Aquatic Biodiversity Impact Assessment – first ideas

White Paper in the context of the research project Biodiversity Valuing & Valuation (BioVal) Phase 1

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Content

1 Introduction and aim

This white paper is part of the research project 'Biodiversity Valuing & Valuation (BioVal) - Phase 1'. The results have been included in the application for phase 2 of the BioVal research project, which is being applied for by the ZNU - Centre for Sustainable Leadership at the University of Witten/Herdecke together with the joint partners Zentrum Technik und Gesellschaft (ZTG) at the Technical University of Berlin, Bochum University of Applied Sciences, Frosta AG, Alfred Ritter GmbH & Co.KG and Seeberger GmbH.

BioVal develops solutions to reduce negative impacts on biodiversity from food along the life cycle. Together with the named companies, it is investigated how they can protect and promote biodiversity along the product life cycle, incorporate this into their management processes and communicate it effectively. BioVal focuses on the food industry, which has the greatest influence on the protection and conservation of biodiversity due to the use of a wide variety of ecosystems (Benton et al., o. J.). It addresses the overarching research question of how to increase biodiversity along the food life cycle from three perspectives: the social perspective on biodiversity, the methodological perspective on the impact assessment of products on biodiversity, and the management perspective.

Aim of the present White Paper is to put together 'first ideas' on the impact assessment of aquatic biodiversity in Life Cycle Assessment (LCA) and thus give input to the methodological development of a Life Cycle Impact Assessment (LCIA) method for aquatic biodiversity to be developed in phase 2 of BioVal. The method will be based on a yet developed LCIA method for terrestrial biodiversity (Lindner et al., 2019b).

2 Impact assessment method for terrestrial biodiversity

Global food systems have the greatest impact on biodiversity loss worldwide (Benton et al., o. J.; Campbell et al., 2017). However, in LCA biodiversity loss is not commonly addressed due to lack of an appropriate methodology.

In 2016 Lindner proposed a method for terrestrial biodiversity impact assessment that was further developed in recent years (Lindner et al., 2019a&b; Lindner et al., 2021) and that will also form the basis for the development of the LCIA method for aquatic biodiversity in BioVal phase 2. The LCIA method (Lindner et al., 2019b) allows to quantify the impacts on terrestrial biodiversity of land-using processes for several land use classes (e.g. forestry, arable land) and it provides a calculation framework. It is based on the framework developed by the UNEP-SETAC Life Cycle Initiative (today: UN Environment Life Cycle Initiative) that defined the impacts of land use as a function of quality (Q), area (A) and time (t) (Milà i Canals et al., 2007; Koellner et al., 2013) (Equation 1).

$$impact = \Delta Q \cdot A \cdot \Delta t$$

Equation 1: Surface-based impact assessment (land use); ΔQ = characterisation factor; A*Δt = inventory quantity

The method of Lindner et al. (2019b) assesses the biodiversity impact on the used land by multiplication of the required area time[1] with a characterisation factor. The characterisation factor results from the quality difference between the maximum possible local biodiversity value (natural vegetation) and the local biodiversity value given by the use. The local biodiversity value depends on the intensity of use and depends on the type of land use (forestry, arable land, mining, resource extraction).

The intensity can be determined in two ways:

- generically through the use of hemeroby values (Annex, Table 1): here, the local biodiversity value is determined by the degree of naturalness (hemeroby) within the respective type of land use. A total of seven hemeroby levels are specified, ranging from natural to artificial or maximum remoteness from nature. For example, a primeval forest not used by humans is assigned to hemeroby level 1, a car park to hemeroby level 7.
- specifically through specific input parameters whose impact on biodiversity is assessed in each case through potential field functions and aggregated to a biodiversity value (BVLU) (Annex, Figure 1). This BVLU is then multiplied by the difference between the maximum and minimum values of the use class-specific hemeroby interval and added to the minimum value of the interval.

[1] The area time indicates how much area is used and how long this area is occupied in order to provide the required quantities of products for consumption. The unit is the area time (m^2*a).

The resulting difference is standardised using so-called ecoregion[2] factors, which quantify the value of the ecoregions. Each of these ecoregions is assigned an ecological value. This includes an assessment based on parameters such as the proportion of wetlands, forest, road-free areas and the global probability of extinction. The scale ranges from 0.035 to 0.519. The higher the value, the higher the ecological value of the respective ecoregion. The result is expressed as a biodiversity difference and indicates the degree of impact on biodiversity - the Biodiversity Value Increment (BVI).

This impact assessment for terrestrial biodiversity builds the basis of the method development in BioVal.

3 Aquatic biodiversity – first ideas for an impact assessment method

As part of BioVal Phase 1, a workshop was held with LCA experts and marine biodiversity experts to discuss initial approaches to developing an impact assessment method for marine biodiversity. In addition, a subcontract was awarded to develop initial ideas for possible use classes of freshwaters and to collect literature on relevant factors that influence the ecological status of freshwaters.

Aquatic ecosystems differ from terrestrial ecosystems, because material flows are much more interconnected. This poses new challenges for the development of an impact assessment method for aquatic biodiversity:

- First of all, it must be discussed how the inventory quantity can be defined. In the case of terrestrial biodiversity, it is area * time (see Equation 1). Is this also adequate for aquatic ecosystems?
- The next question is how use classes - analogous to the impact assessment method of Lindner et al. (2019b) - can be defined, and
- which input parameters could be relevant.

In the following two sections the first ideas collected are presented.

2.1. Marine biodiversity

The great importance of marine ecosystems can be shown by the ecosystem services provided. Photosynthetically active algae and seagrasses, for instance, are responsible for about half of the oxygen found in the atmosphere. Additionally, oceanic ecosystems provide us with more than 100 million tons of fish per year (Jacob & Hillebrand, 2020). Among others, these are a few examples of important marine ecosystem services humans rely on.

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[2] The World Wide Fund for Nature (WWF) defines over 800 terrestrial ecoregions based on influencing factors such as climate, geology and historical species development (https://www.worldwildlife.org/biome-categories/terrestrial-ecoregions).

Marine ecosystems are more interconnected than terrestrial ecosystems, which makes it harder to isolate cause-effect chains. Furthermore, impacts on marine ecosystems arise not only from sea use; many impacts originate on land, e. g. from agricultural run-offs (N/P-emissions, toxic emissions from pesticides), from atmospheric deposition (S/N deposition, from untreated wastewater and littering.

Moreover, in contrast to terrestrial ecosystems zoning of marine ecosystems is defined by abiotic factors such as temperature, salinity, light availability, gas content, tides, currents, wind, water depth and water pressure. Today, most existing biogeographical subdivisions lack open ocean areas or ocean stratification (e. g. Olson & Dinerstein, 2002) and are sometimes very small-scale and detailed. Additionally, a purely geographical distinction is not necessarily meaningful due to the three-dimensionality and strong interconnectedness of the individual areas (Brand, 2020).

In the workshop in February 2021 first ideas presented by BioVal have been discussed regarding the inventory quantity, use classes, and input parameters.

Inventory quantity

First ideas for an adequate inventory quantity for marine biodiversity included the 'sea surface area', the 'seafloor area', a 'water column volume' or the 'length of a coastal strip'.

The participants were split on the issue of an appropriate inventory quantity. Some favoured one single inventory unit like m^2 * year as used in terrestrial land use LCIA, while others stressed the importance of measuring the three-dimensional space of marine ecosystems. A couple of experts mentioned that the inventory quantity needs to be specified for the different use classes.

Use classes

Marine ecosystem services can be divided in (Jacob & Hillebrand, 2020):

- Regulatory services: e. g. climate regulation, protection from storms and floods, as well as coastal stabilisation.
- Provisioning services: e. g. fisheries, aquaculture and water supply
- Cultural services: e. g. recreation, tourism and the waterways

The use of the marine ecosystem can be split in different use classes like fisheries, aquaculture, sea transport, installations & constructions (e. g. wind power, dredging, coastal protection), and tourism. In many aquatic regions, spatial overlap of use classes may occur. Some uses can have positive impacts, e. g. off-shore wind turbines can have a positive impact on fish stocks as there is no (or less) fishing in these areas.

Input parameters

Input parameters should be set specific for the different use classes. The following use classes proposed by Brand (2020) have been discussed in the experts' workshop.

Fisheries

With respect to fisheries the following input parameters have been discussed:

- Catch: amount, age
- Catching method: size & species selectivity, timing, duration
- Catching apparatus: size, weight
- Vessel: size, noise, antifouling, oil spillage

Furthermore, some parameters are more related to the boundary conditions of an activity than the activity itself (e. g. benthic substrate relevant for impact severity of trawling).

Aquaculture

With respect to aquaculture the following input parameters have been discussed:

- Fish: wild-caught juveniles, stocking density
- Installation: total size (volume), antifouling agents
- Inputs: rate fish-in / fish-out, N / P input, pharmaceuticals, plastic waste
- Outputs: effluent contamination, subsequent filtrating ecosystems, escapes

Sea transport

With respect to sea transport the following input parameters have been discussed:

- General characteristics of the vessel: size of the vessel, age of the vessel
- Emissions: noise, antifouling, oil spillage, sulphur emissions
- Migration opportunities: ballast water volume, ballast water cleaning

Installations & constructions

With respect to installations & constructions the following input parameters have been discussed:

- General characteristics of installation: size of installation, age of installation, density of multiple installations
- Emissions: noise (construction phase, use phase), antifouling & anticorrosion agents, oil spillage
- Seafloor / underground interactions: depth of cables, depth of foundation

Tourism

With respect to tourism the following input parameters have been discussed:

- Vessel(s): noise, antifouling agents
- Disturbance by visits: frequency of visits, duration of visits
- Emissions: contamination, littering

2.2. Freshwater biodiversity

Freshwater ecosystems, especially inland waters, are considered sensitive indicator systems for changes in the environment. Natural stress limits are being exhausted by intensive use pressures, leading to changes in water quality, habitat alteration and loss, and other pressures that negatively impact limnetic biodiversity. According to a report by the World Wide Fund for Nature (WWF), the population of vertebrates in freshwaters has declined by more than 80 % in the last 50 years (WWF, 2016). Furthermore, the World Economic Forum (WEF) ranks freshwater availability as number three of the top ten global risks - only weapons of mass destruction and extreme weather events are ranked as more threatening (WEF, 2017). The sharp decline of biodiversity in freshwaters is referred to by some scientists as a 'global biodiversity crisis' (Darwall et al., 2018).

Depending on the point of view, various water pressures can be listed as significant drivers for the decline of limnic biodiversity. The main drivers at the global level include:

- Global warming and / or warming of water bodies
- Changes in the flow of water
- Exploitation or overexploitation
- Invasive species
- Pollution
- Land use change

These and other drivers lead to water problems that have a negative impact on biodiversity. Persistent water body problems include, above all:

- Changes in water quality (e. g. warming, pollution, eutrophication, acidification).
- Destruction and alteration of natural habitats (e.g. lateral and longitudinal fragmentation, watercourse maintenance measures, bottom clearing, changes to the river bed)
- Resource extraction (e. g. overfishing, water abstraction, siltation)
- Invasive species

Characterisation of freshwater ecosystems

According to e. g. Guderian & Gunkel (2000) freshwater ecosystems can basically be characterised by three criteria. While the characterisation of the origin of the water bodies is of great importance from a biogeographical point of view, it is helpful to differentiate between water bodies on the basis of their morphology in order to record their condition and pollution. Another aspect that plays a role is the location of water bodies, which primarily describes the climatic conditions and main climate zones (arctic, temperate and tropical regions), but also includes regional differences (such as equatorial high-mountain lakes in the Andes of South America). In addition to direct influences of climate (e. g. irradiation, water balance and heat balance), indirect influences also occur via biotic and abiotic processes in the catchment area (e. g. through different substance inputs depending on rock weathering and biomass mineralisation). The following is a rough classification of freshwater body types based on their morphology (the type and size) (Guderian & Gunkel, 2000):

- Watercourses (characterised by width):
  - Rivers and streams with > 5 m width
  - Trickles and streams with < 5 m width
  - Estuaries (tidally influenced lower reaches - periodic change in water level, influence of salt water)

- Lakes (characterised by depth, volume, proportion of the shore region and possible stratification processes)
  - large, deep lakes (e. g. the Great Lakes of North America)
  - medium-sized, deep lakes (e. g. the Alpine waters)
  - shallow lakes with a depth of < 10 m (e. g. Rangsdorfer See in Berlin: water surface 250 ha, average depth 1.5 m, maximum depth 2.5 m, thickness of biogenic sediments 15 m)
  - Small water bodies (e. g. ponds and pools)
  - periodic waters (e. g. steppe lakes, rainwater depressions)
  - dams as artificial water bodies

Ecoregions of freshwater ecosystems

Moreover, freshwater ecosystems can be divided into ecoregions. Analogous to the division of the world into 825 terrestrial ecoregions (Olson & Dinerstein, 2002), 426 freshwater ecoregions (Abell et al., 2008) were determined. Similar to the 14 biomes of the terrestrial ecoregions, twelve biomes ('major habitat types') are identified for the freshwater ecoregions:

- large lakes, e. g. Lake Malawi in Africa or Lake Baikal in Siberia (1)
- large river deltas, e. g. Niger in Africa (2)
- montane freshwaters, e. g. the crater lakes in the Andes of South America (3)
- xeric freshwaters and endorheic (closed) basins, e.g. the lower Nile (4)
- temperate coastal rivers, e. g. North Pacific coastal region and South Atlantic ecoregions in North America (5)
- temperate upland rivers, e. g. Ozark Plateau and Ouachita Highlands in North America (6)
- Temperate Floodplain Rivers and Wetland Complexes e.g. Mississippi and Middle Missouri Rivers (7)
- tropical and subtropical coastal rivers, e. g. Kenyan coastal areas and Mata Atlantica (8)
- tropical and subtropical upland rivers, e. g. upper Niger, Brazilian Shield (9)
- tropical and subtropical floodplain rivers and wetland complexes, e. g. lower Congo, Amazon lowlands, and Orinoco llanos (10)
- polar freshwaters, e. g. Lena in Siberia and the Yukon in Alaska (11)
- oceanic islands, e. g. Fiji and Hawaiian Islands (12)

A freshwater ecoregion is defined (Abell et al., 2008) as a large interconnected area containing one or more limnic ecosystems consisting of a diversity of endemic communities and species. In this context, the species and environmental conditions occurring within an ecoregion are more similar than those of the surrounding ecoregions. The mapping of global freshwater ecoregions can be seen as a kind of complement to terrestrial and marine ecoregions, but differs in the way it is defined. The classification or definition of freshwater ecoregions is mainly based on synthesised data on endemism, species richness and threats to fish, amphibians, turtles and crocodiles found in freshwaters. Further information (topography, climate, habitats, ecological and evolutionary phenomena, biodiversity, etc.) can be found in the description of individual ecoregions. A threat analysis based on the evaluation of global information on land use change, nearby major cities, irrigated areas in the region, anthropogenic impacts and water stress is included for each ecoregion (Abell et al., 2008).

Environmental impacts on freshwater ecosystems

Environmental impacts on freshwater ecosystems, primarily caused by the impact of anthropogenic influences, have long been the subject of research (Guderian & Gunkel, 2000). However, due to the underlying complexity of ecosystems, quantifying and assessing environmental changes requires decades of observation and scientific research. In order to record and quantify possible impacts, the following criteria are particularly suitable (Guderian & Gunkel, 2000):

- concentration describes the amount of a substance dissolved or suspended in water [mg / l].
- the load describes the quantity of a substance discharged per unit of time [mg / a].
- the effect on organisms can be described, e. g. via mortality, in comparison with a reference condition [%].
- the change in entire ecosystems is determined qualitatively or semi-quantitatively via regional reference waters.

According to Guderian & Gunkel (2000), these four criteria must be used simultaneously and conscientiously for the assessment of the individual pollution factors, i. e. the most sensitive criterion must be applied. The following example (Gunkel & Guderian, 2000) serves to illustrate this: If the toxic effect of the input of ammonia in a water body is to be monitored and assessed, this must be done via the concentration, since under ecological aspects the toxic effect is the central effect. The situation is different in coastal waters. There, the load is of central importance, since in coastal waters the limitation of the plankton biocoenosis (N or P limitation) is regulated by the substance inputs as loads.

Stress factors of freshwater ecosystems are classified according to the type of disturbance and the origin of the source of disturbance (Guderian & Gunkel, 2000; LAWA, 2018).

The type of disturbance can be divided into the following categories (Guderian & Gunkel, 2000):

- substances or chemicals (e.g. discharge of nutrients or eco-toxic chemicals):
  - from anthropogenic sources, mainly xenobiotics, which include about 50,000 compounds (5,000 - 10,000 of which are considered toxic)
  - inorganic compounds have a toxic effect in dissolved form (e.g. salts, acids, heavy metals, aluminium, inorganic surfactants, but also protons (H+) as acidifying ions).
  - organic compounds, especially plant and animal treatment agents (e.g. pesticides (fungicides, bactericides, herbicides, nematocides, insecticides...), organic surfactants, pharmaceuticals, halogenated hydrocarbons and aromatics, from natural sources: Mycotoxins and algae-borne pollutants).

  - organometallic compounds (e. g. methylmercury) cannot be clearly assigned to the aforementioned, have both ionic and lipophilic effects.
  - wastewater discharges, which, in addition to direct exposure to ecotoxic compounds, can lead to oxygen depletion through inputs of degradable substances and through oxidation of reduced inorganic compounds
  - oxygen depletion through direct inputs of degradable substances or reduced inorganic compounds
  - suspended matter leading to turbidity of the water and disturbance of the biocoenosis
  - input of surfactants leads to a change in surface tension. Lethal for the biocoenosis of the neuston and pleuston, the micro- and macro organisms at the air / water interface (e. g. the water strider Gerris sp.), toxic effects on other aquatic organisms also occur.
  - input of radionuclides leads to increased radiation exposure, which is a burden for aquatic organisms, but also for humans as users of aquatic systems. The main exposure pathway for almost all radionuclides is via the sediment due to its high adsorption capacity.
- Mechanical or morphological stresses (e. g. changes in water flow or damming):
  - change in water flow, damage due to e. g. extreme runoff or low and high water. Organisms are washed away; low water can lead to extreme warming and oxygen deficiency. Peak flows are caused by watercourse development and straightening, as well as sealing and deforestation in the catchment area. Discharges and water withdrawals can also lead to a change in water flow or to extreme discharges and must also be taken into account
  - resuspension of deposited sediments, e. g. through dust management (regular opening of the bottom outlet) or through shipping / screw. Resuspension of sediments primarily leads to oxygen deficiency and can result in the release of pollutants by intensifying the decomposition of organic matter.
  - measures of watercourse maintenance (weeding, mowing, clearing, basic clearing) have a direct effect on the biocoenosis of flowing waters and can lead to habitat loss and the removal of organisms.
  - bottom stabilisation and bank protection can lead to impairment of the water biocoenosis, e. g. through loss of habitat.
  - impoundments (weirs, dams) lead to a change in flow conditions and corresponding mechanical impairment (e. g. silting of the impounded area, interruption of passability).
  - increased sand input or silting of a watercourse section leads to over-sanding (destruction of fauna and flora); in addition, regular ground clearing is necessary to remove sand.
  - mechanical damage through trampling etc. (e. g. through recreational use and livestock farming) leads to the destruction of the flora and fauna close to the shore, but also to increased nutrient input through erosion.

- Physical pressures (e. g. discharge of cooling water or warming as a result of anthropogenic climate change):
  - influence of UV-B radiation on water bodies, flora and fauna (rather unexplored).
  - temperature increase, e. g. through discharge of cooling water, leads to a decrease in oxygen saturation concentration. Long-term temperature fluctuations regulate or limit the occurrence of stenothermic organisms with a low temperature tolerance.
  - acoustic and visual disturbances (e. g. from off-shore wind turbines) lead to stress and impair reproductive performance, among other things.
- Biological stress factors (e. g. invasive species or parasites)
  - competitive factors
  - neozoa or neophytes
  - parasitism

Stress factors can be of both natural and anthropogenic origin. A further classification of these is usually made by the origin of the disturbance:

- direct inputs from point sources (e. g. through watercourse discharge).
- diffuse inputs from the catchment area (e. g. nutrient inputs from agriculture)
- the in-situ origin of pressures (e. g. occurrence of oxygen deficits due to the decomposition of organic substances - saprobisation).

According to Guderian & Gunkel (2000), the following anthropogenic pressures are of particular importance:

- input of nutrients (nitrogen, phosphorus, silicon) and thus also the eutrophication of water bodies,
- input of protons from the atmosphere leads to anthropogenic acidification; geogenic acidification as a result of leaching processes of groundwater in rocks (e. g. from mining tailings).
- input of xenobiotics,
- input of pharmaceuticals (human and veterinary drugs),
- input of surfactants and mineral oils,
- input of PAHs (polycyclic aromatic hydrocarbons) from (incomplete) combustion of fossil fuels,
- increase in UV-B radiation due to ozone depletion in the atmosphere; occurrence of UV-B damage in aquatic organisms,
- technical development of water bodies to improve water drainage (straightening, control profile, bank protection, bed protection),
- construction of barrages for water quantity management; as a result, structural poverty and reduction of settlement occur,
- sediment resuspension by shipping,
- maintenance of water bodies (e. g. desilting, weeding),

- discharge of wastewater (municipal and industrial wastewater…),
- discharge of cooling water,
- discharge of rainwater and drainage water; in addition to substance inputs, an increase in peak discharge values occurs,
- withdrawal of water (e. g. for industry, agriculture or as drinking water); leads to a lowering of the minimum water flow and can possibly mean a lowering of the minimum water flow,
- use of water bodies for tourism and recreational activities; mechanical damage (trampling) and disturbance of organisms (e. g. flushing).

4 Conclusion

The research and initial discussions with experts on the development of an impact assessment method for marine and freshwater biodiversity indicate that it is possible in principle to use the method developed by Lindner et al. (2019b) as a starting point. However, there are some challenges. For example, the experts were divided on the choice of inventory quantity, so there is a need for further research here. The same applies to the use classes for marine biodiversity. Here, too, the development of an impact assessment method for marine biodiversity will involve defining these more precisely, working out the overlaps of some use classes and, in particular, clarifying the question of whether overlaps are relevant. The experts interviewed were nevertheless of the opinion that the overlaps can probably be neglected for most products.

With regard to the development of an impact assessment method for freshwater biodiversity, the differentiation from terrestrial ecosystems (e. g. small ponds) or marine ecosystems (e. g. estuaries) will be an important point of discussion, as will the definition of an adequate inventory quantity.

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6 Annex

Table 1: Allocation of local biodiversity value according to hemeroby and land use classes [3]

Hemeroby level	Land use classes				BV_{LU}
	forest	grassland/ savannah	arable land	resource-use	
1 – natural	virgin forest or no longer used	/	/	/	1.000
2 – near-natural	very near-natural forestry	near-natural grassland	/	/	0.983
3 – partly near-natural	extensive forestry	extensively used grassland	highly diverse agroforestry	/	0.950
4 – semi-natural	middle intensive forestry	middle lintensive used grassland	extensive agriculture	/	0.884
5 – partly far from natural	intensive forestry	intensively used grassland	middle intensive agriculture	high structural diversity	0.754
6 – far from natural	/	/	intensive agriculture	low structural diversity	0.500
7- artificial	/	/	/	sealed or devastated area	0.000

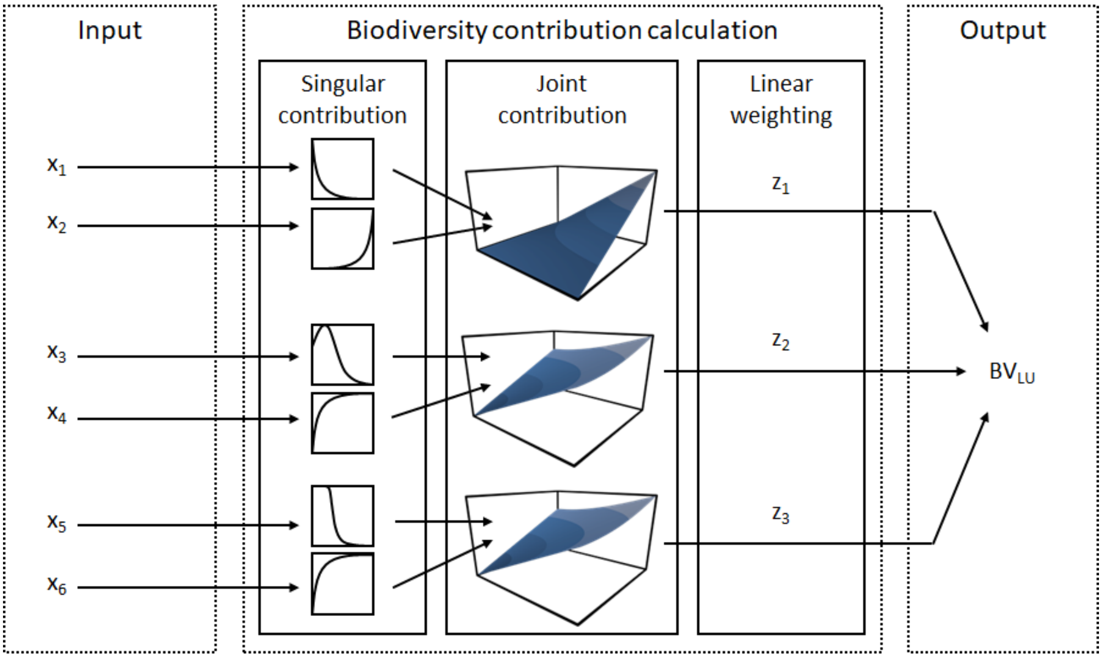

Figure 1: Schematic illustration of the calculation of a specific BV_{LU}

[3] According to Fehrenbach et al. (2015) and with adjustments according to Lindner et al. (2020). Here, hemeroby level 2 means a great closeness to nature, whereas hemeroby level 7 describes the maximum remoteness from nature.