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Quantifying the Non-Use Value of Biodiversity in Cost–Benefit Analysis: The Dutch Biodiversity Points

Abstract: Biodiversity points are a quantitative measure for biodiversity. For over a decade, biodiversity points are being applied in the Netherlands for measuring the impact of roads, enclosure dams, and other water management projects on the non-use value of biodiversity. Biodiversity points are quite similar to the quality-adjusted life years used for cost-effectiveness analysis of healthcare treatments. Biodiversity points can be calculated by multiplying the size of the ecotope (e.g., number of hectare), the ecological quality of the ecotope (0–100 %), and the ecological scarcity of each type of ecotope. For many infrastructure projects, the impact on the non-use value of biodiversity can be a principal purpose or a major co-benefit or trade-off, for example, for a park, a fish sluice, a road, an ecoduct, an enclosure dam, or a marine protected area. Biodiversity points are a simple, transparent, and standardized way to aggregate and quantify the qualitative or ordinal assessments by ecological experts. For measuring the non-use value of biodiversity, they are also more informative than valuation by revealed or stated preferences methods. This paper provides the first overview of the application of this method in the Dutch practice of cost–benefit analysis. It also discusses its merits and limitations. The calculation and use of biodiversity points are illustrated by four case studies.

Keywords: biodiversity; cost–benefit analysis; ecological points method; ecosystem services; habitat equivalence analysis; infrastructure; natural capital; threat-weighted ecological quality area method; water management.

JEL classifications: H43; H54; Q51; Q57

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1. Introduction[1]

Ecosystem services are the services of ecosystems, like forests, soil, water, and air. Three types of ecosystem services are commonly distinguished (OECD, 2018, Chapter 13):

(i) Provisioning services, like food, timber, and drinking water.
(ii) Regulation and maintenance services, like fertility of the soil and water safety provided by dikes.
(iii) Cultural services, like recreation services and the symbolic value of very old trees and whales.

Ecosystem services contribute only to human welfare when they are final and are not intermediate. Provisioning services and cultural services like food and recreation services are final ecosystem services as they contribute directly to human welfare. Regulatory services, like fertility of the soil, are usually intermediate services as they often contribute only indirectly to human welfare. However, water safety provided by dikes is a final service contributing directly to human welfare.

Biodiversity refers to the stock of natural capital in terms of the variation, size, and quality of species, populations, and ecosystems. Biodiversity can have a direct impact on human welfare by providing, for example, a nice environment for walking in a forest. However, it can also have an indirect impact on human welfare by providing a good-quality ecosystem essential for providing future ecosystem services. Biodiversity may also be relevant for human welfare without a clear link to ecosystem services, for example, from an options perspective, an inheritance perspective, or an existence and symbolic perspective. When the link between the quantity and quality of biodiversity and ecosystem services is absent or not very clear, the value of biodiversity could be labeled as the non-use value of biodiversity (OECD, 2018, chapter 13).

For many cost–benefit analyses (CBAs), properly assessing the welfare effects of a policy measure on ecosystem services and biodiversity is important. This does not only apply to CBAs on nature policy but also to those on other policy areas such as mobility, agriculture, and water safety, as the policy measures in these policy areas often have impacts on ecosystem services and biodiversity. For example, a new road connecting two cities through a forest is good for mobility but has also impact on the

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value of ecosystem services and biodiversity. The use value of the forest may increase due to the reduction in travel time for visitors. But the non-use value, for example, the inheritance value of the forest and the biodiversity of its species may be affected severely by fragmenting the forest and by increasing traffic, pollution, and visitors.

Various strands of literature have addressed the question of how to incorporate the effects of a policy measure on ecosystem services and biodiversity in CBAs. However, in the scientific literature on CBA, CBA guidelines and textbooks and CBA practice, there is no general agreement on how to define and assess the welfare impacts due to changes in ecosystem services and biodiversity (Boyd *et al.*, 2014; Schaefer *et al.*, 2015).

In the scientific literature, a large number of studies provide monetary values of the welfare effects of changes in ecosystem services and biodiversity (see e.g., TEEB, 2010; Markandya, 2016) or discuss methodologies that can be used in CBA (see, e.g., Atkinson & Mourato, 2008; Freeman *et al.*, 2014; Johnston *et al.*, 2017). In this literature, there is a clear bias toward studies focusing on novel estimation methods without considering their practical applicability (Laurans *et al.*, 2013). Effects on ecosystem services and biodiversity in these studies are described in general terms, without specifying the impact of the policy measure, for example, on the number of visitors to a park, the appreciation of an area, the health effects of replacing a forest by a road, or effects on land productivity. As a consequence, they often yield unreliable estimates in monetary terms or estimates that cannot be used in a CBA for analyzing specific policy measures (Bartkowski *et al.*, 2015). Furthermore, by focusing on the valuation of the use of various specific ecosystem services, like recreational services, they generally ignore the non-use value of biodiversity.

General CBA guidelines,[2] general CBA textbooks (e.g., Boardman *et al.*, 2006; Mishan & Quah, 2007), and guidelines on CBA and the environment (e.g., OECD, 2006, 2018; USEPA, 2014) only pay limited attention to ecosystem services and biodiversity and do not specify how changes in natural capital affect welfare and how they can be measured. Due to this lack of clear guidelines, CBA practitioners employ varying definitions of what “natural capital” is and how changes in natural capital affect welfare. This also causes problems in defining what to value.

The studies on the Economics of Ecosystems and Biodiversity (TEEB, 2010, 2013) take stock of existing national and international initiatives and country practice on measuring ecosystems and biodiversity. They have advanced substantially the literature in this field. In contrast to the scientific literature, they are more frank about the costs and limitations of various valuation methods. For example, contingent

2 See for example, World Bank (2010), the Asian Development Bank (2013), the USA Benefit–Cost Centre (Zerbe *et al.*, 2010), the UK Greenbook (HM Treasury, 2011), and the EU CBA guidelines for investment in infrastructure financed by the cohesion funds (European Commission, 2014).

valuation and choice modeling are described as "Expensive and technically difficult to implement. Prone to biases in design and analysis" (TEEB, 2013, p. 66). However, the TEEB studies do not provide clear guidelines on which ecosystems to include, how to include biodiversity, how to deal with substitutability, and how to prevent double counting. Furthermore, it is explicitly stated that the purpose is to provide guidance for environmental accounting at the national or regional level and that the purpose is not to provide guidance for CBA, i.e., analysis of a project or specific (set of) policy measures (TEEB, 2013, p. 48).

The new OECD Handbook on CBA and the environment (OECD, 2018) devotes one chapter to the measurement of ecosystem services and biodiversity. It is clear about the limitations of measuring the non-use value of biodiversity by looking at ecosystem services and stated preference methods, for example:

(i) "the emphasis on ecosystem services says little about the value of biodiversity defined by the Convention on Biodiversity … typical ecosystem services at best only implicitly reflect the contribution of this biodiversity (the contribution of the richness, complexity and resilience of species and the ecosystems that they inhabit) if at all." (OECD, 2018, p. 318)

(ii) "Therefore, while stated preferences may provide sound valuation for high-experience, use value goods, the further one moves to consider indirect use and pure non-use values, the more likely one is to encounter problems." (OECD, 2018, p. 317)

(iii) "preferences need not always conform to what is ecologically feasible or sustainable. Thus, in the Morse-Jones *et al.* study, respondents had a massively stronger preference for iconic, 'charismatic' animals which outweigh concerns regarding ecologically crucial issues such as extinction threat. So, for example, willingness to pay to conserve lions, even where these animals are not threatened by extinction, hugely outweighs stated values for say a species of frog, even when it is on the brink of extinction." (OECD, 2018, p. 325)

The Netherlands have a long tradition in CBA.[3] The way ecosystem services and biodiversity have been incorporated in Dutch CBAs has changed drastically over time: from CBAs in which major impacts on ecosystem services and biodiversity were not even mentioned to CBAs in which the impact on ecosystem services are valued as much as possible and effects on the non-use value of biodiversity are measured by biodiversity points. These biodiversity points measure the impact of a

3 For more than a century, CBA is used to support decision-making on public investments in the Netherlands, see Bos and Zwaneveld (2017) and Bos (2008).

policy measure on the amount and the quality of biodiversity in a standardized way. It takes into account the area of ecosystems affected, the ecological quality of each area, and a weight factor per type of ecosystem reflecting the contribution of the ecosystem to species richness and the threat level to that ecosystem. Unlike the qualitative, nominal or ordinal, assessments by ecological experts in environmental impact assessments (EIAs), biodiversity points are a quantitative measure. As a consequence, biodiversity points can be used for cost-effectiveness and also for assessing the net benefits of a project with biodiversity as a co-benefit or trade-off.

The biodiversity points method was developed in 2009 at PBL Netherlands Environmental Assessment Agency, the official expert institute of the Dutch government on the environment (Sijtsma *et al.*, 2009; precursors are Strijker *et al.*, 2000; Sijtsma, 2006); it is sometimes also referred to as ecological points method or the Threat-weighted Ecological Quality Area Method (T-EQA). It could be regarded as a specific way of habitat equivalence analysis, which is commonly used for natural resource damage assessment or biodiversity loss compensation (Roach & Wade, 2006; Ray, 2008; Wende *et al.*, 2018). In the Netherlands, the biodiversity point method has been applied to many different case studies by many different researchers and consultancy firms (see Sections 4 and 5). According to the Dutch guidelines on CBA (Romijn & Renes, 2013; Klooster *et al.*, 2018), biodiversity points are an innovative and practical method to measure the impact on the non-use value of biodiversity. The biodiversity points method and its application to various Dutch case studies has been published in international journals and books on ecology and land use (Strijker *et al.*, 2000; Sijtsma *et al.*, 2011, 2013, 2020; van Puijenbroek *et al.*, 2015). European legislation on protection of birds and their habitats and of marine waters requires impact analysis and CBA of the policy measures. Estimating the benefits is commonly regarded as the major problem in conducting such analysis. Recently, in European networks of policy-makers and researchers on the socioeconomic analysis of such policy measures, the biodiversity points method has been put forward as one of the solutions to this (Liefveld *et al.*, 2011; van Oostenbrugge *et al.*, 2015; Spaans, 2020).

This paper contributes in two ways to the existing literature on CBA, ecosystem services and biodiversity. Firstly, international guidelines on CBA and environmental services (e.g., OECD, 2006; EPA Guidelines, 2014) focus on discussing valuation issues, usually in rather general terms and with a strong focus on the analytical methodologies and with very limited attention to CBA practice and the link to actual policy decision-making. This paper stresses that to support public decision-making, it may be more informative to put an equally strong focus on defining which effects on the quality and quantity of ecosystem services and biodiversity to include in CBA and how to ensure that only the effects relevant for welfare are included.

Secondly, this paper provides an overview of methods applied in Dutch CBAs to measure the volume and quality of ecosystem services and biodiversity, with a focus on the biodiversity point method. This paper provides the first overview of the history and application of this method in the Dutch CBA practice during the past decade. It also discusses its merits and limitations.

The outline of this paper is as follows. Section 2 starts with a general overview on CBA, ecosystem services, and biodiversity ("nature") in the Netherlands. In the subsequent sections, six case studies of CBA on water management and transport infrastructure are discussed. Two case studies in 2000 and 2005 on "Room for water" and "Room for the river" compare various alternatives on their cost-effectiveness for nature. They are the topic of Section 3. These case studies illustrate how the impact on ecosystem services and biodiversity was measured in cost-effectiveness analysis (CEA) *before* the introduction of the biodiversity point method. The method of biodiversity points and its application in four different case studies are discussed in Section 4. Conclusions about the merits and limitations of biodiversity points in CEA and in a full-fledged CBA are drawn in Section 5.

2. General overview on CBA and nature in the Netherlands

2.1. Early CBAs on flood risk ignoring major negative effects on nature

Up to the 1970s, CBAs in the Netherlands mainly pertained to major investments in flood risk management. Examples include the 30 km long Southern Sea enclosure dam (Afsluitdijk) and the Delta Works, each of which costed about 6–7 % of GDP. The impact on nature was usually not included in these CBA's.

The 1901 CBA on the Southern Sea enclosure dam (Lely, 1901) looked at many different types of costs and benefits but ignored the negative impact on water quality and fish. Similarly, following the massive flooding of the southwestern part of the Netherlands in 1953, the CBA on the Delta Works (Tinbergen, 1953) compared two alternatives to ensure sufficient flood risk safety: raising and strengthening dikes all along the waterways versus shortening the coastline by blocking the estuary mouths with barrier dams (Delta Works). Many different types of costs and benefits were monetized, quantified, or at least mentioned. But closing off the estuary mouths by barrier dams would turn tidal salt water areas into fresh water lakes like the IJsselmeer (the former Southern Sea); these substantial negative effects on nature were ignored in the CBA. Decades later, following increasing international concerns about the

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environment, in Dutch public debate, these negative effects on nature were stressed. As a consequence, in 1976, the Dutch government decided to construct semi-closed barrier dams. This was an entirely new option for flood risk management (only in case of serious danger of flood risk, the barrier dams are closed). It was much more expensive than closed barrier dams but the advantage was that negative effects on nature were limited.

2.2. Environmental impact analysis (EIA) as an input for CBA

Reporting environmental effects of public investments, including those on nature, started in 1978. NEI and RIN (1978) presented an EIA and a CBA comparing the extension of the port of Den Helder with alternative solutions in the ports of IJmuiden and Rotterdam. Basically, most effects on nature were given in physical terms that were presented next to monetary costs and benefits. Some effects on nature were monetized, in particular, the foregone revenues of fishing, the loss of shell lime production, and the loss of water cleaning capacity. The impact of a new port on five basic functions of nature (production function, intermediary and supporting function, informative function, regulatory function, and conservation function) was specified and at least scored qualitatively (with minus and plus signs) for 13 different sub-functions. Nearly a decade later, a European Act[4] made EIAs obligatory.

Unlike a CBA, EIA does not translate positive and negative effects in nature into monetary terms and usually do not consider double counting, that is, whether several environmental impacts lead to the same impact on welfare. But their information can be used as an input for CBA. For example, in the CBA on deepening the Westerschelde waterway from the Netherlands to Antwerp (Saitua, 2004), the EIA was used to claim that from a European perspective, the environmental effects were negligible. This is still the role of EIA in most Dutch CBAs on transport infrastructure and spatial projects (see Annema & Koopmans, 2015).

2.3. National guidelines on CBA and nature

To improve the quality and consistency of CBA, national CBA guidelines for transport infrastructure were developed (Eijgenraam *et al.*, 2000). It does not explicitly discuss the effects on nature of infrastructure projects. The valuation of effects on nature in CBA was separately addressed in a supplementary guidance

4 Act on environmental impact assessment, Directive 85/337 EEC.

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(Ruijgrok *et al.*, 2004) and an overview with key figures for such valuation of nature (Witteveen & Bos, 2006).

As a result, attempts were made to include, for example, effects of changes in nature on housing prices, health, and recreation. Due to shortage of data and a lack of primary valuation studies, this resulted in many cases in arbitrary assumptions or token entries – indicating that the effect was relevant but that no reliable monetary value could be estimated. In other CBAs, some impacts were double counted, or other errors were made in quantifying and valuing the welfare effects in terms of cost and benefits.

In 2013, an updated CBA guideline was published by CPB and PBL (Romijn & Renes, 2013), which included a brief discussion of accounting for the impact on nature.[5] This topic is covered more in-depth in the supplementary guideline on CBA and nature (Klooster *et al.*, 2018). The guideline on CBA and nature recommends the use of biodiversity points for measuring the impact on biodiversity. It also stresses the importance of providing clarity about the welfare effects of changes in nature. For example, most regulating services, like natural pest control and water purification are intermediate services. They indirectly affect welfare as they are an input in the production function for final ecosystem services. Effects on these intermediate services are to be indirectly included in a CBA through their effects on the final services and the valuation thereof (see Boyd & Banzhaf, 2007). Biodiversity – the variety of genes, species, and ecosystems – holds a special position in these guidelines. Biodiversity is important to guarantee a continued delivery of ecosystem services over the long term and for maintaining ecosystem resilience (Cardinale *et al.*, 2012; Isbell *et al.*, 2017).

3. Two CBAs without biodiversity points

In 1993 and 1995, the water levels in the major Dutch rivers Rhine, Meuse, Waal, and IJssel rose to such levels that serious breaching of river dikes was only just avoided. The projects “Room for Water” and “Room for the River” meant a new flavor on the Dutch menu of preventive water management policies: spatial adjustments to increase safety. Two evaluation studies of these projects characterize how nature was included in CEA in the years before biodiversity points were introduced.

5 Recently, also the discount rates to be used in Dutch CBAs have been changed. In 2016, it was decided to reduce the official discount rate from 5.5 to 3 %. For nature, an annual relative price increase of 1 % is prescribed (Werkgroep Discontovoet, 2015; Koetse *et al.*, 2018). As a consequence, the net effect for nature is a discount rate of 2 %.

Table 1 A CEA of four projects in the lower river region for increasing safety (Stolwijk & Verrips, 2000, table 8, p. 55).

	Rerouting of Meuse	Rerouting of Waal	Changing discharge distribution	Raising dikes
Net annual monetary benefits (bln euro)	0.88	0.90	0.93	<0.93
Non-monetary effects				
Quality of environment	+	+	++	
Spatial beauty	+	+	++	–
Social consequences for farmers	–	–	–	
Flexible water management	+	+	++	–

3.1. CEA comparing four strategies for improving safety along the rivers and coast

The first study refers to a quick-scan CEA[6] of a set of projects for improving safety along the rivers and coast (Stolwijk & Verrips, 2000). Table 1 provides an overview of the monetary costs and benefits of water safety for four project alternatives.

Next to monetized cost and safety benefits, four different types of non-monetary effects were taken into account: quality of landscape spatial beauty, quality of environment (including biodiversity), social consequences for farmers, and flexible water management. The scores in terms of a simple ordinal scale (5-point Likert scale) were assessed by experts. Insufficient information was available to estimate the monetary value of these non-monetary effects.

The approach provided some evidence that the spatial adjustments proposed in the alternatives (rerouting of the river Meuse, rerouting of the river Waal, and changing the discharge distribution over various rivers) had more positive environmental effects than the conventional approach of raising dikes; in particular,

6 A CEA is a CBA in which a specific purpose, like a target for extra transport capacity on a road, a water safety norm, an emission goal or a nature policy standard. It is defined in advance and only the net benefits of policy proposals that meet this purpose are compared. The distinctive feature of CEA is that the costs and benefits of meeting the specific purpose are ignored.However, if one of the policy alternatives scores better than the target for the specific purpose, these extra benefits of this policy alternative should also be included in the CEA. Furthermore, in case the net benefits of meeting a specific targeted purpose may be clearly negative, it is often wise not to take this purpose for granted and to conducting a full CBA, i.e. including the costs and benefits of meeting that specific purpose.

changing the discharge distribution would have positive effects on the environment according to the experts. However, the scores by the experts were not transparent, for example, it was not clear which criteria they had used to assess the change in the quality of environment. As a consequence, these expert scores were not very informative for redesigning the plans to better mitigate negative impacts or exploit potentials for co-benefits.

3.2. Four-dimensional CEA

The second study refers to an assessment of 600 specific policy measures and 4 packages of policy measures as part of the project Room for the River by Ebregt *et al.* (2005). These policy measures were very heterogeneous, ranging from deepening trenches, moving dikes further away from the river, introducing extra channels, rerouting rivers, and raising dikes. They had a broad range of effects, many of which could not be well translated into monetary terms. For that reason, a so-called four-dimensional CEA was developed. For all investigated measures, four types of benefits were distinguished:

(i) Extra landscape with high environmental quality (biodiversity) in hectares.
(ii) Extra landscape with spatial beauty per kilometer along the river.
(iii) Extra landscape attractive for leisure activities per kilometer along the river.
(iv) Extra safety in terms of a reduction of high water level to the target high water level in m^2.

For the three different types of landscape effects, the impact of the policy measures was estimated as the change in acreage of a wide range of ecotopes. This was used to create an expert opinion on biodiversity, spatial beauty, and leisure effects. The measure for biodiversity can be seen as a precursor of the biodiversity points method discussed in Section 4. Similar types of ecotopes are distinguished as in the biodiversity points approach but here weighting was done through expert review instead of through a predefined set of quality and weighting indices. The predefined weighting indices make the biodiversity points more transparent and objective.

For all 600 measures, information on their costs per unit of benefit was compared to the standard cost per benefit. The typical cost rate for an extra hectare of landscape with high environmental quality was 230,000 euros. This was much higher than the average costs per ha for increasing environmental quality in the Netherlands. Hence, combining extra safety with environmental development did not seem to be very cost-effective.

4. Four CBAs with biodiversity points

The unsatisfactory and arbitrary way of incorporating biodiversity in CBAs encouraged Sijtsma *et al.* (2009) to develop an alternative approach; this was a joint effort by ecologists and economists. They proposed a CEA in which the net benefits that can be valued are compared with the change in so-called *biodiversity points*. The biodiversity point method, sometimes also referred to as the T-EQA method (van Puijenbroek *et al.*, 2015) or the ecological points method (Liefveld *et al.*, 2011), measures the impact on the amount and quality of biodiversity in a standardized way (Sijtsma *et al.*, 2009; Jaspers *et al.*, 2016).

This biodiversity points approach is increasingly being applied in Dutch CBAs, resulting in more and more insight in what would be reasonable costs for obtaining an additional biodiversity point. This is in particular useful to compare the cost-effectiveness of project alternatives with respect to their impact on biodiversity. However, biodiversity points are also useful for assessing the net benefits of projects in which the impact on biodiversity is a major co-benefit or trade-off. In Dutch CBA practice, the impact on biodiversity is still often included as a token entry with some qualitative assessment. As a consequence, major impacts on biodiversity may be ignored in public decision-making. It also happens that the impacts on biodiversity are very limited but are exaggerated in public debate about the project for strategic reasons. Biodiversity points can be an important method to avoid such misunderstanding and misleading communication about the size of the impact on biodiversity.

4.1. How to measure biodiversity points?

The biodiversity points are calculated by multiplying three components:

(i) The area of natural or semi-natural ecosystems affected (in hectares or square kilometers);
(ii) The ecological quality of each area (0–100 %);
(iii) A weight factor per type of ecosystem, reflecting the contribution of the ecosystem to species richness at national, European, or global level, which depends on the species present in the ecosystem and their threat level.

The ecological quality is measured by an intactness or robustness score, in a range from 0 to 1. This measure is determined for each of the relevant ecotopes based on the number of characteristic species present in the area relative to their presence in an intact ecosystem. These ecotopes and characteristic species are derived from the universal set of biodiversity indicators (Convention on Biological Diversity (CBD), 2004), the

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more detailed European set of biodiversity indicators (European Environment Agency, 2012) and the mean species abundance used in UNEP's Global Environment Outlook. For this, national reference lists containing the species in an intact ecosystem are available, for example, for the Netherlands.[7] In Europe, for the Habitat Regulation (European Commission, 1992) and the Water Directive (European Commission, 2000), each EU-member country had to assess the ecological quality of ecotopes in comparison to a healthy ecological condition. So, this is already roughly similar to providing intactness or robustness scores for ecological quality.

EIAs generally provide a lot of information on ecological quality, before and after the policy intervention. This information should first be extended and translated into ecotypes and per ecotype, the number of characteristic species in the area. This can then be translated into ecological quality scores, before and after the policy intervention. Multiplying the ecological quality scores for the different ecotopes by the acreage of their area gives the Ecological Quality Area score (EQA) per ecotope.

Finally, the EQAs of the ecotopes are multiplied with standardized weight factors that indicate the threat level to the ecotope. This threat level is related to the relative number of red list species in the ecotope. Extremely threatened ecotopes have the highest weight, while commonly occurring ecotopes with common and not threatened species have the lowest weight. As a result, an intervention in a highly threatened ecotope results in a higher score than a similar intervention in a non-threatened ecotope. For example, salt marshes have a weighting factor of 2.4, nutrient-poor peatlands and moist heather lands have a weighting factor of 1.2, and agricultural grasslands have a weighting factor of 0.4.

Determining the weighting factors is not fully straightforward, and different methods and data sources are possible (see Sijtsma *et al.*, 2009). However, most important is that these weights are standardized for each country and based on systematic ecologic data collection which is objective and transparent. This approach is similar to how CO_2-equivalents are used to aggregate different types of emissions or how different health effects are summarized by the indicator Disability Adjusted Life Years.

Biodiversity points or T-EQA is defined as:

$$\text{Biodiversity points} = \sum\nolimits_{i=1}^{n} \text{Area}_i \times \text{Quality}_i \times \text{Weight factor}_i$$

with $i \in \{1, \ldots, n\}$, the different types of ecosystem types or nature types.

7 Reference lists of "species pursued" have been prepared for monitoring the Dutch nature policies and contain the pursued biotic and abiotic characteristics of each nature type. Using the reference lists, for each nature area measurable objectives can be set and monitored. They provide the basis for conservation planning and management and national and European nature policies.

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This implies that in order to calculate the impact of a policy measure in terms of biodiversity points, 10 steps are necessary:

(i) Select the relevant ecosystem types in the impacted area.
(ii) Assess the size of the area of each ecosystem type in the current situation (baseline).
(iii) Assess the characteristic species for each ecosystem type.
(iv) Assess the threat level or relative weight of each ecosystem type; the relative number of red list species within this ecosystem might be used as a first proxy.
(v) Calculate the local intactness of these ecosystems based on the presence of characteristic species relative to the number that would be present in an intact ecosystem. This gives a % score ranging generally from 0 to 100 %. Rescale this ecological quality from 0 to 1.
(vi) Calculate the number of biodiversity points for the current situation using the information from steps 1 to 5.
(vii) Specify the acreage of the changes in ecosystem types in the project alternatives.
(viii) Assess the ecological quality per ecosystem type in the project alternatives.
(ix) Calculate the number of biodiversity points in the project alternatives using information from steps 4, 7, and 8.
(x) Compare the number of biodiversity points in the project alternatives with those in the current situation.

An advantage of the biodiversity point method is that decision-makers have a single objective measure to compare biodiversity effects of alternative interventions. For some questions, this is more useful than the range of impacts shown by an EIA. Where an EIA is useful to assess whether legal norms are exceeded, it is not very useful to compare, for example, an intervention impacting fish stocks with an alternative intervention impacting water quality in an adjacent area. The scarcity-based weighting of the biodiversity points allows decision-makers to compare these incomparable impacts.

Four examples can illustrate the use of the biodiversity point method in Dutch CBA practice.

4.2. CBA of increasing biodiversity by raising groundwater levels

For three areas in the peatlands in the Netherlands, two alternatives for raising the groundwater levels are compared in a CBA (Witteveen *et al.*, 2006). Both alternatives

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Table 2 Results from a full CBA and a CBA presenting biodiversity points separately.

Area	Alternative	Willingness to pay for biodiversity (mln euro)	Biodiversity points
Krimpenerwaard	Alt.1	0.5	1352
	Alt.2	0.5	1751
Groot Wilnis Vinkeveen	Alt.1	1.2	808
	Alt.2	1.2	1730
Wormer and Jisperveld	Alt.1	0.5	976
	Alt.2	0.5	1691

improve biodiversity. The second alternative leads to the highest levels of aquatic biodiversity. In this CBA, the effects on biodiversity were monetized on the basis of willingness to pay. The scientific basis for the willingness-to-pay values, however, was rather weak.[8] The study did not properly present the biodiversity change to the respondents and was not clear about the population impacted by the change proposed. This is a well-known risk of poorly designed stated preference research.

Sijtsma *et al.* (2009) recalculated the results but now with biodiversity points. Table 2 presents the results of this study, with the biodiversity effects given both in monetized values as well as in biodiversity points. Monetized values hardly differ between the two alternatives, whereas the biodiversity points clearly show that the alternatives have different biodiversity impacts. Next to that, the area having the largest monetized value does not result in the highest gain in biodiversity points.

The revised information provides more relevant information to policy-makers than the original information in the badly designed stated preference research. It enables them to distinguish between the effects on nature of both alternatives. Moreover, it allows them to evaluate for which alternative and in which area they obtain the highest extra biodiversity per euro invested. This information in biodiversity points may be supplemented with a more qualitative description of the impacts on nature of the various alternatives.

4.3. CBA extra lanes for the highways near Amsterdam

Extra lanes for the highways near Amsterdam is currently the largest infrastructure project in the Netherlands. The investment is about 4 bln euros. These lanes are extended near a nature reserve (Naardermeer). As a consequence, there are negative

8 The willingness to pay was calculated as 5 euro per household per year times the local impact population. The 5 euro was an arbitrary value with no clear link to the three major previous studies on peatland in the Netherlands; information from the latter studies was also not applicable, as the project and situation were quite different.

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Table 3 Extra lanes for the highway Schiphol–Amsterdam–Almere on the nature reserve Naardermeer, impact in terms of biodiversity points for four project alternatives.

	Location specific	**Streamline 4×2 lanes**	**Streamline 2×4 lanes**	**Most environmental friendly alternative**
Destruction of habitat	−92	−83	−102	−106
Less disturbance of animals	85	98	90	146
Net impact in comparison to the baseline project	−7	15	−12	40
PM. Baseline project in comparison to the current situation: −149				

effects on this nature reserve. In the official CBA on this project, the impact on biodiversity was only mentioned briefly and qualitatively.

Sijtsma *et al.* (2009)[9] calculated the impact in terms of biodiversity points for this project and four nature-friendly project alternatives (see Table 3). This reveals that:

(i) The impact of the baseline project of extra lanes compared to the current situation is a reduction of 149 biodiversity points (last line of Table 3).

(ii) In comparison to the baseline project, the four supplementary project alternatives lead to less disturbance of animals but also to a destruction of habitat. As a consequence, in terms of biodiversity points, only two of these alternatives have a net positive impact (streamline 4×23 lanes and most environmental friendly alternative). And these net impacts (15 and 40 biodiversity points) are not sufficient to overcome the negative impact in the baseline project (−149 biodiversity points). The other two project alternatives have small net negative impact (−7 and −12 biodiversity points).

(iii) The net impact on biodiversity of this project and its project alternatives is quite small (e.g., effect of the baseline project is −149 biodiversity points). It is also small compared to the impact of the peatland projects in the example above, ranging from 800 to about 1800 biodiversity points. This reflects that the peatland project is very effective in limiting the rapid deterioration of peatland due to drought, while the extra lanes have a much less and more indirect effect on the unique flora and fauna nearby.

4.4. CEA for the reconstruction of the Afsluitdijk

The first major official study using the biodiversity-points methodology for public decision-making was the CEA of the Afsluitdijk by Grevers and Zwaneveld (2011).

9 See also Sijtsma *et al.* (2011) and Sijtsma *et al.* (2013).

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In order to meet legal safety standards of flooding once every 1/10,000 years, this enclosure dam needed fundamental reconstruction. The dam should also continue to meet two other functions: managing the water level in the IJsselmeer and providing good connections for transport by car and by ship. This renovation could be combined with new functions with respect to nature, for example, dikes combined with trees (green dikes) and special sluices for fish.

The CEA showed the effects on nature in two different ways: the extent to which legal environmental protection standards were met and the score in biodiversity points. In contrast to the perspective of minimal legal standards for the environment, the score in biodiversity points does not only look at negative effects on the environment but also takes into account how much extra biodiversity can be created.

In order to calculate the biodiversity points for the plans to renovate the Afsluitdijk, the impact area and the different habitats had to be distinguished and the quality and relative weight of each habitat had to be assessed. The impact area considered was 3 km on both sides of the 33 km long Afsluitdijk. Table 4 provides an overview of

Table 4 Habitats surrounding the Afsluitdijk (Wessels *et al.*, 2011).

Type of habitat	Relative weight	Quality of current situation (percentage)
I. IJsselmeer and Afsluitdijk		
Landzone		
Roadside grass	0.4	13
Makkumer Noorwaard	1.8	55
Paved road surface	0.0	
Shoreline and marshes		
Makkumer Noorwaard	1.6	54
Brackish	2.4	
Brackish and sweet–salt gradient	3.4	
Open water		
Shallow and sweet	1.3	35
Shallow and brackish	2.0	
Shallow, brackish, and sweet–salt gradient	3.0	
Deep and sweet	0.7	34
II. Waddensea (north side of Afsluitdijk)		
Saltmarsh including pioneer and climax stages and sweet–salt gradient	3.4	
Saltmarsh including pioneer and climax stages	2.4	
Dry falling sand plates including mussel banks	2.0	52
Permanently flooded sand plates including mussel banks	2.5	40
Gullies	0.7	37
Land zone, roadside grass	0.4	

Table 5 Renovating the enclosure dam: cost-effectiveness of various options for extra biodiversity (see Wessels *et al.*, 2011).

Alternatives	**Biodiversity points**	**Costs (mln euro)**	**Of which: costs for biodiversity (mln euro)**	**Cost-effectiveness (mln euro per biodiversity point)**
Current situation	11,770			
Major alternatives		Difference with the current situation		
2100-Robust	−30	1640		
Basic alternative	−10	1390		
Monument in balance	0	1560		
Natural enclosure dam (Green Afsluitdijk)	1600	2670	550	0.34
Waddenworks	−330	1630		
Supplementary options				
500 ha Marshes (option for Waddenworks)	3600	135	135	0.04
Brackish water zone (option for Natural enclosure dam)	1330	240	240	0.18
Fish sluice (option for all major alternatives)	1500	10	10	0.01

the different habitats in the area, their weight scores and their current, pre-intervention quality scores. The quality scores were based on earlier estimates for the European Water Framework Directive.

The table shows that the ecological quality varies from zero for paved surface to 3.4 for areas with a sweet–salt water gradient. The current situation is 11,770 biodiversity points, for an area of 19,000 ha. The average ecological quality is 37.5 % and the average weighting factor is 1.6.

This table shows that the current quality of the shoreline and marshes in the Makkumer Noorwaard is 54 per cent. Its weighting factor is 1.6. With an area of 300 ha, these marshes have $300 \times 1.6 \times 0.54 = 259$ biodiversity points. Suppose that you build an additional 100 hectares of marshes at the expense of shallow sweet open water, then you gain $100 \times 1.6 \times 0.54 = 86.4$ biodiversity points but lose $100 \times 1.3 \times 0.35 = 45.5$ biodiversity points.

According to Table 5, only some alternative interventions and options have substantial impact on nature, for example, the natural enclosure dam, the 500 ha extra marshes, and the fish sluice. They either result in larger areas of rare habitat types (with high weighting scores) or result in substantial quality improvements. The table also shows that the option Green Afsluitdijk has a clear positive effect on biodiversity: an increase of 1600 biodiversity points. An interesting result was that

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nearly the same amount of biodiversity points (1500) could be obtained by constructing a fish sluice in the Afsluitdijk but at only a fraction of the costs: not 550 mln euro but 10 mln euro. Hence, fish sluices were much more cost-effective for improving biodiversity.

This CEA was well received by policy-makers. The results were almost completely adopted in the final decision of the Dutch Cabinet (Zwaneveld *et al.*, 2012). The option Green enclosure dam was rejected, and it was decided to construct a fish sluice. Subsequent political decision-making led to a much more advanced and fish-friendly but also much more expensive fish sluice (35 mln euro).

4.5. CEA meta-study on infrastructure defragmentation

On request of the Dutch government, recently a meta-study was conducted about the cost-effectiveness of 175 defragmentation policy measures in the period 2004–2018 (see Table 6). Four types of defragmentation alternatives were distinguished: ecoduct, viaduct, big faunatunnel, and small faunatunnel. The major conclusions are:

(i) Large faunatunnels and viaducts with share use of traffic and animals are more cost-effective for stimulating biodiversity than ecoducts (0.08 mln euro per biodiversity point versus 0.18 mln euro per biodiversity point); small faunatunnels are by far the least cost-effective, i.e., on average more than double as costly than ecoducts (0.38 mln euro per biodiversity point).
(ii) The cost-effectiveness differs between ecoducts: the more nature areas are in the direct vicinity, the more cost-effective.
(iii) Buying agricultural land and using this for nature purposes is about as cost-effective for biodiversity than a viaduct or big faunatunnel. However, the ecological improvement of existing nature zones is even much more cost-effective (0.02 mln euro per biodiversity point).

The results of this meta-study can also be compared with those of the other biodiversity studies. In terms of impact on biodiversity, one ecological connection results in somewhat more than 10 biodiversity points (146 connections, in total 1791 biodiversity points). This implies that, for example, the fish sluice in the enclosure dam with 1500 biodiversity points is broadly comparable to 150 ecological connections. In terms of cost-effectiveness, a fish sluice in the enclosure dam (0.02 mln euro per biodiversity point) is comparable to ecological improvement of existing areas and much more cost-effective than ecological connections.

Table 6 CEA meta-study on infrastructure defragmentation (Sijtsma *et al.*, 2018, 2020).

Defragmentation alternatives	Number of connections	Biodiversity points	Costs (mln euro)	Cost-effectiveness (mln euro per biodiversity point)
1. Ecoduct	26	1074	194	0.18
2. Viaduct	20	195	16	0.08
3. Big faunatunnel	44	427	33	0.08
4. Small faunatunnel	56	95	36	0.38
Total	146	1791	279	0.16
Expanding nature areas (less agriculture)		41,000	3080	0.08
Ecological improvement of existing areas		58,413	1370	0.02

5. Conclusions and looking forward

The way nature has been incorporated in Dutch CBAs has changed over time: from CBAs in which major impacts on nature were not even mentioned to CBAs in which the impact on ecosystem services are valued as much as possible and effects on biodiversity are measured in biodiversity points. In some earlier CBAs, like Stolwijk and Verrips (2000) and Ebregt *et al.* (2005), in particular, for measuring the non-use value of biodiversity, ordinal scaling or quantitative measures were used, like the change in the number of hectares of high-environmental quality. But no detail was shown, and the rarity of the species in the habitat was not taken into account.

In a whole range of recent Dutch studies on CBA and measuring the impact of projects on nature, biodiversity effects are quantified in terms of "biodiversity points," and the method has been further standardized for terrestrial and aquatic nature (salt, sweet):

(i) the revisited CBA on raising groundwater levels in order to protect peatlands (Sijtsma *et al.*, 2009);
(ii) the revisited CBA on extra lanes for the highway Schiphol–Amsterdam–Almere (Sijtsma *et al.*, 2009);
(iii) the CEA on renovating the enclosure dam of the Lake IJssel (Grevers & Zwaneveld, 2011);
(iv) the report on evaluating biodiversity of the North Sea using eco-points, in order to test its applicability for European Marine Strategy Framework Directive assessments (Liefveld *et al.*, 2011);
(v) the CEA on the management of the water level in the Lake IJssel in view of the rising sea level (Bos *et al.*, 2012);

(vi) the report on evaluating projects in the Delta program for water management with six new case studies on biodiversity points (van Gaalen *et al.*, 2014);
(vii) the CBA on seabed protection on the Frisian front and central oyster grounds (van Oostenbrugge *et al.*, 2015);
(viii) the report on nature-friendly area development, water safety, and transport infrastructure projects with five new case studies on biodiversity points (Jaspers *et al.*, 2016);
(ix) the CEA meta-study on defragmentation policy measures to enable wildlife crossing of roads, channels, and railway tracks (Sijtsma *et al.*, 2018, 2020).
(x) the CEA on (re)introducing limited tidal waves in the Grevelingen Lake in order to improve water quality (Fiselier & Botman, 2020).

The biodiversity point method is recommended by the new national guidelines on CBA (Romijn & Renes, 2013; Klooster *et al.*, 2018). Also specific guidelines have been developed on how to measure and use biodiversity points in CBA and ecological studies to support public decision-making on natural capital, ecosystem services, and biodiversity (Jaspers *et al.*, 2016; Rijkswaterstaat, 2020).

Biodiversity points are a practical and transparent method to quantify the impact of policy measures on biodiversity. They are especially useful for policy measures that have a major impact on ecosystems, such as nature policies or infrastructural works near nature or protected areas. They can be very helpful to formulate more nature friendly or cheaper policy alternatives or to find more cost-effective compensation measures.

For assessing the net benefits of projects, biodiversity points provide a standardized quantitative summary measure for the impact on biodiversity. This biodiversity measure can be decomposed into its constituent parts, is based on acreage of the impact area, internationally standardized ecological quality indicators and nationally standardized threat weights, and can be checked on its consistency of application for various CBAs. For assessing the overall effects of a project, this is more informative than qualitative (nominal or ordinal) expert opinions on a policy measure's impact on biodiversity; these are generally not standardized and comparable for different CBAs and cannot provide an indicator of change in biodiversity per euro invested. Sensitivity analysis with a range of reasonable different cost-prices per biodiversity point can show net benefits fully in monetary terms.

The use of biodiversity points can be advanced by providing overviews of their costs per point for various types of nature at various locations and various types of policy measures. This overview can give concrete examples of relatively cheap interventions for improving or protecting nature (e.g., a fish sluice in the enclosure dam of the Lake IJssel) and much more expensive ones. The overview can also discuss the factors determining these differences in cost-effectiveness. If such an overview is available, this would be a great help for assessing biodiversity points in another CBA or CEA.

Biodiversity points are quite similar to the quality-adjusted life years used for CEA of healthcare treatments. The major merits of biodiversity points are:

(i) simple, as a simple linear function is used with only four variables;
(ii) transparent, as it consists of a standardized formula with scores per ecotype for three variables;
(iii) relatively cheap, as it only involves assessment by ecological experts, and no expensive well-designed survey of willingness to pay is needed. But, of course, in comparison to a purely qualitative assessment by ecological experts, it is more expensive. This is the price to be paid for quantification;
(iv) uses a lot of information already available in EIA;
(v) linked to international classifications and lists of scarcity and therefore objective and independent and useful for international comparisons.

There are also various clear limitations:

(i) the biodiversity points measured generally pertain to one specific year and not to the whole period of the CBA;[10] this is a limitation if the impact is not the same for the whole period.
(ii) shows linear relationships, for example, biodiversity points of 2 hectare of an ecotope is two times 1 hectare and no threshold or minimum size required; this is a limitation as in ecology many relationships are nonlinear.
(iii) In calculating biodiversity points, perfect substitution is assumed between the size of the area, ecological quality, and ecological scarcity; this is a limitation as substitution will often be quite imperfect according to ecological experts.
(iv) no other criteria are used for assessing biodiversity; according to ecological experts, information like the number of birds or fish per species[11] may also be important to assess biodiversity.
(v) different methods of operationalization are possible, by choosing somewhat different ecotypes or species.[12]
(vi) the applicability of biodiversity points differs per type of biotope. It is more difficult for water quality-related biodiversity than for land biodiversity, as

10 So, preferably the average of the whole period should be taken; even some discounting may be considered (Koetse *et al.*, 2018).
11 Such other criteria can be added to the basic formulae of the biodiversity points (Fiselier & Botman, 2020; Rijkswaterstaat, 2020). However, this makes the method less simple.
12 This can be partly resolved by guidelines on this (see Jaspers *et al.*, 2016; Rijkswaterstaat, 2020).

the impact area is larger and the impacts are more difficult to define (Bos *et al.*, 2012; Bos & Ruijs, 2019).

(vii) expert opinions on scarcity are used, not a valuation reflecting preferences of individual citizens. As a consequence, biodiversity points are in particular suited for measuring the non-use value of biodiversity.

If the focus of the study is on the biodiversity impacts of a change, biodiversity points may not provide sufficient information for decision-making. Yet, as it summarizes all biodiversity information in one number, it makes it a very useful method in CEA.

The biodiversity points method is a practical method for measuring the non-use value of all kinds of biodiversity. It is ready for international application in CBA. During the past decade, the biodiversity points method has been applied to dozens of case studies in a wide range of contexts in the Netherlands. Various articles on the biodiversity points method and these case studies have been published in international journals and books on ecology and land use (Strijker *et al.*, 2000; Sijtsma *et al.*, 2011, 2013, 2020; van Puijenbroek *et al.*, 2015), and various other papers are available in English (Liefveld *et al.*, 2011; van Oostenbrugge *et al.*, 2015; Spaans, 2020) and have been presented internationally. Some translations of Dutch guidelines and case studies are also forthcoming.

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