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A Conceptual Framework for Assessing Comparability Between Corporate Biodiversity Impact Accounting Tools

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ABSTRACT

Integrating corporate biodiversity impacts into investment decisions can direct investments toward nature-positive firms, creating a market signal. The strength of this signal is a function of how closely investments align. This alignment depends on the comparability (the degree of similarity and difference) of tools used to account for biodiversity impacts. Yet, whether and why such tools are comparable remains poorly understood. Here, we develop a framework to discern and explain comparability by introducing two analytical perspectives: an input-based view (similarity in metrics and methods) and an output-based view (similarity in results). Expanding on the input-based view, we review 20 widely used tools, enabling 11 input-comparability assessments: five showed strong comparability, two moderate, and three low, with one unable to assess because of limited information. We then demonstrate how evidence from both perspectives can be integrated, revealing opportunities for future research, and discuss the implications of comparability for users and tool quality.

1 | Introduction

The world is in the midst of a biodiversity crisis. By various measures, biodiversity's decline has been precipitous. For example, a set of monitored wildlife populations has declined in abundance by 73% between 1970 and 2020 (WWF 2024). The current rate of species extinctions is tens to hundreds of times higher than the average background rate over the past 10 million years (IPBES 2019). Degradation of biodiversity is driven by land- and seascape change, resource overextraction, pollution, invasive species, and climate change (IPBES 2019; Maxwell et al. 2016). These drivers emerge, in part, from the actions of companies, which face only limited regulatory requirements to internalise into corporate decision-making the damages that their own operations and supply chains may have on ecosystems (Dasgupta 2021; Díaz et al. 2015). The impact of the economy on biodiversity, as well as the recognition of its dependence on goods and services that derive from nature (Dasgupta 2021), has raised the profile of reducing impacts as a priority for business and conservation policy worldwide (White et al. 2024).

Target 15 of the Kunming-Montreal Global Biodiversity Framework (Kunming-Montreal Global Biodiversity Framework 2022) is designed, in part, to catalyse the reduction of businesses' impacts on biodiversity through a system of enhanced non-financial disclosures and associated market action, referred to as 'greening finance' (Irvine-Broque and Dempsey 2023; Spinaci 2021). Alongside information on risks and dependencies, this target aims for the disclosure of data relating to corporate impacts on biodiversity. Using the stock market as one illustrative example, the process embedded in Target 15 could be considered to have three main phases (Irvine-Broque and Dempsey 2023; TNFD 2023b). Firstly, businesses create and disclose accounts containing quantitative metrics of biodiversity impact, broadly defined (i.e., to include genes, species, ecosystems). Secondly, investors use these accounts to improve the assessment of risk and opportunity in their investment decisions, buying more of lower impact and less of higher impact companies. This 'capital flow' between the stocks of investable companies results in stock price increases and decreases, respectively. Finally, stock price shifts incentivise operational changes in higher impact firms or drive such firms out of business, reducing the overall impact of the economy on biodiversity. However, the implementation of Target 15 and the emergence of a market incentive to reduce impact are undermined, in large part, by a lack of convergence on how corporate biodiversity impact accounts ought to be created (CBD 2024; Jwaideh and Dalin 2025; Karolyi and Tobin-de la Puente 2023; McKenzie et al. 2025).

Lack of convergence is coupled with limited guidance from policy-makers and standard-setting organisations. For example, the European Union's European Sustainability Reporting Standards (ESRS) E4 (European Commission 2023) has improved clarity on the topics to include in corporate biodiversity impact accounts but is agnostic on how these accounts ought to be created—allowing lack of agreement to persist. This lack of consensus and guidance has spawned a proliferation of different tools for producing and using biodiversity impact accounts (Ballon et al. 2025; Barth et al. 2025; Burgess et al. 2024; Granqvist et al. 2025; Katic et al. 2023; Lammerant et al. 2018, 2021; Layman et al. 2023). These *tools* use a range of *methods* to produce *metrics* to measure, *inter alia*, corporate pressures on biodiversity, the state of nature in a corporate location, and/or the impact of these pressures on the state of nature.

This broad and confusing landscape may have contributed to a situation in which there are few disclosures of biodiversity impact accounts by companies, and the lack of standardisation in these disclosures (Addison et al. 2020; Adler et al. 2018; Blanco-Zaitegi et al. 2022; Marco-Fondevila and Álvarez-Etxeberría 2023). Consequently, few investors have, as yet, integrated this information on biodiversity impacts into portfolio construction (Xin et al. 2023), despite recognising the materiality of these impacts (Giglio et al. 2023; Wagner 2023). Research on the pricing of biodiversity by markets has proved inconclusive, with conclusions seemingly a function of the biodiversity impact data chosen for analysis (Coqueret et al. 2025; Garel et al. 2024; Giglio et al. 2023; Xin et al. 2023). Nevertheless, financial products using tools for biodiversity impact accounting, applied to portfolio companies, are already available for investment (e.g., from AXA (2022) and BNP Paribas (2024)), with more likely to be launching soon. This emerging range of tools to assess corporate biodiversity impact by companies and financial institutions (Responsible Investor 2025), without scientific and standard-setter agreement, risks, at best, dispersing supportive capital flows (Berg et al. 2022; Chatterji et al. 2016), and, at worst, embedding ecologically harmful flows into the market (Layman et al. 2023).

One way to predict how financial flows for biodiversity may disperse and concentrate is by analysing the comparability of corporate biodiversity impact accounting tools, demonstrating where and when different tools are expected to agree and disagree. Comparability refers to the extent to which corporate biodiversity impact accounts created by two different tools for the same aspect of biodiversity lead to the same conclusions on relative corporate performance (FASB 2018). Beyond understanding capital concentrations and dispersions, understanding comparability has potential practical utility in helping companies choose tools, by shrinking the range of possible options, and helping financial institutions to make rational investments given unstandardised data. The concept of comparability, and a framework to assess it, has not yet been developed for corporate biodiversity impact accounting tools. Nevertheless, recent work has demonstrated that, while rankings of 500 large publicly traded US companies exhibit generally low correlation between eight available tools, certain individual tools are more similar to each other than others (Hickman et al. 2025).

In this paper, we aim to provide a framework that can discern and explain the comparability of one tool to another. To do so, we introduce two concepts of comparability: the input-based view (the similarities and differences of the metrics and methods used in a tool) and the output-based view of comparability (the correlation between results produced by tools). We then undertake an input-comparability assessment of 20 widely used and commercially available corporate biodiversity impact accounting tools. To do so, we use a schematic framework to review the metrics and methods of tools and qualitatively assess input-comparability between tools of the same type and for the same aspect of biodiversity. We then demonstrate how results from this input-based view of comparability integrate with emerging results under the output-based view, including how this integration can help develop new research questions that would further explain tool comparability. Finally, we discuss the implications of establishing comparability for tool users and for the emerging problem of establishing tool quality.

1.1 | A Conceptual Framework for Assessing Tool Comparability

1.1.1 | Defining Tools, Metrics, and Methods

We define a tool for producing a corporate biodiversity impact account as a specific data product, framework, or guidelines that produces a defined metric, using specific metrics, methods, and conceptual definitions. In principle, such tools can be used to create biodiversity impact accounts for the direct operations of a company (equivalent to the Scope 1 concept in the Greenhouse Gas Protocol (WRI and WBCSD 2004)) and also for the impact of sites, assets, and suppliers in a company's value chain (Scope 3).

We present a methodological schematic of a tool in Figure 1. Throughout this paper, we use italics when referring to the metric types that tools can produce, rather than when discussing the related concepts.

Phase 1 captures the definition, measurement, and metric creation for the pressures that a company or corporate asset places on biodiversity (*pressure* metrics). Pressures, sometimes known as 'impact drivers' (TNFD 2023b), are measurable quantities of natural resources that are used as inputs to production (e.g., water withdrawal for business consumption), or measurable non-product outputs of a business activity (e.g., water pollutants discharged) (Nature Capital Coalition 2016), that have some

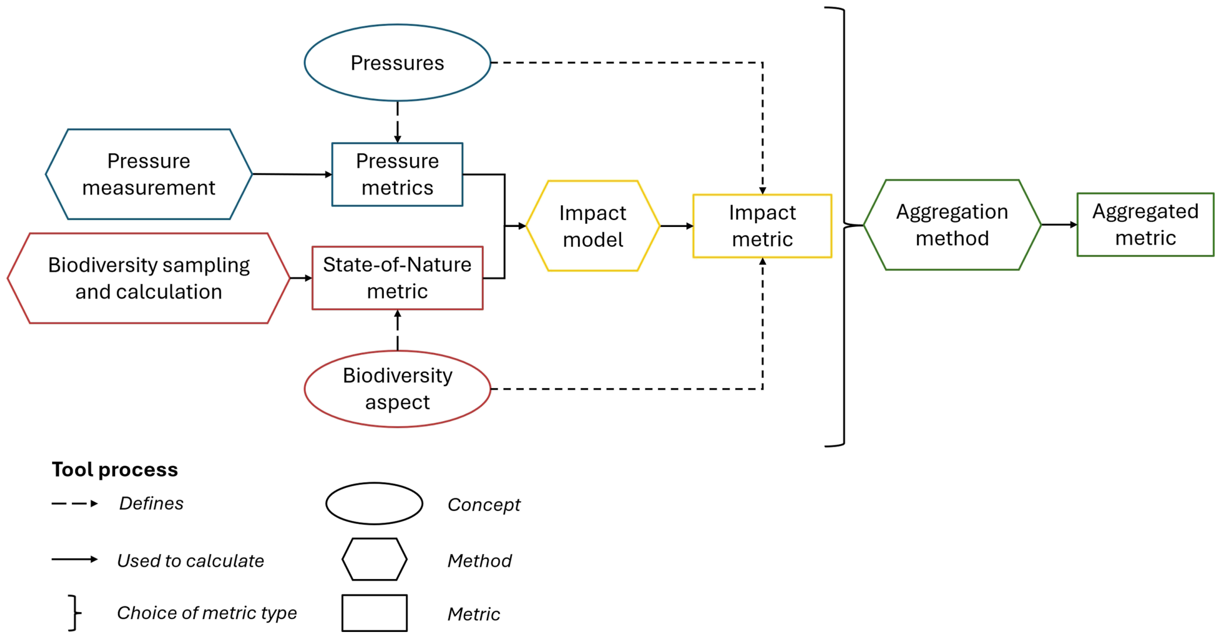

FIGURE 1 | A schematic of a tool used to produce corporate biodiversity impact accounts, showing four phases producing metrics of: (1) pressure (blue); (2) state-of-nature (red), (3) impact (yellow); (4) aggregated pressure, state-of-nature, or impact.

effect on biodiversity. The pressures selected to use in a tool will be those deemed material to a company, its financial position, and/or its stakeholders, given its sector (Jørgensen et al. 2021). Phase 2 captures the definition, sampling and calculation, and metric creation, for a focal biodiversity aspect, such as genetic diversity, species population, or ecosystem integrity (*state-of-nature* metrics). The state-of-nature refers to the quantitative status of an aspect of biodiversity in a particular area (Nature Positive Initiative 2024; TNFD 2023b). Phase 3, which is not conceived of in tool classifications like the TNFD (2023b), captures the modelling of an *impact* metric. This metric type describes the change in the state-of-nature that is attributable to the pressures produced by the company or specific asset that is being assessed.

The initial three phases of tool development can be applied to a specific asset, such as a factory. Phase 4 captures the aggregation of asset-level metrics to a company-level metric. At present, it is an open question which of the three metric types produced by the initial phases (*pressure* metric, *state-of-nature* metric, *impact* metric) ought to be aggregated to corporate level.

1.1.2 | Two Concepts of Comparability

Within a financial system with a wide variety of different tools in use, assuming consistent accounting scopes and equal use of tools, comparability is what causes capital flows to concentrate towards and away from certain companies. This is either because multiple investors are coming to the same conclusion about the relative biodiversity performance of a focal investable company by using different tools or because a rational investment decision can be made between two investable companies that report two different, but comparable metrics. In line with the accounting literature (Jia et al. 2022), the comparability of tools can be assessed under an 'input-based view' or an 'output-based view'.

Under an 'input-based view' (Barlev and Haddad 2007), tools are held to be comparable if similar metrics and methods are used. Tools found to be comparable under this view can be called 'input-comparable'. This is the approach used to establish comparability of freshwater bioassessments, through matching inferences from alternative steps in a methodological pipeline (Cao and Hawkins 2011). The input-based view conceives of comparability as movement down a qualitative gradient of similarity, away from 'identical'. Hence, classifications of comparability under the input-based view must come from content analysis of different tools and expert judgement. To facilitate content analysis, comparability assessments under the input-based view can use the schematic in Figure 1 to systematically review the concepts, metrics, and methods used in each tool.

Under an 'output-based view' (De Franco et al. 2011), firm-specific accounting procedures are allowed for by assuming that firms with comparable accounting methodologies will produce similar financial statements in response to the same economic events (De Franco et al. 2011). Hence, two tools for the same aspect of biodiversity may be comparable if, despite using different metrics and methods, they come to the same conclusions about the relative performance of a focal company against its peers. An output-based view of comparability, then, would allow one company, with accounts on a given biodiversity aspect prepared using one tool, to be reasonably compared with another company, with accounts prepared by a different tool. The output-based view, therefore, conceives of comparability as a measure of correlation between metrics produced by different tools under a range of situations. Tools demonstrated to fulfil these assessments can be thought of as 'output-comparable'. We were unable to source data that would allow for comparability assessments under the output-based view. Hence, we focus our quantitative analysis on the input-based view. See Hickman et al. (2025) for an example of a comparability assessment under an output-based view.

2 | Methods

We used the schematic in Figure 1 to review examples of tools for producing biodiversity impact accounts (Appendix S1). We identified tools from two previous reviews (Ballon et al. 2025; Lammerant et al. 2018) and screened for duplicates between the two sources and for the availability of publicly disclosed source documentation, excluding tools that did not produce quantitative metrics of corporate biodiversity impact or were sector-specific. We supplemented this list with tools from globally recognised sustainability disclosure frameworks: the Global Reporting Initiative (2024); the Nature Positive Initiative (2024); Accounting for Nature Framework (2023); and the Biological Diversity Protocol (Endangered Wildlife Trust 2020). In cases where tools calculated impacts on multiple aspects of biodiversity, we separated each of these out as separate tools. This screening produced 20 tools for review (Table 1). Classifying tools into one of three types, corresponding to the first three phases of Figure 1, we identified two producing *pressure* metrics, six producing *state-of-nature* metrics, and 12 producing *impact* metrics. For this analysis, we define the accounting scope as the direct impact of the company, which all tools can create. Some tools, such as Iceberg Data Lab's Corporate Biodiversity Footprint, can also create accounts for broader accounting scopes.

To assess input-comparability, we qualitatively assessed the similarity of the metrics and methods of tools within the same type and for the same aspect of biodiversity at each methodological phase (Figure 1; Appendix S1). The pairwise similarities and differences between tools were summarised by BS (Appendices S2–S4). SR and BS independently classified these assessments into one of four categories that represented their view on the strength of comparability: 'strong', 'moderate', 'low', and 'unable to assess'. They agreed with a Cohen's *kappa* of 0.5. Actual disagreement was on four of the 11 assessments, with the assessors disagreeing by one level in each case. These disagreements were discussed and resolved, with agreed results presented below.

3 | Results

From our review (Appendix S1), we found that, of the 18 *state-of-nature* and *impact* metric tools, there were 14 unique aspects of biodiversity measured. The most common aspects (used in > 1 tool) were species composition ($n = 4$), ecosystem condition/integrity ($n = 4$), extinction risk ($n = 2$), and species richness ($n = 2$). This led to one input-comparability assessment between tools producing *pressure* metrics (Figure 2; Appendix S2), two between tools producing *state-of-nature* metrics (Figure 2; Appendix S3); one between tools for ecosystem condition/integrity, and another between tools for species extinction risk, and eight for *impact* metrics (Figure 2; Appendix S4; seven between tools for impact on species composition and one for impact on species richness). Since our review of documentation found that only two tools (BDP(E), BDP(S)) clearly stated their aggregation method (Appendix S1; Figure 1), we discounted this phase from comparability assessments.

We found that the two tools producing *pressure* metrics in our sample (LIFE Key; GRI 101-6) had low comparability (Figure 2; Appendix S2). They both include land use and water stress but handle them differently: LIFE Key focuses on water consumption as a metric for water stress whilst GRI 101-6 uses water withdrawal. Beyond this, the tools share little in common, including using different pressure concepts, and LIFE Key's unique step of integrating pressures into a single index value, which has no comparison in GRI 101-6.

Comparability between tools producing *state-of-nature* metrics (IND2, IND4, STARt, AfN; Figure 2; Appendix S3) varied. We were unable to make a good assessment of the comparability of IND2 and AfN, both tools for metrics of ecosystem condition/integrity, because both tools allow the specific metrics and calculation methods to be selected on a case-by-case basis. Additionally, although we have represented AfN as a tool for ecosystem condition, it is able to generate condition metrics for a range of other environmental assets—which would not be comparable with IND2. However, we found comparability to be strong between IND4 and STARt, which both produce metrics on extinction risk. Methodologically, the two tools are extremely similar. Both tools calculate an extinction risk score for a given area by using the summed proportions of each threatened species' area of habitat present within the area. Key observed differences were the following: the optional weighting of area of habitat proportions by species threat status in IND4, an essential part of the methodology of STARt; the requirement to verify species areas of habitat in the most advanced versions of IND4; and the resolution of satellite data used—IND4 can use data with less than 1 km resolution, while STARt uses 5 km resolution. Nevertheless, as most of these differences are optional adjustments in IND4, we held the methodologies to be strongly comparable in the main.

We found a range for comparability assessments between *impact* metric tools (NRP(C), NRP(A), CBF, BFFI, BIA-GBS, BDP(E), Bioscope; Figure 2; Appendix S4). We assessed NRP(C) to have low comparability to other tools for species composition (CBF, BIA-GBS, BFM/C), as the only pressure shared between the tools is land use and NRP(C) uses PREDICTS (Hudson et al. 2017) and BII (Biodiversity Intactness Index, Newbold et al. 2016; Scholes and Biggs 2005) as its method and metric for *state-of-nature*, while the other metrics use GLOBIO3 (Schipper et al. 2016) and MSA (Mean Species Abundance, Alkemade et al. 2009).

On the other hand, the other *impact* metric tools for species composition (CBF, BIA-GBS, BFM/C) were assessed as having strong comparability because of extremely similar methods, particularly their use of GLOBIO3. Nonetheless, there were key differences in their sourcing of pressure measurements from different databases and inventories, such as EXIOBASE, Ecoinvent, and CBF's proprietary 'Wunderpus'. Importantly, however, in BIA-GBS and CBF, pressures can be sourced directly from the company, leading to high input-comparability in certain use cases. The three tools also differed slightly in the

TABLE 1 | Tools reviewed using schematic in Figure 1. Tools were screened from previous reviews (Ballon et al. 2025; Lammerant et al. 2018) and supplemented with tools from globally recognised frameworks that did not appear in these reviews.

Tool	Abbreviation	Creator	Origin	Source	Metric type	Aspect of biodiversity
LIFE Key (Biodiversity Pressure Index)	LIFE Key	LIFE Institute	2018	(Life Institute 2024)	*Pressure*	N/A
Global Reporting Initiative 101-6	GRI 101-6	Global Reporting Initiative	2024	(Global Reporting Initiative 2024)	*Pressure*	N/A
IND1 Ecosystem Extent (change and classification)	IND1	Nature Positive Initiative	2024	(Nature Positive Initiative 2024)	*State-of-nature*	Ecosystem extent
IND2 Ecosystem Condition	IND2	Nature Positive Initiative	2024	(Nature Positive Initiative 2024)	*State-of-nature*	Ecosystem condition
IND3 Landscape Intactness	IND3	Nature Positive Initiative	2024	(Nature Positive Initiative 2024)	*State-of-nature*	Landscape intactness
IND4 Species Extinction Risk	IND4	Nature Positive Initiative	2024	(Nature Positive Initiative 2024)	*State-of-nature*	Extinction risk
Species Threat Abatement and Restoration (Threat abatement STAR, STARt)	STARt	IBAT Alliance	2021	(IBAT Alliance 2021)	*State-of-nature*	Extinction risk
Accounting for Nature Framework	AfN	Accounting for Nature	2023	(Accounting for Nature 2023)	*State-of-nature*	Ecosystem integrity/condition
Nature Risk Profile (Composition)	NRP(C)	S&P Global Sustainable1 & UNEP	2023	(UNEP and S&P Global Sustainable1 2023)	*Impact*	Species composition
Nature Risk Profile (Structure)	NRP(S)	S&P Global Sustainable1 & UNEP	2023	(UNEP and S&P Global Sustainable1 2023)	*Impact*	Ecosystem structure
Nature Risk Profile (Function)	NRP(F)	S&P Global Sustainable1 & UNEP	2023	(UNEP and S&P Global Sustainable1 2023)	*Impact*	Ecosystem function
Nature Risk Profile (Aggregated)	NRP(A)	S&P Global Sustainable1 & UNEP	2023	(UNEP and S&P Global Sustainable1 2023)	*Impact*	Ecosystem integrity
Corporate Biodiversity Footprint	CBF	Iceberg Data Labs	2022	(Molinier 2023)	*Impact*	Species composition
Biodiversity Footprint for Financial Institutions	BFFI	PRé Sustainability	2015	(Broer et al. 2021)	*Impact*	Species richness/'Ecosystem health'
Biodiversity Impact Analytics – Global Biodiversity Score	BIA-GBS	CDC Biodiversité	2020	(CDC Biodiversité 2020, 2023)	*Impact*	Species composition

(Continues)

TABLE 1 | (Continued)

Tool	Abbreviation	Creator	Origin	Source	Metric type	Aspect of biodiversity
Biodiversity Footprint Methodology/ Calculator	BFM/C	Plansup	2016	(van Rooij et al. 2016; van Rooij and Arets 2017)	*Impact*	Species composition
Biological Diversity Protocol (Ecosystems)	BDP(E)	Endangered Wildlife Trust	2020	(Endangered Wildlife Trust 2020)	*Impact*	Ecosystem condition/integrity
Biological Diversity Protocol (Species)	BDP(S)	Endangered Wildlife Trust	2020	(Endangered Wildlife Trust 2020)	*Impact*	Focal species population size
Biodiversity Indicators for Site-based Impacts	BISI	UNEP-WCMC, Conservation International, and Fauna & Flora International	2020	(UNEP-WCMC et al. 2020)	*Impact*	Focal biodiversity feature, identified from site-level species and habitat lists using vulnerability, sustainability, and significance criteria
Bioscope	Bioscope	PRé Sustainability, Arcadis, and CODE	2016	(PRé Sustainability et al. 2016, 2022)	*Impact*	Species richness/"Ecosystem health"

range of pressures included, and their handling (e.g., air pollution in CBF) and the metrics used for shared pressures (e.g., BIA-GBS and BFM/C specification of kg N-eq. and kg P-eq. for water pollution, compared with CBF's use of 'business specific eco-toxic chemicals'). Finally, the three tools differed in their biodiversity measurement and impact model modules for water pollution (Appendix S1): BFM/C uses GLOBIO-Aquatic; BIA-GBS models water pollution impacts with PDF (potentially disappeared fraction of species) and ReCiPe (Huijbregts et al. 2017) before converting back to MSA with a proprietary link function; CBF does not state more detail than use of GLOBIO. Outside of these differences, however, the three tools are extremely similar (Appendix S1; 4). Hence, we concluded that accounts produced by these three tools will have strong comparability to each other.

For similar reasons, Bioscope and BFFI were found to be strongly comparable, sharing very similar pressure metrics and methods: ReCiPe2016 (Huijbregts et al. 2017) for biodiversity data and impact modelling, and *impact* metrics, which are proxies for impact on ecosystem health. The metrics produced by the two tools are the main point of difference: BFFI produces PDF.yr, which is interpreted as a proxy for natural area lost, while Bioscope multiplies a PDF.yr metric by average species density to produce species.year.

We found moderate comparability between the two *impact* metric tools for ecosystem integrity/condition (NRP(A), BDP(E)). These methods did not share similar inputs and may account for ecosystem condition/integrity differently; however, the 'most generally accepted or recognised method' (Endangered Wildlife Trust 2020, 5) clause in BDP(E) may allow for the use of NRP(A)'s Ecosystem Integrity Index, resulting in effectively identical tools in certain cases. Both methods also use a similar approach to calculating an overall *impact* metric: multiplying the average condition/integrity value over an area to derive a condition-adjusted footprint. However, NRP(A) calculates this over a focal asset's entire area, whereas BDP(E) restricts this calculation to within particular ecosystem types. In cases where there is only one ecosystem type in the asset area, the two methods would be equivalent.

4 | Discussion

We have presented a novel conceptual framework for understanding the comparability of tools for corporate biodiversity impact accounting. This framework introduces definitions of the tools, metrics, and methods of biodiversity impact accounting; a schematic that allows for review of tool documentation; and two concepts of comparability: the input-based view and the output-based view.

Using the schematic, we found that there is a wide range of different biodiversity aspects being considered by available tools (Table 1). In principle, this is encouraging, given the need for businesses to use multiple tools and metrics to capture different aspects of biodiversity (Purvis 2020). Nevertheless, all aspects of biodiversity captured by the tools we reviewed were at the level of species and ecosystem, with no tools capturing genetic diversity (Zhu et al. 2024). However, certain aspects of biodiversity were more commonly captured by tools than others. Of the 18

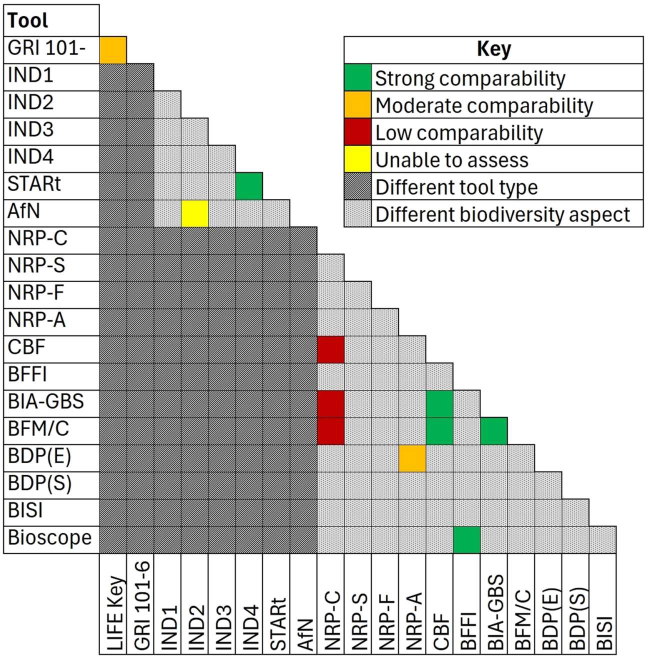

FIGURE 2 | Results of comparability assessments of biodiversity impact accounting tools of the same type and for the same aspect of biodiversity, under an input-comparability view. Metrics and methods of each tool were qualitatively assessed for similarity using expert judgement and separated into three categories. Because of the flexibility of AfN, we were unable to assess comparability to IND2. See Table 1 for abbreviations.

tools that incorporate *state-of-nature* metrics, nearly half captured change in, or corporate impact on, either species composition ($n = 4$) or ecological condition/integrity ($n = 4$). Practically, this reduced the number of comparability assessments we undertook. Indeed, it may suggest that if multiple metrics are used for different aspects of biodiversity (Addison et al. 2020), then issues related to divergence between tools will be restricted to species composition and ecological condition/integrity only. This may only be true for now as, if diversification of tools can happen for two aspects of biodiversity, it may very well happen for others soon.

4.1 | Integrating Emerging Output-Comparability Findings and Identifying Future Research Opportunities

Because of data limitations, we were unable to make our own assessments of comparability between tools under the output-based view. Some evidence of this kind is emerging in the literature, however. Our framework could help to conceptually integrate and clarify this evidence, generating further avenues for investigation.

For example, Hickman et al. (2025) demonstrated the correlations of S&P 500 firms, as ranked by various tools. In some cases, results strongly align. Their output-comparability finding of strong correlation ($r = 0.69$) between BIA-GBS and CBF aligns broadly with our input-comparability finding of strong comparability.

In other cases, an input-based view helps to clarify findings from output-comparability studies and motivate more detailed studies into what is driving comparability, including further development of the framework. For example, Hickman et al. (2025) compared NRP(A) with BIA-GBS and CBF, finding moderate levels of correlation. From an input-based view, this is not surprising, as BIA-GBS and CBF measure species composition, a subset of NRP(A)'s broader focus on ecological integrity (Noss 1990). However, the input-based view encourages further scrutiny of the drivers of this correlation by isolating NRP(C) as a species composition *impact* tool. We concluded that NRP(C) has low input-comparability with BIA-GBS and CBF. This would indicate that Hickman and colleagues' finding may be due to the correlation of these latter tools with NRP(A)'s structural and functional components. Since the tools share little input data (Appendix S1), the correlation is unlikely to be a methodological artefact (see Stevenson et al. 2024 for an example of this effect). Instead, this finding could be interpreted as a statistical artefact, as the tools are measuring fundamentally different aspects of biodiversity. Or, it could be interpreted as showing that compositional integrity ultimately underpins structural and functional integrity (Duelli and Obrist 2003) and that they would produce comparable biodiversity impact accounts. Edge cases like this highlight the importance of reflexivity when applying and developing the comparability framework (Montana et al. 2020). Future users should consider how strictly comparability should be bounded, whether only between tools addressing the same aspect of biodiversity, or also across complementary dimensions. Such reflection underlines that tool comparability is ultimately an ecological problem and illustrates the value of the input-based view in explaining the conceptual drivers of the output-based view's correlations.

The input-based view also facilitates an experimental approach to output-comparability that focuses on the methodological drivers of comparability. The schematic in Figure 1 allows structured manipulation of metrics and methods to create new tools. This would allow cross-sectional analysis that would identify whether comparability between tools is driven by the metrics or methods used. For example, MSA can be calculated using PREDICTS (Kuipers et al. 2025; Schipper et al. 2020), the method generally used to create BII (Hudson et al. 2017), as well as by other methods (Schipper et al. 2016). Comparing the outputs of a constant metric from these different datasets can inform the comparability of commercially available tools that use these methods. Similarly, methods could be held the same with variations in the metric, assessing the comparability of MSA and BII, for example.

Output-comparability assessments could also be conducted in virtual environments (Fulton et al. 2005; Nicholson et al. 2012), in which tool variations can be tested under controlled conditions. The same process could be replicated for real assets as data become available (Christiaen et al. 2025). At a much finer spatial scale, different metrics and methods, creating more and less input-comparable tools, could be applied to the same case-study corporation using directly sourced data (e.g., Martínez-Ramón et al. 2024).

Using both perspectives, further research is also needed to assess the comparability of tools within sectors. For example, NRP(C) only includes land use as a pressure. From an input-based perspective, it may be comparable with MSA/GLOBIO3-based tools within land use-heavy sectors, like forestry, but not in sectors producing a greater range of more indirect pressures (such as via climate change or pollution). Within-sector output-comparability assessments can cond.

4.2 | Implications of Establishing the Comparability Between Tools

The establishment of tool comparability by researchers has practical utility for tool users, in companies and financial institutions. First, identifying which tools are comparable may facilitate easier choice of tools by non-specialist users. If a set of tools is shown to be comparable, users can then select from that set based on more tractable criteria, such as price—a key consideration for companies in biodiversity monitoring (Herzog and Franklin 2016). Indeed, advancing a set of comparable tools could help to overcome debate on a single industry standard (Karolyi and Tobin-de la Puente 2023), which may be inhibiting the uptake of tools (EY 2025). Second, understanding comparability can help financial institutions negotiate the likely scenario in which investable companies begin to report their accounts using different tools. Knowing whether two company accounts are comparable allows rational allocation of investments between those companies. For example, our results and Hickman et al.'s (2025) suggest that a direct comparison can be made between a company using CBF and one using BIA-GBS. As further evidence on comparability develops through use of this framework, findings could be incorporated into decision-tools and guidance documents for metric users, such as the TNFD (TNFD 2023a).

On a broader scale, understanding comparability can help predict whether strong market signals for reduced biodiversity impact will develop. This has implications for predicting the conservation impact of Target 15. Our results, which align with Hickman et al. (2025), suggest, for example, that investments will be broadly similar between investors using CBF, BIA-GBS, and BFM/C. This would create strong pricing signals in the market, creating the incentive for investable companies to reduce biodiversity impact.

However, in real markets, price signals are affected by four other important factors, alongside comparability—on which more research is needed. First, the relative frequency at which tools are used can both concentrate and disperse capital flows. For example, we concluded that NRP(C) has low input-comparability to CBF, BIA-GBS, and BFM/C. Hence, equal or more frequent use of NRP(C) by investors would weaken pricing signals, as investments flow to a different set or ranking of companies. Present evidence of relative tool use is unrobust (Responsible Investor 2025), incentivizing further research—including on the relative use of different tool types. Second, the accounting scope used by investors is poorly understood. Broader scopes can account for the risks associated with supply chains, which are often large (Wilting and van Oorschot 2017). However, highly diversified 'universal owners' (Quigley 2020) may favour narrower accounting scopes, to avoid double counting. Third, the complexity of biodiversity is not well captured by a single tool (Purvis and Hector 2000). Thus, although some tools offer composite approaches, for example, NRP(A), most tools would need to be used in combination. How tools are combined can strongly affect the output (Burgass et al. 2017); hence, this process must be investigated.

4.3 | Comparability and Tool Quality

Research on comparability is complementary to efforts to establish the quality of different tools. Quality here relates to a tool's ability to discern a company's actual biodiversity impacts and its contribution to a nature-positive economy. Understanding a tool's quality can also help companies to choose between tools that have been shown to be comparable.

Traditionally, conservation science assesses tool quality by defining clear criteria and testing tools against them (Collen and Nicholson 2014; Czúcz et al. 2021). Such criteria are emerging for biodiversity impact accounting tools (e.g., TNFD 2023a) but testing and verification are rare to non-existent. Evaluating tool quality is complicated by sparse and imprecise source documentation for tools, which we found in our review process and has been noted elsewhere (Hickman et al. 2025). For example, few tools clearly describe the aggregation method of individual sites to corporate-level metrics (Appendix S1). This lack of transparency makes robust quality assessment difficult.

As yet, almost none of the tools assessed here have been externally verified against ground-truth data—a persistent issue in biodiversity metric evaluation (Collen and Nicholson 2014). A rare exception is a verification study of BII and PREDICTS, used in NRP(C) (Jung et al. 2017). They showed that BII and PREDICTS poorly predicted bird abundance along land use gradients in Kenya and Tanzania. More studies of this kind are required to assess performance over time and across ecological contexts, including in corporate settings.

Improving transparency and verification is essential to ensure that capital flows, directed by tool usage and their comparability, are ecologically helpful (Hickman et al. 2025). While the comparability framework does not directly assess tool quality, it can inform inferential quality assessments. For example, if one tool has been convincingly verified, tools sharing similar methodologies (input-comparability) or generating comparable results (output-comparability) can be inferred to be of similar quality. However, such inference should not replace direct verification. Thus, as quality assessments become more common, findings can feed into the construction of recommended sets of tools, alongside evidence from comparability. However, if a new tool is of extremely high quality and has poor comparability to tools of lower or unknown quality, clearly comparability is a secondary consideration for inclusion in recommendations. This new tool then becomes the new benchmark for comparability.

5 | Conclusion

Scientific research on comparability, as an important enabling function of nature-positive financial flows in markets, can support users to make better decisions, drive broad agreement across the market, and support monitoring of success in achieving Target 15. Hence, we strongly encourage fellow biodiversity scientists and financial economists to develop research on when and why the myriad tools available to create biodiversity impact accounts are similar or different.

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Supporting Information

Additional supporting information can be found online in the Supporting Information section. **Appendix S1:** Review of tools using schematic. **Appendix S2:** Input-comparability assessments of tools for *pressure* metrics. **Appendix S3:** Input-comparability assessments of tools for *state-of-nature* metrics. **Appendix S4:** Input-comparability assessments of tools for *impact* metrics. **Appendix S5:** References for Appendices 1–4.