforest	0.09	0.064705882
Tropical seasonal forest	0.054	0.04
Temperate evergreen forest	0.045	0.024
Temperate deciduous forest	0.07	0.027142857
Boreal forest	0.022	0.014166667
Woodland and shrubland	0.022	0.01375
Savanna	0.068	0.07
Temperate grassland	0.031	0.033333333
Tundra and alpine	0.002	0.001875
Desert	0.002	0.0015
Rock, ice and sand	0.000004	4.16667*10-6
Cultivated land	0.002	0.002857143
Swamp and marsh	0.045	0.09
Lake and stream	0.022	0.048
Total terrestrial	0.031	0.024966443
Open ocean	0.011	0.034337349
Upwelling zone	0.045	0.125
Continental shelf	0.027	0.073308271
Algal bed and reef	0.09	0.183333333
Estuaries	0.068	0.178571429
Total marine	0.0124	0.038116343

Table A.3.2. *Species richness of terrestrial vertebrate species by biogeographic realm based on Millennium Ecosystem Assessment (2005)*

Biogeographic realm	Amphibians	Birds	Mammals	Reptiles	Total
Australasian	545	1669	688	1305	4207
Antarctic	0	36	0	0	36
Afrotropical	930	2228	1161	1703	6022
Indo-Malayan	882	2000	940	1396	5218
Nearctic	298	696	481	470	1945
Neotropical	2732	3808	1282	2561	10,383
Oceanic	3	272	15	50	340
Palearctic	395	1528	903	774	3600
Total	5785	12,237	5470	8259	31,751

Due to human activities, all three characteristics have significantly decreased. The current extinction rate is about 48 species per million per year, up to 40 times the average Cenozoic extinction rate (Barnosky et al., 2011). Most endangered species are in the tropics, following the trend of species richness (Pereira et al., 2012). While animal biomass and species richness decline, human and domestic animal biomass, as well as human-made structures, are increasing. In 2020, the mass of buildings and infrastructure was 1100 Gt, and non-degradable plastic was 8 Gt (Elhacham et al., 2020). In 1900, humans carried 13 Mt of carbon, and domestic animals accounted for 35 Mt. Over the past century, these values have increased to 55 and 120 Mt, and they continue to grow rapidly (Smil, 2011).

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A.4. We have not investigated the role of microorganisms

Ecosystem processes largely depend on microorganisms and their metabolic activities (Schlesinger, 1997; Madsen, 2011). For example, microorganisms play a key role in the global carbon cycle by decomposing organic matter and producing carbon dioxide and methane, which are returned to the atmosphere (Conrad, 1996; Singh et al., 2010; Bridgham et al., 2012). Earth is a closed system where matter cycles between the lithosphere, atmosphere, hydrosphere, and biosphere. Microbial metabolism drives these biogeochemical cycles, ensuring a continuous replenishment of carbon dioxide through processes like photosynthesis.

Microorganisms are crucial for maintaining ecosystem health, climate stability, agriculture, and human well-being. Their biomass per unit area and carbon content vary across ecosystems. He et al. (2020) provided biomass estimates using methods like phospholipid fatty acids, microscopic data, CFU, respiration data, and glucosamine and muramic acid concentrations, presented in *Tables A.4.1.* and *A.4.2.* These values are for the upper 30 cm of soil. Interestingly, the highest values are found in cold climates like taiga and tundra, rather than in tropical rainforests. This is likely due to the high organic matter content in taiga and tundra soils (He et al., 2019).

For the rainforest, the biomass values are 64.42 and 51.58 g per hectare, which equal 0.6442 and 0.5158 tons. Dividing these by the rainforest equivalent value of 210 tons gives 0.0031 and 0.0025 equivalents.

The microbial metabolic quotient is the ratio of microbial respiration to biomass, indicating the activity of microbial communities in the soil. Its average value varies by biome and changes with microbial biomass levels. Xu et al. (2017) conducted a global

DOI: http://dx.doi.org/10.15666/aeer/2206_56175667

synthesis of published data. In a tropical forest, the microbial metabolic quotient is 1.35 mmol C mol MBC^{-1} h^{-1} (Microbial Biomass Carbon). Tropical forest microbial biomass carbon is 51.58 g per m², or 4.3 mol. 1.35 mmol equals 0.00135 mol. The ratio of 4.3 mol of biomass carbon per m² becomes 58 mol per hectare, or 696 g. Given the rainforest's biomass carbon content of 210 tons, the microbial metabolic quotient is quite low. *Table A.4.3.* shows microbial quotients across different habitat types.

Delgado-Baquerizo and Eldridge (2019) estimated global microbial diversity using 237 soil samples from all biomes and continents. They identified 25,424 OTUs using 16S rRNA analysis. The chart shows OTU distribution per biome. The highest species richness, or OTU count, is found in arid and temperate regions with lower soil carbon content and limited precipitation. Diversity negatively correlates with soil carbon content. *Figure A.4.1.* shows the geographic distribution of OTUs.

Table A.4.1. *Fungal and bacterial biomass carbon content per unit soil per biomolecule based on He et al. (2020)*

Ecosystems	Fungal biomass carbon (mg kg^{-1} soil)	Bacterial biomass carbon (mg kg^{-1} soil)
Unvegetated ground	192.74	24.6
Desert	16.92	6.83
Grassland	215.19	62.69
Pasture	632.15	270.65
Cropland	212.69	65.77
Shrub	218.14	45.42
Savanna	103.36	44.37
Tropical forest	451.4	209.96
Temperate forest	258.39	53.05
Boreal forest	1234.08	226.37
Tundra	3683.59	428.37
Wetlands	329.81	92.58

Table A.4.2. *Fungal and bacterial biomass carbon per unit area based on He et al. (2020)*

Ecosystem	Fungi biomass C density (g C m^{-2})	Bacteria biomass C density (g C m^{-2})
Boreal forest	304.44	58.66
Temperate forest	88.89	29.88
Tropical forest	64.42	51.58
Grassland	88.69	46.14
Shrub	48.06	17.31
Tundra	226.96	32.65
Desert	59.04	15.28
Wetlands	70.44	32.96
Croplands	67.61	30.09
Pasture	62.34	23.68
Globe	96.92	33.5

Table A.4.3. *Microbial metabolic ratio values per biome according to Xu et al. (2017)*

Ecosystems	**Soil (mmol C mol MBC^{-1} h^{-1})**
Bare soils/desert	3.21
Boreal forest	1.28
Cropland	3.11
Grassland	1.11
Natural wetlands	3.75
Pasture	1.12
Shrubland	1.96
Temperate broadleaf	1.19
Temperate conifer	0.99
Tropical/subtropical	1.35
Global average	2.01
Area-weighted global average	1.81

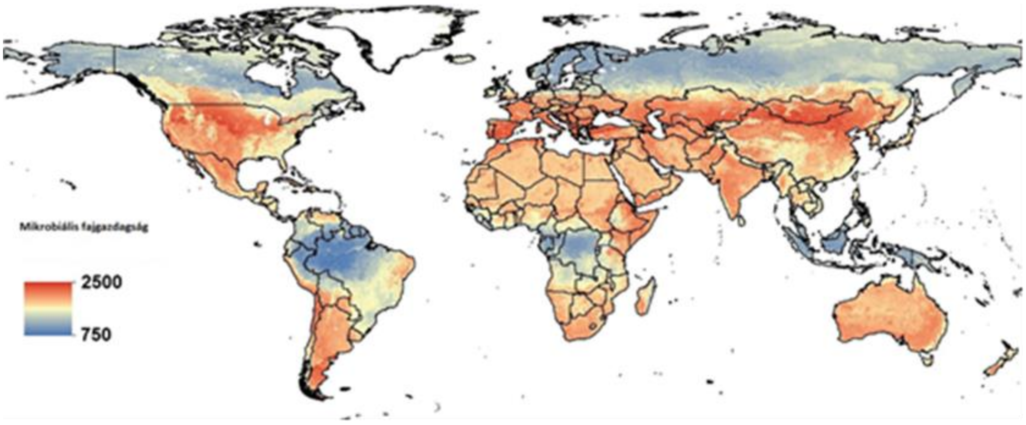

Figure A.4.1. *Geographical distribution of microbial OTUs (phylotype) according to Delgado-Baquerizo and Eldridge (2019)*

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A.5 We have not taken into account the seas and oceans, and to a limited extent, the mangrove habitat

In the sea, 207,821 animal species have been described (Mugnai et al., 2021). Marine habitats are less well-known than terrestrial ones, so many new species are expected to be discovered (Danovaro et al., 2017). Most known species are found in shallow waters, and species richness decreases with depth, although the proportion of unknown species is higher in deeper waters. Toward the poles, the decline in species richness is less pronounced due to the lack of extreme temperature changes in the water. Species richness is also higher on continental shelves, near continents, where there is more nutrient availability and higher production. Habitat complexity also affects species richness, which is not necessarily limited to the tropics (Costello and Chaudhary, 2017).

Terrestrial plants are excellent climate indicators because climate regulates their annual growth, reproduction, and senescence cycles. Seasonal phases in plant activity provide opportunities to observe climate-related changes. Terrestrial plants follow a 12-month cycle, while phytoplankton have much greater temporal variability, up to 100 times higher, due to their rapid growth and consumption rates (Calbet and Landry, 2004; Behrenfeld et al., 2006). Given these scale differences, differences in periodicity are expected.

While plankton phenology is not as distinct as that of plants, seasonal changes are still observed in temperate regions. Phytoplankton blooms occur in spring due to rising temperatures and increased radiation (Cushing, 1959; Sommer et al., 1986; Smayda, 1997; Friedland et al., 2017). These blooms last for a few weeks due to limited nutrients, cell sinking, and consumption. Another peak in nutrient abundance occurs in late summer and autumn (Longhurst, 1995). These blooms are sensitive to climate change, and their timing changes similarly to those on land (Edwards and Richardson, 2004; Winder and Schindler, 2004). *Table A.5.1* shows changes in photosynthetic pigment concentration, which approximates biomass changes.

In the Atlantic Ocean, chlorophyll-a concentration is higher in the temperate zone (>40°) and the tropical region (10° S – 20° N), but lower in the subtropical region between them. Despite the lower concentration, the subtropical region contributes significantly to global oceanic primary and export production due to its vast area (Karl et al., 1996; Karl, 1999; Lin et al., 2011). Picoplankton dominates phytoplankton in terms of NPP, chlorophyll-a, and cell density, though nano- and microplankton are also present in significant amounts (Zubkov et al., 1998; Marañón et al., 2000; Fernández et al., 2003).

In the equatorial region, phytoplankton biomass and NPP are high year-round (Pérez et al., 2005a, b). Picoplankton, mainly Prochlorococcus and Synechococcus, dominates, but Synechococcus spp., picoeukaryotes, and nanoflagellates also become more common (Zubkov et al., 1998; Vaulot et al., 2008). Pigments from dinoflagellates and diatoms are found in higher concentrations (Gibb et al., 2000; Barlow et al., 2002).

Table A.5.1. *Phytoplankton biomass and production in the seas, based on data from the International Ocean-Colour Coordinating Group (www.ioccg.org). Biomass and production are estimated on the basis of carbon or pigment per unit area or volume*

Biomass	Typical oceanic range (per unit volume)	Typical oceanic range (per unit area)
Mass of carbon per unit volume and area	10-60 μg C/L	1-2 g C/m^2
Mass of photosynthetic pigment per unit volume and area	0.01-2 μg Chl a/L	10-70 mg Chl a/m^2
Primary production	10-100 mg C/m^3/day	75-1000 mg C/m^2/day

Like in the subtropical zone, picoplankton remains dominant here (Marañón et al., 2000; Pérez et al., 2005b). Near coastal upwelling areas, NPP and chlorophyll-a concentration stay high, with diatoms and dinoflagellates becoming more prevalent (Gibb et al., 2000; Barlow et al., 2002). In these areas, biomass and NPP are dominated by nano- and microplankton (Marañón et al., 2000; Tarran et al., 2006).

The polar regions are the least studied in this regard. Research is challenging due to harsh conditions, continuous ice cover, long winter darkness, rapid seasonal changes, and complex physical-biological interactions in the marine ecosystem (Lee et al., 2015). Since the 1990s, phytoplankton NPP has increased by 30% due to longer seasons and reduced ice cover (Arrigo and van Dijken, 2015). Ice and snow limit light penetration, but thinning ice allows more light to reach the ocean (Arrigo and van Dijken, 2011).

For example, the Chukchi Sea’s shallow waters have become more productive due to a longer warm period, leading to summer algal blooms (Arrigo et al., 2008; Yun et al., 2019). However, seawater salinity has decreased, and the Ekman transport effect has strengthened (McPhee et al., 2009; Mauritzen, 2012). This causes the deepening of the nitrocline and chlorophyll maximum (McLaughlin and Carmack, 2010). The stronger Ekman effect and stratification reduce nutrient supply to the euphotic layer, decreasing NPP in the summer around Canada. Reduced salinity appears to negatively affect marine phytoplankton production (Yun et al., 2014).

In the Arctic region, NPP on continental shelves reaches about 225 mg C/m^2/day or 27 g C/m^2/year, while in the open ocean it is around 75 mg C/m^2/day or 9 g C/m^2/year (Subba Rao and Platt, 1984). Pabi et al. (2008) reported NPP in the Arctic from March to September as 420 ± 26 mg C/m^2/day, with an annual average of 419 ± 33 Tg from 1998-2006 and a 26% variation per year.

Joo et al. (2012) identified 71 taxa in the Chukchi and Beaufort Seas, including Dinophyceae, Cryptophyceae, Bacillariophyceae, Chrysophyceae, Dictyochophyceae, Prasinophyceae, and Prymnesiophyceae. In the Bering Sea, Fragilariopsis was most common at the surface, while Cryptomonas sp. dominated at the Subsurface Chlorophyll Maximum (SCM). In the Bering Strait, Phaeocystis sp. was most common at both surface and SCM depth. Chaetoceros sp. dominated at the surface in the Chukchi Sea, while Halosphaera sp. was dominant at the surface in the Canadian Basin, with Navicula sp. dominant at the SCM depth. *Table A.5.1* summarises the phytoplankton data.

In the ocean, the highest NPP values do not necessarily occur at the equator, unlike on land. Nutrient availability limits NPP, and various factors affect nutrient concentrations. Rare events can cause sudden increases in nutrients. For example, Hamme et al. (2010) reported increased iron levels after a volcanic eruption, which boosted NPP. Light is also crucial for photosynthesis, so its intensity, duration, and depth of penetration are important for algae. In polar regions, long winter darkness limits NPP due to a lack of light (Sigman and Hain, 2012). Some ocean areas experience upwelling, where currents bring nutrient-rich water to the surface, increasing productivity (Gómez-Letona et al., 2017). Conversely, low nutrient levels can result in low algal productivity (Sigman and Hain, 2012).

Seagrasses belong to two families, Potamogetonaceae and Hydrocharitaceae (Alismatales), which include 12 genera and about 50 species (Hemminga and Duarte, 2000). Other sources divide them into three families: Hydrocharitaceae, Cymodoceaceae, and Zosteraceae (Les et al., 1997; Ross et al., 2016). Seagrasses are found worldwide, except in polar regions (Orth et al., 2006). They regulate the physical, chemical, and biological characteristics of their surroundings and provide many ecosystem services (Constanza et al., 1997; Hemminga and Duarte, 2000; Wright and Jones, 2006).

Seagrasses are important food sources for marine turtles (Chelonia mydas), dugongs (Dugong dugon), and manatees (Trichechus spp.). They also provide habitat for many fish species (Beck et al., 2001). By stabilising sediments, seagrasses can store large amounts of carbon (Orth et al., 2006). They have developed unique adaptations to underwater life, including internal gas transport, epidermal chloroplasts, underwater pollination, and propagation (den Hartog, 1970; Papenbrock, 2012). Like other photosynthetic marine organisms, the abundance and distribution of seagrasses along vertical gradients largely depend on light availability.

As ocean depth increases, light decreases rapidly, challenging plant photosynthetic machinery, which very few species can handle. Underwater radiation is determined by the light attenuation coefficient (k), which depends on factors like eutrophication, turbidity, sedimentation, latitude, and topography (Minguito-Frutos et al., 2023). This makes seagrasses highly sensitive to changes in water quality. Due to the declining water quality from human activities, fishing practices, and invasive species, seagrass meadows are continually shrinking (Orth et al., 2006).

Kelp forests grow in cold, shallow, rocky coastal waters and are primarily composed of brown algae from the Laminariales order (Dayton, 1985; Bolton, 2010). The largest species, Macrocystis, can reach up to 45 m and are found on the western coasts of North and South America. Nereocystis leutkeana dominates from central California to Alaska, Ecklonia maxima in South Africa, and Alaria fistulosa in East Asia. Globally, only 20 species form these forests, distributed across 16 mostly monotypic genera (Abbott and Hollenberg, 1976; Reed and Brzezinski, 2009; Smale, 2020).

Kelp forests are morphologically diverse, with different layers similar to terrestrial rainforests (prostrate, stipitate, and canopy forms) (Dayton, 1985; Smale and Moore, 2017). This structural diversity supports various marine mammals, fish, crustaceans, sea urchins, molluscs, and other algae, making it one of the world’s most species-rich ecosystems (Mann, 1973; Port et al., 2016). Despite high productivity, individual kelp organisms have short lifespans, typically living only 2-5 years and reaching maximum length in 1-3 years (Steneck and Dethier, 1994).

In temperate zones, mangrove forests are replaced by salt marshes, which are formed by halophytic herbaceous plants in the intertidal zone. These marshes develop in areas with low wave action, allowing sediment to settle and accumulate, creating a suitable habitat for salt-tolerant plants. Near river deltas, salt marshes often gradually transition into freshwater marshes (Allen and Pye, 1992; Wang et al., 2023).

Open water areas often interrupt the vegetation, creating complex habitats shaped by tidal movements (tidal creeks) (Allen, 2000). Zonation occurs based on distance from the sea and elevation (Chapman, 1974; Moffett et al., 2010). There is an inverse relationship between a plant's competitive ability and its stress tolerance. Species with stronger competitive abilities occupy less stressful areas, while those with weaker abilities are confined to more stressful zones, such as areas with high salt concentration and oxygen depletion (Bertness, 1992; Pennings and Bertness, 2001).

In North America, Spartina alterniflora dominates the US coast, while Puccinellia phryganodes is common on the Canadian coast up to the Arctic (Poaceae). In Western Europe, Spartina maritima and Spartina anglica are most common. Other characteristic plants include Limonium nashii, Salicornia spp., Aster tenuifolius, Distichlis spicata, Gerardia maritima, Spartina patens, Spergularia marina, and Ruppia maritima. Typical large algae species are Ascophyllum nodosum, Fucus vesiculosus, Enteromorpha, Ulva spp., and Codium fragile. The biomass of A. nodosum is often greater than S. alterniflora. Diatoms are the most characteristic microscopic algae (Teal, 1986; Rinke et al., 2021).

These marshes provide habitat for many arthropods, birds, and mammals (Vince, 1979; Wong et al., 2011). They serve as breeding and nursery grounds for fish and play a key role in erosion control (Levin et al., 2001). Environmental pollution has caused heavy metal accumulation in sediments and plant tissues (Giblin, 1985; Tupan and Azrianingsih, 2016). Overgrazing threatens marshes by altering plant communities and animal populations (Knottnerus, 2005). Some marshes are diked off for agriculture (Reise, 2005), and salt production also poses a threat by creating ponds and removing vegetation (Laszlo, 2001). *Table A.5.2* summarises data on seagrasses, macroalgae, and salt marshes, while *Tables A.5.3* and *A.5.4* provide data on net primary productivity (NPP) in marine realms.

Table A.5.2. *Multicellular algae (kelp), seagrass and salt marsh biomass and production data. Production was measured as assimilated carbon. For kelp and seagrass, I found carbon stock data in the form of Mg CO_2 e/ha*

	Kelp forest	
Carbon stock	120-720 g C/m^2	Weston et al., 2012
Assimilated carbon	400 g/m^2/	Yee, 2010
	Sea grass	
Carbon stock	7.29 ± 1.52 Mg C/ha	Fourqurean et al., 2012
Assimilated carbon	Between -77 and + 85 Mg CO_2 e/ha/year	Sifleet et al., 2011
	Salt marsh	
Biomass	5.1-18.3 Mg CO_2 e/ha	Sifleet et al., 2011
Assimilated carbon	0.01-62.81 Mg CO_2 e ha/year	Sifleet et al., 2011

By slightly modifying the concept, it can also be used to describe aquatic habitats. The net primary productivity (NPP) of phytoplankton depends on the amount of dissolved nutrients in the water. According to Taelman et al. (2014), unlike on land, the highest ocean NPP value (exergy-based potential NPP) is found in the temperate Pacific. The calculation method is detailed in their paper (Eppley-VGPM model). *Tables A.5.5* and *A.5.6* summarise the exergy-based potential NPP values and their area-weighted products.

Table A.5.3. *NPP values for biogeographical realms. Spatial data are from Spalding et al. (2007)*

Realm	Area (km^2)	Mean potential NPP (MJexm^{-2} yr^{-1})	NPP per realm (MJexrealm^{-1} yr^{-1})
Central Indo-Pacific	$1.20*10^{7}$	10.12	$1.22*10^{8}$
Eastern Indo-Pacific	$2.78*10^{7}$	1.92	$5.34*10^{7}$
Open Ocean	$1.89*10^{7}$	1.67	$3.16*10^{7}$
Southern Ocean	$2.24*10^{8}$	3.72	$8.32*10^{8}$
Temperate Australasia	$1.03*10^{7}$	1.51	$1.56*10^{7}$
Temperate Northern Atlantic	$5.65*10^{6}$	5.48	$3.10*10^{7}$
Temperate Northern Pacific	$9.85*10^{6}$	15.96	$1.57*10^{8}$
Temperate South America	$9.65*10^{6}$	16.18	$1.56*10^{8}$
Temperate Southern Africa	$6.04*10^{6}$	10.6	$6.40*10^{7}$
Tropical Atlantic	$1.99*10^{6}$	14.48	$2.88*10^{7}$
Tropical Eastern Pacific	$1.37*10^{7}$	4.87	$6.79*10^{7}$
Western Indo-Pacific	$4.19*10^{6}$	7.04	$2.95*10^{7}$
Marine systems	$1.53*10^{7}$		

Table A.5.4. *NPP values for biogeographic realms in descending order. Since the Temperate Northern Pacific has the highest value, this provides the basis for the calculation of the equivalent value*

Realm	Area (km^2)	Mean potential NPP (MJexm^{-2} yr^{-1})	NPP per realm (MJexrealm^{-1} yr^{-1})
Temperate Northern Pacific	$1.20*10^{7}$	10.12	$1.22*10^{8}$
Temperate Northern Atlantic	$2.78*10^{7}$	1.92	$5.34*10^{8}$
Temperate Southern Africa	$1.89*10^{7}$	1.67	$3.16*10^{7}$
Temperate South America	$2.24*10^{8}$	3.72	$8.32*10^{8}$
Arctic	$1.03*10^{7}$	1.51	$1.56*10^{7}$
Tropical Eastern Pacific	$5.65*10^{6}$	5.48	3.10*10
Temperate Australasia	$9.85*10^{6}$	15.96	$1.57*10^{8}$
Tropical Atlantic	$9.65*10^{6}$	16.18	$1.56*10^{8}$
Western Indo-Pacific	$6.04*10^{6}$	10.6	$6.40*10^{7}$
Open ocean	$1.99*10^{6}$	14.48	$2.88*10^{7}$
Central Indo-Pacific	$1.39*10^{7}$	4.87	$6.79*10^{7}$
Eastern Indo-Pacific	$4.19*10^{6}$	7.04	$2.95*10^{7}$
Southern Ocean	$1.53*10^{7}$	10.12	

Table A.5.5. *The equivalents for marine biogeographic realms. Calculated according to the method already presented for terrestrial plants*

Realm	Equivalence
Temperate Northern Pacific	1
Temperate Northern Atlantic	0.986402967
Temperate Southern Africa	0.894932015
Temperate South America	0.65512979

Arctic	0.625463535
Tropical Eastern Pacific	0.435105068
Temperate Australasia	0.33868974
Tropical Atlantic	0.300988875
Western Indo-Pacific	0.235475896
Open ocean	0.229913473
Central Indo-Pacific	0.118665019
Eastern Indo-Pacific	0.103213844
Southern Ocean	0.093325093

Table A.5.6. *The product of the equivalent and the area data.*

Realm	Equals and area multiplied
Temperate Northern Pacific	$9.65*10^6$
Temperate Northern Atlantic	$9.72*10^6$
Temperate Southern Africa	$1.78*10^6$
Temperate South America	$3.95*10^6$
Arctic	$7.51*10^6$
Tropical Eastern Pacific	$1.82*10^6$
Temperate Australasia	$1.92*10^6$
Tropical Atlantic	$4.19*10^6$
Western Indo-Pacific	$3.61*10^6$
Open ocean	$5.14*10^7$
Central Indo-Pacific	$3.30*10^6$
Eastern Indo-Pacific	$1.96*10^6$
Southern Ocean	$9.63*10^5$

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