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Facilitating the wise use of experts and evidence to inform local environmental decisions

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Abstract

1. Using biodiversity management within New Zealand's agricultural landscape as a case study, we apply 'boundary science' approaches to overcome two persistent deficiencies in environmental decision-making: 'evidence disparity', the discrepancy between evidence desired and evidence generated, and 'evidence complacency', where evidence is not sought or used, or is out of date, incomplete or biased.
2. Specifically, we assess how recent innovations in gathering, evaluating and communicating of evidence syntheses can: give local stakeholders from a diversity of roles and interests a voice in setting biodiversity priorities, systematically tailor global evidence to meet local needs and make wise use of local biodiversity specialists to enhance the accuracy and reliability of their judgements.
3. Initial case study material comprised comprehensive lists of 18 biodiversity outcome groups and 84 management actions of potential value to New Zealand farms. Of the 40 management actions that mattered most to stakeholders, 90% of actions required editing in preparation for the expert assessments. The final list of priorities encompassed 43 management actions and 11 biodiversity groups.
4. New Zealand evidence gaps were detected for the two actions and 10 biodiversity groups assessed, with half of these gaps plugged using global studies. Six experts then systematically evaluated this evidence to tailor its interpretation to the local context, mitigating the risks of using different evidence subsets and evaluation criteria.
5. The effectiveness of each action in delivering biodiversity benefits was also assessed by a panel of 10 experts using their specialist judgement to mitigate the risks of: (a) a skewed assessment derived from one or two experts; (b) overwhelming panellists with a long list of issues to debate; and (c) conflicts arising from misunderstandings about action outcomes.
6. The stakeholder priorities delivered useful insights, which could be used to direct and facilitate inclusive policy investments beyond our project. Auditing local studies using existing evidence synopses is recommended to help improve

local conservation evidence bases. Our study highlights the crucial role that a boundary-spanning team can play in gathering, organising, summarising and integrating datasets to address evidence disparity and complacency issues affecting local biodiversity management decisions required by global policy.

KEYWORDS
biodiversity, boundary science, decision-making, evidence, experts, judgement, priorities

1 | INTRODUCTION

Collective action is critical to achieving positive outcomes for the environment (Barnaud et al., 2018; Díaz et al., 2019; Pretty & Smith, 2004; Vermeulen et al., 2019), a need that is widely recognised by international policy (e.g. Aichi Biodiversity Targets Strategic Goal A). With calls for such action being largely driven by global bodies, a key challenge is how to empower national and local stakeholders to make policy and management decisions that deliver to such calls (Boyd, 2020).

Scientific evidence is a key source that environmental stakeholders draw on to inform decision-making. However to access such evidence, practitioners and policymakers are confronted with navigating the vast, growing and scattered body of scientific literature (Pullin et al., 2020), given that over 2.5 million scientific papers are published annually (Plume & van Weijen, 2014) and publication rates are increasing by 3%–4% p.a. (Ware & Mabe, 2015). In addition, determining how to interpret conflicting results or identify key knowledge gaps can be challenging (Pullin et al., 2020). Expert judgement is often drawn on, particularly, when resources are limited, timeframes are tight or empirical data or scientific evidence are characterised by high uncertainty or lacking altogether (Kuhnert et al., 2010; Sutherland, 2006). However, the value of expert judgement is dependent on how it is gathered because it can be influenced by cognitive and motivational biases (Burgman et al., 2011; Martin et al., 2012; McBride et al., 2012; Tversky & Kahneman, 1974). Furthermore, the value and use of expertise can be influenced by political and economic interests (Hajer, 1993; Sarewitz, 2004; Strassheim & Kettunen, 2014).

These issues drive two persistent deficiencies in environmental decision-making that deleteriously affect the actions chosen and the outcomes achieved: 'evidence disparity', the discrepancy between evidence generated by science and that desired by decision-makers (Christie et al., 2020; Fazey et al., 2005; Knight et al., 2008; Sutherland et al., 2011), and 'evidence complacency', whereby the available evidence is not sought or used by policymakers and practitioners, or the scientific information and knowledge used are out of date, incomplete or biased (sometimes relying on a single study or researcher; Dicks, Hodge, et al., 2014; Dicks, Walsh, et al., 2014; Donnelly et al., 2018; McKinnon et al., 2015; Pullin et al., 2020; Sutherland & Wordley, 2017).

'Boundary science', research that both advances scientific understanding and contributes to decision-making (Cook et al., 2013), aims to overcome such deficiencies. It requires the explicit development of boundary-spanning institutes or procedures to work at the interface of expert and decision-maker communities (Cash et al., 2003). Key proposals in this field include: (a) recognition of the different norms and expectations within different communities (e.g. researchers versus stakeholders) and facilitating effective communication between them (Cash et al., 2003); (b) the production of an accurate, concise and unbiased synthesis of the available evidence (Donnelly et al., 2018); and (c) use of structured processes for eliciting expert opinion (carefully selecting and facilitating expert panels to minimise bias and values) to increase their accuracy and reliability (Sutherland & Burgman, 2015). However, rigorous assessment of how such proposals are implemented, and the impact they have on environmental decision-making, is needed to substantiate their value.

Here we conduct such an assessment for a case study aiming to identify biodiversity priorities for New Zealand's agricultural landscape. Specifically, we assess the performance of (a) giving local stakeholders from a diverse range of roles and interests a voice in setting biodiversity priorities (MacLeod, Brandt, Collins, & Dicks, 2022), (b) systematically tailoring global evidence to meet local needs and (c) making wise use of local biodiversity specialists (i.e. applying the evidence on how advisors make assessments and predictions to mitigate the risk of cognitive weaknesses) to enhance the accuracy and reliability of their judgements in overcoming both 'evidence disparity' and 'evidence complacency' to improve the knowledge base made available to inform local environmental decision-making.

2 | METHODS

2.1 | Case study set-up

The case study was co-ordinated by two local researchers working in collaboration with the international Conservation Evidence initiative (Sutherland et al., 2019) to adapt and implement existing resources and protocols to meet local needs. The team aimed to function as an informal boundary-crossing institution (Cash et al., 2003; Cook et al., 2013), facilitating input from global and local experts, policymakers and practitioners, while enabling knowledge transfer among them and the different workstreams. Research was undertaken under social ethics approval (Manaaki Whenua – Landcare Research, application number: 1718/06); participation in our research was based on written informed consent through voluntary submission of the online surveys (Brandt et al., 2017) and email acceptance of the stakeholder or expert panel invitations.

Initial case study material comprised systematically generated comprehensive lists of 18 biodiversity outcome groups and 84 management actions of potential value or relevance in the New Zealand farming context (MacLeod et al., 2021; MacLeod, Brandt, Collins, & Dicks, 2022). These candidate lists were compiled, independently of any perceived value or effectiveness (Sutherland et al., 2014), from a range of existing resources (see MacLeod, Brandt, Collins, & Dicks, 2022 for more details). As detailed below, formal protocols were then applied to the gathering, organising, summarising and integrating of datasets on stakeholder priorities, study summaries and action effectiveness for agricultural biodiversity in New Zealand that incorporated: (a) local stakeholder perspectives, (b) unbiased evidence synthesis and (c) structured expert opinion. Unless noted otherwise, protocols and results were fully documented and reviewed by scientists external to the project (but within the host organisation). How these elements influenced the knowledge base subsequently available for stakeholders to draw on for potential future decision-making was assessed at each stage.

2.2 | Incorporating local stakeholder perspectives in biodiversity priorities

Online surveys were used at the outset to gather information on individual stakeholders' biodiversity interests and needs (MacLeod, Brandt, Collins, & Dicks, 2022). These surveys were designed using the candidate lists of biodiversity groups and management actions to ensure stakeholders considered the widest range of options and to provide transparency about what was deferred for future developments and why (Sutherland et al., 2014). Over 250 stakeholders, from public, private and civic sectors involved in managing New Zealand's farmland biodiversity, completed the surveys. The different perspectives of three key stakeholder groups (advisors, farmers and non-farmers), both overall and partitioned by the sectors or roles within each group, were summarised with the aim of reflecting the 10 specific biodiversity outcomes that matter most to stakeholders involved in managing New Zealand's agricultural landscape. The 40 management practices stakeholders considered most relevant to achieving those biodiversity outcomes were also summarised. These practices were associated with different farm areas, encompassing production areas as well as small or large biodiversity refuges, with the latter defined as areas expected to provide spatial and/or temporal protection from disturbances, or advantages in biotic interactions (Moller et al., 2008; Selwood & Zimmer, 2020), and include habitats that are management priorities in national environmental policy (e.g. Minister for the Environment, 2020). These summaries were reviewed by a stakeholder panel, who participated in a structured workshop led by a professional facilitator to ensure that the discussion was inclusive and deliberative, to reach consensus on final inclusive priorities (MacLeod, Brandt, Collins, & Dicks, 2022). The rationale for these agreed priorities and recommendations for refining the management practices' definitions or descriptions were documented.

2.3 | Drawing on unbiased synthesis of relevant global evidence

A small pilot exercise was conducted with the aim of demonstrating how existing global evidence syntheses could be adapted for local needs (Brandt et al., 2018a). The Conservation Evidence initiative was selected as the basis for this pilot study as it uniquely offers open-source subject-wide syntheses of patchy evidence for conservation interventions world-wide gathered from over 280 relevant journals and grey literature (Sutherland et al., 2019; Sutherland & Wordley, 2018). Global evidence was compiled for two priority management actions (*tillage methods* and *shelterbelts present*) and the 10 specific priority biodiversity groups. These two actions were selected as examples of actions that are implemented in different management areas of the farm (a production area and a small biodiversity refuge) and are relevant to different industry sectors. For each action, relevant study summaries were extracted from six synopses previously published by the Conservation Evidence initiative (Supplementary Information 2.1). A new synopsis was then composed for each priority management action (*tillage methods* and *shelterbelts present*), summarising the key messages from individual study summaries (Brandt et al., 2022), which were also presented (specifying their geographical locations, main methodological setups, direction of effects, response metrics, taxa and original sources; Sutherland et al., 2019).

2.4 | Using structured expert opinion to evaluate action effectiveness

2.4.1 | Expert panel purpose and construction

The effectiveness of each priority management action in delivering biodiversity benefits was assessed by a panel of biodiversity experts based on their specialist judgement (Brandt et al., 2018b). Experts scored 43 actions for overall biodiversity (comprising all taxa potentially occurring in the production landscape) and 10 biodiversity groups (i.e. $n = 473$ cases) based on their own working knowledge of the primary and grey literature, as well as their experience in New Zealand ecology, research and management. We also explored how providing evidence synopses to such a panel would influence their assessment, using a second panel to re-assess a subset of 10 cases (two actions for overall biodiversity and four biodiversity groups) for which such synopses were provided, as described above (Brandt et al., 2018a). Panel structure was guided in both cases, aiming to secure at least two specialists for each target biodiversity group and to facilitate an inclusive process by sending invitations to recipients (23 for the first panel and 35 for the second) across a diversity of roles, genders and institutes. Invitation recipients were selected from a candidate list of scientists from New Zealand and Australia specialising in at least one of the target biodiversity groups and with expertise in production landscapes. This list was assembled by searching all major New Zealand research organisations' staff lists and requesting expert recommendations. To test for potential biases in expert involvement, the probability of participation was modelled as a function of the panel method, role, gender and institute type (Supplementary Information 3.1).

2.4.2 | Expert panel process and behaviour

Structured assessments were undertaken by both panels via email, using a modified Delphi technique (Mukherjee et al., 2015; Sutherland et al., 2019). Each assessment involved two or three rounds of scoring for each management action, with the evidence synopses provided to the second panel. Experts initially independently scored the *benefits*, *harms* and *certainty* of each action based on their specialist judgement or the available evidence (Supplementary Information 3.2); for the evidence evaluation, panel experts also scored the *relevance* of the available evidence. Following one or two rounds of sharing and refining the anonymised scores (and any comments) with the expert panels, a consensus was reached on the effectiveness category (Supplementary Information 3.2; Sutherland, Dicks, Everard, & Geneletti, 2018; Sutherland, Dicks, Ockendon, et al., 2018) assigned to each management action and biodiversity group combination (hereafter, 'case'); these categories were adapted from the Clinical Evidence Handbook (2013). We tested for predictable patterns in scoring behaviours among the experts (based on their final scores) participating in the specialist judgement assessment in relation to their expertise, which farm area actions were applied, whether the biodiversity group was predominantly native or not and, for the benefits and harms models only, the aligned certainty scores (Supplementary Information 3.3).

3 | RESULTS

3.1 | Incorporating local stakeholder perspectives in biodiversity priorities

Although this exercise initially aimed to reflect the 10 biodiversity outcomes (in addition to overall biodiversity) that matter most to stakeholders involved in managing New Zealand's agricultural landscape and the 40 management actions they considered most relevant to achieving those outcomes (MacLeod, Brandt, Collins, & Dicks, 2022), 90% of those actions required editing in preparation for the expert assessments (Figure 1).

Half of the edits focused on translating the language used to define the action, aiming to make them more relevant to, and readily

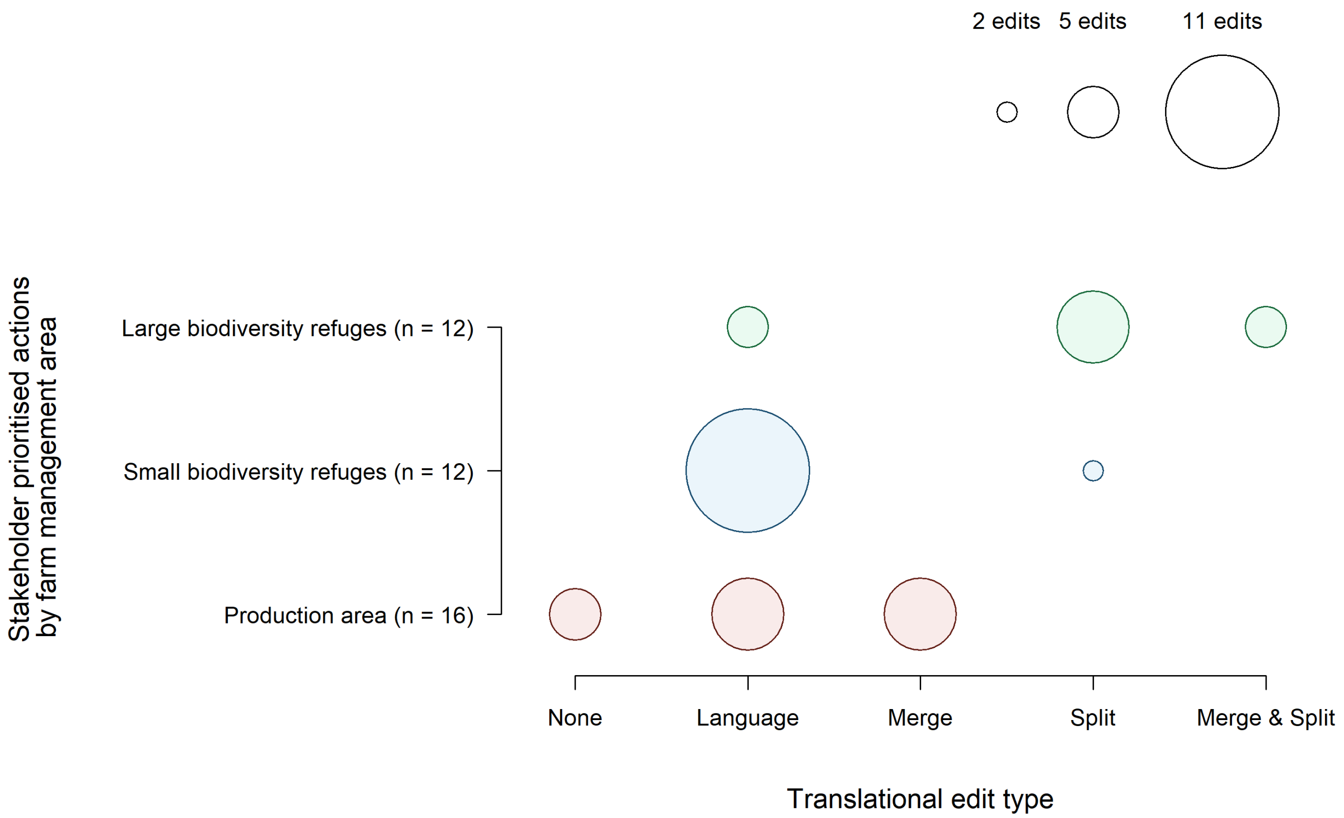

FIGURE 1 Types of translational edits applied to the original set of stakeholder prioritised actions ($n = 40$), grouped according to the management area on the farm in which they are implemented, in preparation for expert assessments, where the point size is proportional to the number of actions

understood by, New Zealand farmers; these edits were largely based on stakeholder recommendations and for actions aligned with small biodiversity refuges. Almost a quarter of actions (mostly associated with production areas) were merged with others to create fewer broad actions, as stakeholders recommended. A quarter of actions (mostly affiliated with large biodiversity refuges) were split to enable a clearer assessment of their expected biodiversity benefits (e.g. actions associated with natural habitats were aligned with a specific habitat to make it clearer which biodiversity groups would be likely to be affected), a judgement made by the boundary team based on their own specialist ecological knowledge.

On completion of this translational step, the final list of priorities encompassed 43 management actions (Supplementary Information 1) and 11 biodiversity groups (including one for *overall biodiversity*; MacLeod, Brandt, Collins, & Dicks, 2022).

3.2 | Drawing on unbiased synthesis of relevant global evidence

For the two management actions evaluated in our pilot, there were 104 relevant studies for *tillage methods* and 24 for *shelterbelts present* available in the Conservation Evidence synopses. The evidence was drawn from 47 journals (primarily English-language publications), with a small proportion coming from other sources; most journals (77%) only contributed one or two studies, with only two journals providing ≥15 studies (Table S2.2).

Evidence was gathered from studies across Africa, Asia, Europe, North America and Oceania (Figure S2.1), but no New Zealand studies were available in the Conservation Evidence synopses for the two focal actions. The studies also encompassed a variety of agricultural systems (Figure S2.2) but were predominately associated with cereal crops for *tillage methods* ($n = 81$ studies). Most evidence for both actions was derived from uncontrolled and observational study designs, with only a small subset having controlled and randomised designs (Figure S2.3). Overall, the available evidence encompassed multiple taxonomic groups but only aligned with four of the 10 target biodiversity groups prioritised by stakeholders (Figure S2.4): *soil life*, *beneficial insects*, *native birds of open habitats* and *native grassland plants*. Evidence was also patchily distributed among the biodiversity groups, with 10–65 studies per group for *tillage methods* but only 2–18 for *shelterbelts present*. Only a small proportion (7–8%) of individual studies provided evidence for multiple biodiversity groups.

By compiling these synopses, we have highlighted the lack of a New Zealand evidence base for two actions and 10 biodiversity groups considered priorities by stakeholders. By drawing on existing synopses of global evidence, we were able to plug nearly half of these gaps, albeit with limited evidence for one action (*shelterbelts present*). Gathering this evidence required the boundary team to fulfil another translational role: identifying the subset of relevant actions and biodiversity outcomes from the Conservation Evidence synopses and mapping them appropriately to the stakeholder priorities (Table S2.1). For example, the *tillage methods* evidence aligned to 11 actions dispersed across five Conservation Evidence synopses.

3.3 | Using structured expert opinion to evaluate action effectiveness

3.3.1 | Expert panel composition

Fifty-nine invitations to participate in our two panels were sent to 48 experts, with 14 local biodiversity specialists from nine institutes contributing (Table S3.1.1). At least two panellists had expertise in each taxonomic group being assessed (except domestic biodiversity and birds for the specialist judgement and evidence evaluation panels respectively).

Overall, the probability of participation was not related to institution type or role, but was threefold higher for the specialist judgement versus evidence evaluation panel and for male versus female experts (Table S3.1.2). The specialist judgement panel ($n = 10$) was more diverse than the evidence evaluation one ($n = 6$) in terms of genders, roles and institutes represented, albeit still with a high gender imbalance. This was despite a 52% increase in recruitment effort for the latter panel, where a high proportion of experts, who had previously agreed to participate, dropped out when the assessment was initiated.

Expert score ranges were larger when our structured recruitment process delivered a more diverse panel (Figure 2), suggesting it successfully captured a wider, and more representative, array of perspectives from New Zealand's biodiversity specialist community (with the caveat that assessment method may also account for some of this variance). The issue of the gender imbalance is not unique to our study, as women are generally under-represented across science disciplines, particularly among higher status roles, or are less likely to receive or accept invitations to participate (Fox et al., 2015; James et al., 2019; Wehi et al., 2019); hence, alternative strategies may be required to overcome this challenge and that of cultural inclusivity (which was not proactively addressed by our recruitment process).

3.3.2 | Specialist judgement panel

Each case (i.e. management action × biodiversity group combination; $n = 473$) was scored by seven to 10 assessors in the initial round, with five to eight assessors then indicating whether they agreed with the preliminary effectiveness categorisation for each case.

For 55 cases where ≥30% of respondents disagreed, a third round was conducted, with the revised median scores then used to assign the final effectiveness categories. These cases were distributed across 28 actions and all biodiversity groups. Compared to the initial scoring round, median benefit scores generally increased, while the score ranges decreased for *benefits*, *harms* and *certainty* (Figure S3.3.1). Most cases (67%) were initially categorised as *likely to be ineffective or harmful*, with 20% and 40% of those being reclassified as

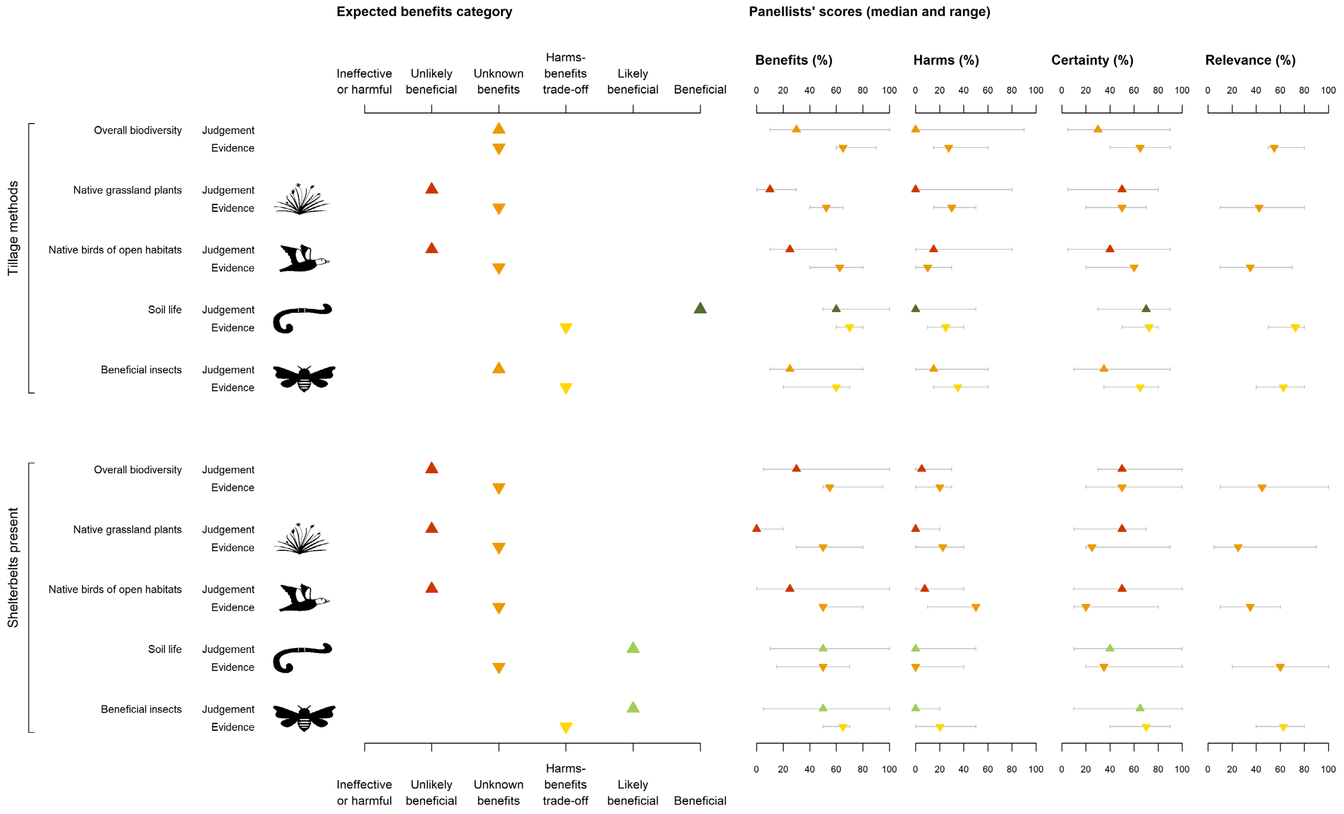

FIGURE 2 A comparison of effectiveness categories and panellist scores (benefits, harms and certainty) derived from the specialist judgement and evidence evaluation assessments (upward- and downward-pointing triangles respectively) for two actions and five biodiversity groups, including four target groups considered priorities by stakeholders. Note, relevance scores were only gathered in the evidence evaluation assessment

likely and *unlikely beneficial* respectively (Figure S3.3.2). Just over a quarter of the original effectiveness categories remained the same after the rescoring.

The final scores were highly variable among the experts participating in the specialist panel (when considering their score medians and distributions), particularly for *benefits* and *certainty* (Figure S3.3.3). Hence variation in scores was also generally high within and among the 473 cases (Figure S3.3.4), with ranges ≥70 for 81% of cases for *certainty*, 45% for *benefits* and 16% for *harms*. Judgement scores varied predictably in relation to four factors (Figure 3; Table S3.3.1): (a) specialists with expertise in the target biodiversity group scored benefits lower, but harms and certainty higher, than the other panellists; (b) cases associated with production areas were scored lower for benefits and certainty but higher for harms compared to small and, particularly, large biodiversity refuges; (c) when comparing biodiversity groups that were predominantly native with other groups, scores were generally lower, especially for benefits; and (d) *certainty* scores were positively associated with *benefits* but negatively associated, albeit very weakly, with *harms* (Figure 3).

Of the 473 cases assessed, 37% were categorised as *beneficial* or *likely to be beneficial* (Figure 4), with 57% *unlikely to be beneficial* or *likely to be ineffective or harmful* (Figure S3.3.5). Of the 43 actions assessed, ≥70% were expected to benefit *overall biodiversity*, *soil life* and *beneficial insects*, but only 13% to benefit *native grassland plants* (Figure 4). For the other seven biodiversity groups, ≥25% of actions were expected to deliver benefits. Seven biodiversity groups (but particularly *overall biodiversity*, *soil life* and *beneficial insects*) were expected to benefit from actions implemented in all three farm areas. However, for all but two biodiversity groups (*livestock, crop and variety*, and *native aquatic animals*), the highest number of effective actions was associated with large biodiversity refuges (Figure S3.3.5). Just 5% of cases were categorised as *unknown benefits* (Figure S3.3.5) because they had low certainty scores, most frequently for *native birds of open habitats* in large biodiversity refuges as well as *overall biodiversity* and *native aquatic animals* in production areas.

Approximately half of the actions were considered *beneficial* or *likely to be beneficial* for five or more biodiversity groups (Figure S3.3.6). *Control introduced herbivores* was judged the most effective action, benefiting nine biodiversity groups, with more than five biodiversity groups also benefiting from having small forests, controlling mammal predators, and four actions focusing on waterway and wetland areas. Actions expected to benefit a single biodiversity group were primarily associated with production areas.

By applying this specialist judgement panel process, we mitigated the risks of: (a) relying on the opinions of one or two experts, which would have provided a skewed assessment; (b) overwhelming time-pressured panellists by debating scores assigned to every biodiversity-action case, instead focusing on key issues of contention; and (c) potential conflict

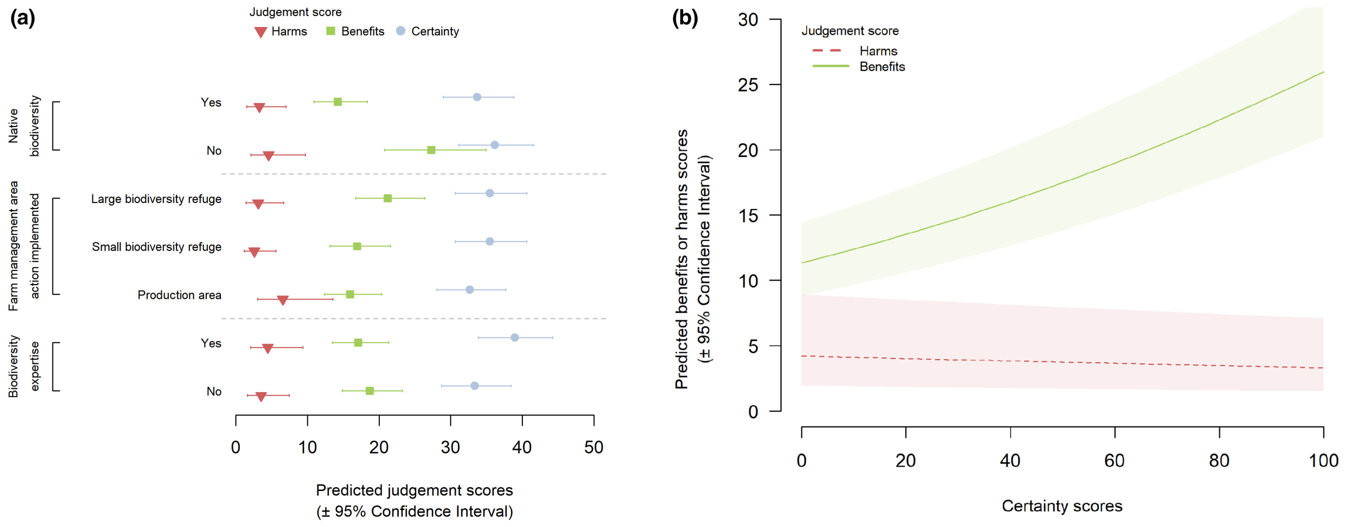

FIGURE 3 (a) Predicted estimates for harms (triangles), benefits (squares) and certainty (circles) from the specialist judgement panel in relation to three predictor variables: whether the specialist had expertise in the target biodiversity group or not, which management area of the farm each action was implemented and whether the biodiversity group was primarily made of native species or not. (b) Associations of benefits (solid line) and harms (dashed line) scores with the certainty scores were also investigated. These estimates were derived from the minimum adequate models, which accounted for repeated measures from each expert and action identity nested within each biodiversity group (see Table S3.3.1 for model results)

arising from farm performance evaluations based on misunderstandings about the expected biodiversity outcomes of given actions.

3.3.3 | Evidence evaluation panel

The score ranges for *benefits*, *harms* and *certainty* were generally lower for the evidence evaluation assessment than for the specialist judgement assessment (Figure 2). Of the 10 cases assessed by the evidence evaluation panel, only in one (*tillage methods* on *overall biodiversity*) did the effectiveness category match that of the specialist judgement assessment. Seven cases were categorised as having *unknown benefits*, primarily because the available global evidence was considered of low relevance to the New Zealand context. Three cases were classified as a *trade-off between harms and benefits* due to an increase in the median harms scores, thus downgrading two cases previously considered *beneficial* by the specialist judgement panel (*tillage methods* on *soil life*; *shelterbelts present* on *beneficial insects*).

By incorporating the evidence synopses into the panel evaluation process, we mitigated the risks of expert evaluations being based on: (a) different subsets of evidence, which are not transparent, are potentially biased and are derived from study designs of variable quality; and (b) different evaluation criteria, with a lack of transparency about the relevance of global evidence to the local context.

4 | DISCUSSION

Our research tackled the challenge of empowering local institutes and people to make better decisions to enhance environmental outcomes (Boyd, 2020) by ensuring stakeholders set the priorities—identifying the actions practitioners are most likely to implement (MacLeod, Brandt, Collins, & Dicks, 2022). Then, to begin to deliver to those priorities, by facilitating processes for making more effective use of local experts and global evidence when evaluating the expected outcomes of those actions (Figure 5). Specifically, we demonstrate here that (a) giving local stakeholders from a diverse range of roles and interests a voice in setting priorities that are immediately relevant and useful to them, (b) tailoring global evidence systematically to meet local needs and (c) making wise use of local biodiversity specialists to enhance the accuracy and reliability of their judgements can all help to overcome the persistent issues of 'evidence disparity' (Christie et al., 2020; Fazey et al., 2005; Knight et al., 2008; Sutherland et al., 2011) and 'evidence complacency' (Dicks, Hodge, et al., 2014; Dicks, Walsh, et al., 2014; Donnelly et al., 2018; McKinnon et al., 2015; Pullin et al., 2020; Sutherland & Wordley, 2017) in environmental decision-making. In our case study aiming to identify biodiversity priorities for New Zealand's agricultural landscape, without these elements the knowledge base available for stakeholder decision-making would have been more limited, more skewed and less accurate.

An informative approach to better understand the likely mechanisms by which these elements would enhance local decision-making, and whether their impact could be further improved, is to assess them against the four principles of inclusivity, rigorousness, transparency and accessibility, originally recommended for those producing or commissioning evidence syntheses (Donnelly et al., 2018).

4.1 | Inclusive

Our case study involved substantial in-kind contributions from a diverse range of individuals and organisations engaged in managing

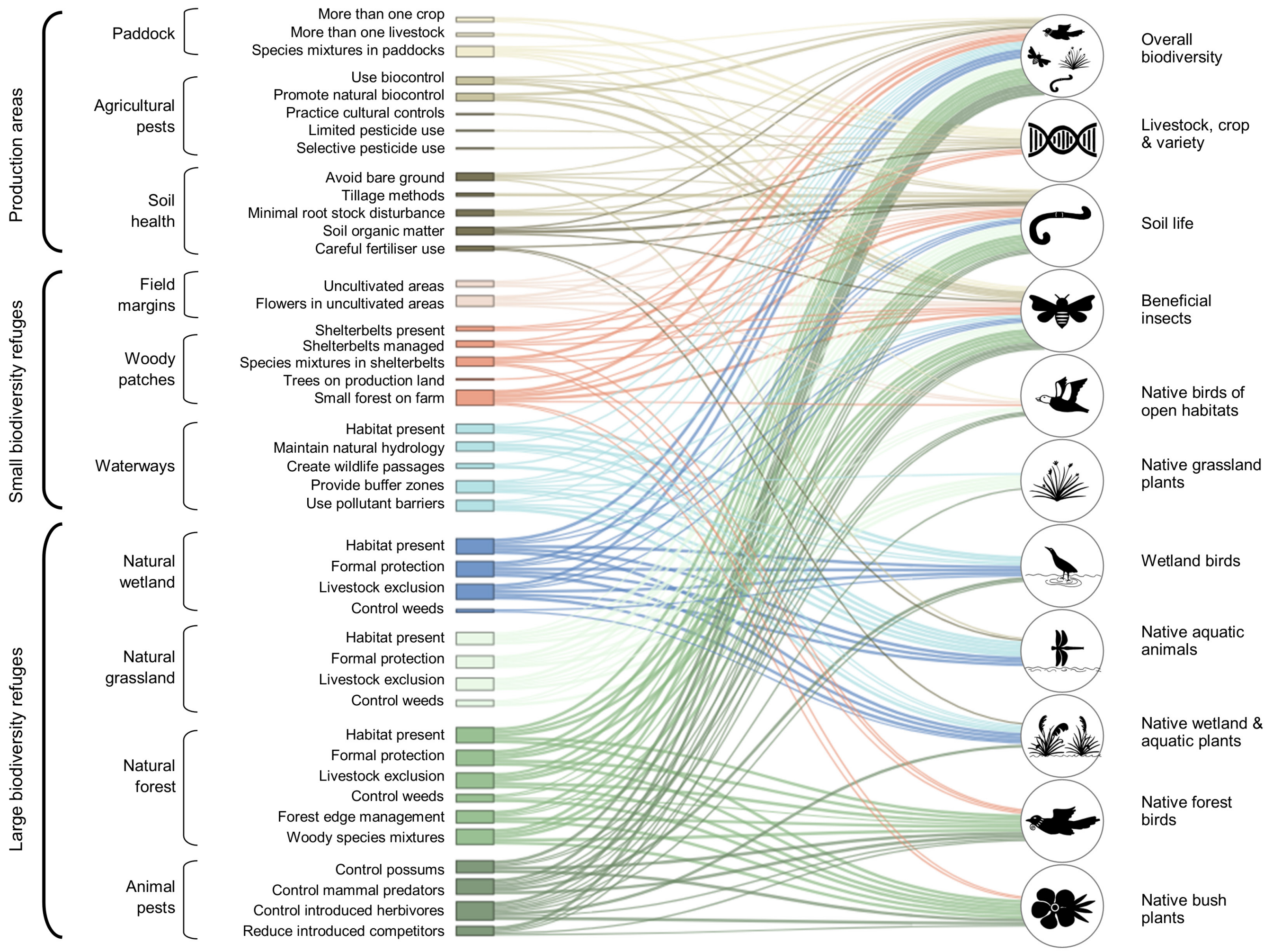

FIGURE 4 Forty-three management actions (applied in production areas, small or large biodiversity refuges of New Zealand farms) and the 11 biodiversity groups, 10 of which were target groups considered priorities by stakeholders. Actions are grouped and colour-coded in relation to their primary focus (e.g. a particular habitat or management issue); the box size for any given action is proportional to the total biodiversity benefit it is expected to deliver. Flows show the biodiversity groups expected to benefit from the implementation of each action (i.e. actions classified as *beneficial* or *likely to be beneficial* based on specialist judgement scores and effectiveness categorisation). See MacLeod, 2021 for more details on the underlying scores and benefit categories on specific actions or biodiversity groups

biodiversity in New Zealand's production landscape, thus drawing on and integrating their broad skills, roles and knowledge bases. It also achieved a fast start by incorporating information from a wide range of existing global and local resources (i.e. tools, frameworks, protocols, evidence synopses, policy documents and research publications), as well as valuable insights provided by overseas collaborations (Pullin et al., 2020).

Specifically, over 250 stakeholders helped decide which management actions and biodiversity outcomes should be prioritised and refined for the two expert evaluation processes (MacLeod, Brandt, Collins, & Dicks, 2022). Thus, multiple parties were empowered to influence the decisions and outcomes of our structured and transparent prioritisation process by adding their values, opinions and perspectives (Fung, 2006; Reed, 2008). Advantages of this approach included mitigation of bias or conflict risks (Midgley, 2016) as well as identifying a tangible starting point for our project (MacLeod, Brandt, Collins, & Dicks, 2022). It also provided a strong foundation for opening a much wider discussion about potential pathways for helping stakeholders to incorporate this expert knowledge and evidence into their decision-making, recognising that uptake and application of that information will be influenced by a complex array of behavioural factors (MacLeod, Brandt, Collins, Moller, & Manhire, 2022). However, our prioritisation process could also be improved in the future to better incorporate the perspectives of New Zealand public, including Māori, and overseas consumers.

Recognising that judgements drawn from diverse expert groups with expertise across relevant areas (Sutherland & Burgman, 2015), rather than one or two highly regarded individuals (e.g. McBride et al., 2012), are less prone to psychological and motivational biases, our evaluation processes involved 14 local biodiversity specialists with diverse expertise, roles and host institutes. However, our recruitment process could have more proactively sought junior scientists to reduce panel homogeneity (Sutherland & Burgman, 2015)

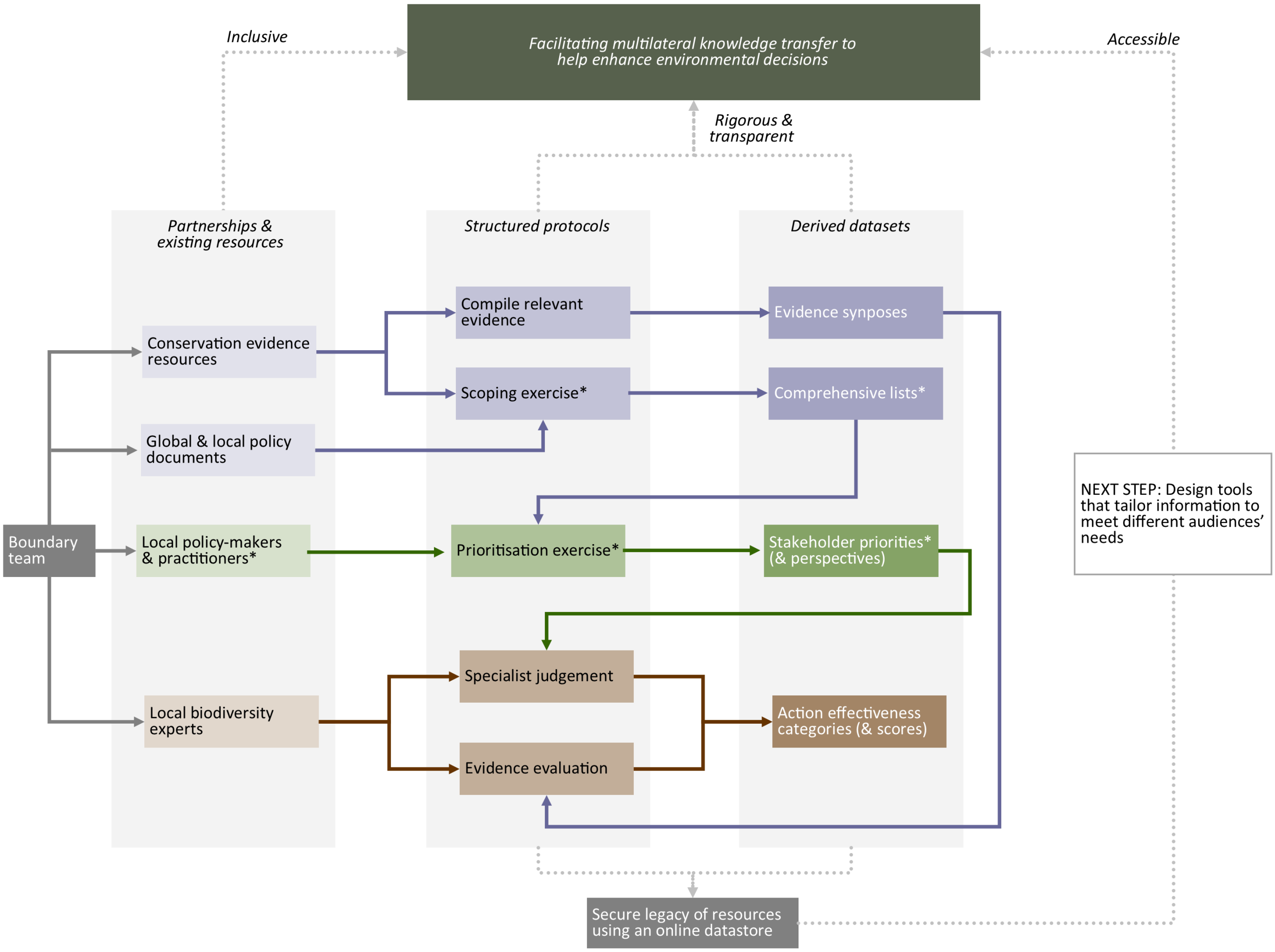

FIGURE 5 Overview of processes used to provide a proof of concept for environmental evidence synopses tailored to local needs. The design aims to meet four principles for good evidence synthesis (Donnelly et al., 2008): inclusive, rigorous, transparent and accessible. Asterisks indicate processes previously documented in detail in MacLeod, Brandt, Collins, and Dicks (2022)

and redress the gender imbalance (Fox et al., 2015; James et al., 2019; Wehi et al., 2019). Our specialist judgement assessment was also much faster and more intensive than typically conducted by the Conservation Evidence team due to our tight timeline, the limited pool of available local experts and the assessment's breadth. Potential strategies for overcoming these recruitment challenges include shortening the assessment, narrowing its focus to a specific biodiversity specialty and allowing for more flexible deadlines. In our case, a direct assessment of the relative influence of panel composition versus assessment method on the expert score ranges was not feasible; this could be the focus of a future meta-analysis across a range of panel evaluations.

4.2 | Rigorous

Our boundary-spanning team fulfilled a crucial role, working with stakeholders, experts and evidence to implement the different workstreams and facilitate information transfer among them (Figure 5), ultimately aiming to deliver a rigorous process within the project's resource and time constraints (Cash et al., 2013; Cook et al., 2013; Donnelly et al., 2018). This was achieved by implementing structured protocols to mitigate the risk of bias and minimise engagement costs. However, a key challenge was balancing the need for an inclusive stakeholder process with a meaningful expert evaluation. For example, stakeholders recommended some generic actions encompassing multiple management options important to land managers, which biodiversity specialists felt should be more specific to enable a more accurate assessment of their effectiveness. When independently testing a biodiversity assessment tool for New Zealand farms, which incorporated these actions and biodiversity scores (MacLeod et al., 2018), similar recommendations for more specific actions emerged as users believed this would provide a fairer evaluation of farm performance (MacLeod, Brandt, Collins, Moller, & Manhire, 2022). Difficulties aligning actions tailored to meet specific local needs with those in the Conservation Evidence synopses (Sutherland et al., 2019) is another potential challenge. Overall, stakeholders and specialists agreed that some actions needed editing to make the terminology more relevant to the New Zealand context (MacLeod, Brandt, Collins, & Dicks, 2022).

Our pilot study adapting global evidence for local needs demonstrates how local institutes can capitalise on the substantial global research effort implemented to evaluate the effectiveness of management actions (Pullin et al., 2020) by using existing open-source resources (e.g. Sutherland et al., 2019) that ensure evidence is gathered systematically (Haddaway et al., 2020). Specifically, our pilot study shows how the available global conservation evidence, which is often sparse and patchy (Sutherland & Wordley, 2018), can be readily used and tailored for local decision-making contexts, a task that might otherwise be prohibitive from a cost and time perspective. For example, our small pilot study evaluating the evidence available for two management actions alone drew on 128 studies across the globe (Figure S2.1), already systematically gathered from almost 50 journals and other publications (Table S2.2; Sutherland et al., 2019).

4.3 | Transparent

Evidence synthesis is more likely to be reliable, repeatable and beneficial if the authors are candid about its methods and limitations (Donnelly et al., 2018). Recognising that building local evidence bases will be a slow process, our project demonstrated mechanisms for how local institutes can make wiser use of experts to meet their biodiversity management needs in the short term. By highlighting the high variability of independent expert judgements and evidence evaluations among specialists (e.g. Figure 2), we reiterate the potential risk posed by seeking advice from only one or two individuals who may provide a very skewed perspective (Sutherland & Burgman, 2015). To resolve this issue, we demonstrate an iterative and transparent process for working with a specialist panel to efficiently reach a consensus when evaluating interventions by, for example, focusing their discussion on the 12% of action-biodiversity cases where there was most disagreement. However, it is difficult to determine from our process whether this exchange of information, and subsequent adjustment of scores, was indicative of group learning (i.e. new understandings emerging from the peer discussion) or anchoring (i.e. the group gravitates towards a particular estimate; McBride et al., 2012; Sutherland & Burgman, 2015).

Our analysis also shows that the expert panels behaved in predictable ways (Figure 3). For example, experts were more certain but more conservative in scoring benefits when evaluating their specialty biodiversity groups; a future challenge is, therefore, to determine whether this trend reflects their more in-depth knowledge and/or an unconscious bias. Such insights will be valuable for deciding whether their judgements should be weighted higher than those of the other panellists, and if training is required to help improve their judgement abilities (Sutherland & Burgman, 2015). Future research could also explore whether asking specialists to focus solely on their own area of expertise and evaluate biodiversity outcomes for different contexts (e.g. sector, region, landscape) not only helps to account for more variation among their judgements but also better addresses specific stakeholder needs (MacLeod, Brandt, Collins, Moller, & Manhire, 2022). Ideally, such a process should be informed by the available global evidence, with interactive tools developed to help experts work through a stepwise but systematic process of tailoring evidence for different contexts (e.g. Shackelford et al., 2020). Experts also tended to give lower benefit scores to native biodiversity groups and for actions associated with production areas (where they were less certain of the outcome). This warrants further investigation as it may reflect a lack of experts well-versed in both ecology and agronomy, subjective bias or an evidence gap. Comparing these judgements with those derived from an evidence evaluation could provide useful reflections on whether these expert perspectives are justified or not.

Panellists raised three recommendations for improving evaluations: (a) blend the standardised categories with those recommended by specialists to encourage end-user uptake of actions that benefit any biodiversity group; (b) make 'lack of benefit' and 'likelihood of harm' two independent categories, which would be valuable and remove a point of contention for some panellists; (c) adjust the threshold used to identify *beneficial* actions depending on evaluation method and the proposed use of the effectiveness categories.

Evidence maps provide a useful visual tool for quickly indicating data gaps as well as highlighting data-rich resources for exploring trends and examining causal relationships (McKinnon et al., 2015). In our pilot study, these maps highlighted several limitations, with the available global evidence gathered primarily in Europe and North America and none from New Zealand (Figure S2.1), and for only four of our 10 target biodiversity groups (Figure S2.4). Yet, despite these issues, local biodiversity specialists classified the evidence gathered from these studies as moderately relevant to the New Zealand context overall, but most relevant for *soil life* and *beneficial insects* (Figure 2). However, when evaluating the intervention effectiveness based on the available evidence, most desired biodiversity outcomes were classified as having *unknown benefits*, downgrading benefit categories derived from the specialist judgements.

Although practitioners and policymakers often prefer locally relevant studies, where the results are more readily transferable to their context, this lack of available local evidence is not an issue unique to New Zealand (e.g. Christie et al., 2020). However, it warrants further investigation, as a broader exploration of the Conservation Evidence synopses (Sutherland et al., 2019) signalled that this issue was not limited to our two focal actions. For example, New Zealand evidence only accounted for 7% of 164 management actions and 1% of 2449 studies within the synopses relevant to our project. Of the journals systematically searched by Conservation Evidence, currently only three are locally based, so it is possible that other local journals or grey literature could contribute relevant and reliable evidence. However, a quick search of the *New Zealand Journal of Zoology* using the terms 'tillage' and 'shelterbelt' found 65 and eight articles, respectively, indicating that these studies did not meet the criteria for inclusion in the evidence database.

4.4 | Accessible

By providing more transparency on which actions are most likely to deliver benefit and to which biodiversity groups (Figure 4), our research also holds huge potential to help land managers specify which biodiversity groups they are targeting and then make evidence-based decisions to prioritise their actions to deliver enhanced outcomes (Lobley et al., 2013; McCracken et al., 2015). Thus, making the results from our study (available in online research reports; Brandt et al., 2018a, 2018b), more readily accessible is a critical next step. It will require efficient and effective communication, tailoring the information for a diverse range of audiences as well as using appropriate channels to ensure potential users are aware of the available resources and know how to access and interpret them (Cooke et al., 2017; MacLeod, Brandt, Collins, Moller, & Manhire, 2022; Pullin et al., 2020; Wilson et al., 2016). Such communication strategies need to recognise that many users will have limited time and resources to engage or gather relevant information from detailed reports (Elliott et al., 2014; Travers et al., 2021), while others may question the resource's credibility (Cash et al., 2003; Cook et al., 2013). Work is underway to address these challenges of reaching and engaging different audiences through the development of online tools, ultimately aiming to increase the likelihood of stakeholders applying these results when developing and implementing land management policies and practices to enhance the biodiversity outcomes within New Zealand's production landscape (MacLeod, 2021, 2022; MacLeod et al., 2018; MacLeod, Brandt, Collins, Moller, & Manhire, 2022).

5 | CONCLUSIONS

Based on this assessment, the inclusion of (a) giving local stakeholders from a diverse range of roles and interests a voice in setting priorities, (b) tailoring global evidence systematically to meet local needs and (c) making wise use of local biodiversity specialists to enhance the accuracy and reliability of their judgements directly addressed three principles of good evidence synthesis (Figure 5; Donnelly et al., 2018). Our approach lays the groundwork for meeting the fourth principle of making evidence accessible (MacLeod, 2021, 2022; MacLeod et al., 2018; MacLeod, Brandt, Collins, Moller, & Manhire, 2022), which recognises that a lack of infrastructure for discovering, retrieving and processing relevant information from the scientific literature (Dicks, Walsh, et al., 2014) is one factor likely to be contributing to evidence complacency. The long-term aim is to shift the 'professional norm' to routinely demonstrating that stakeholder priorities are being met (Donnelly et al., 2018; McKinnon et al., 2015); experts are making clear the sensitivity of their decisions and insisting that robust techniques are used to evaluate their judgements (Sutherland & Burgman, 2015); and policymakers and practitioners are incorporating systematically gathered available evidence into their decision-making processes (Sutherland & Wordley, 2017).

Our process for identifying stakeholder research priorities delivered useful insights into a wide range of stakeholder perspectives, which could be used to direct and facilitate inclusive policy investments beyond our project. For example, this information could be used by researchers to develop strategic partnerships for building new research modules (either tailoring existing evidence or, where required, adding new evidence) focusing on key biodiversity groups or actions of interest. Policymakers could use the stakeholder priorities, in conjunction with evidence maps, to direct and evaluate future funding investments to address key knowledge gaps (Sutherland & Wordley, 2017) and track progress in the generation of evidence to meet them (Donnelly et al., 2018; McKinnon et al., 2015).

To work towards improving the state of local conservation evidence bases, we recommend a more in-depth audit of local studies in the context of the Conservation Evidence initiative, which would include: (a) investigating how often and why studies, for a given location, reviewed for the initiative are failing to meet the inclusion criteria, and if there is a need to expedite the use of more credible designs (Christie et al., 2020; Haddaway et al., 2020; Sutherland et al., 2019); (b) identifying and removing barriers to including any existing local evidence (e.g. grey literature), which will probably require much greater collaboration between research and practice (Baylis et al., 2016; Christie et al., 2020); and (c) determining if there is a genuine mismatch between local research investments and stakeholder needs, resulting in local evidence gaps that need to be proactively addressed (Christie et al., 2020; Fazey et al., 2005; Knight et al., 2008; Sutherland et al., 2011).

Our study provides an exemplar for the crucial role that a boundary-spanning team plays in gathering, organising, summarising and integrating datasets (Figure 5; Cash et al., 2003; Cook et al., 2013) to address evidence disparity and complacency issues affecting local biodiversity management decisions required by global policy (Christie et al., 2020; Donnelly et al., 2018; McKinnon et al., 2015; Pullin et al., 2020; Sutherland & Wordley, 2017). This essential translational role is time-consuming but often overlooked and/or under-resourced, as scientists favour prioritising research over investments that intentionally develop collective partnerships to increase the relevance, speed and likelihood of science informing and improving decision-making (Enquist et al., 2017). To tackle this challenge, funders should ensure that resourcing is appropriately allocated to the development of boundary-spanning teams (either as formal or informal institutes), and fulfilment of their role in ensuring mutual multiway learning and building the social capital necessary for delivering positive environmental outcomes (Pretty & Smith, 2004).

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CONFLICT OF INTEREST

The authors have no conflict of interest to declare.

AUTHORS' CONTRIBUTIONS

C.J.M. led the project and writing of the paper; C.J.M., A.J.B. and L.V.D. conceived the ideas and designed the methodology; A.J.B. led the implementation of the expert panel processes; A.J.B. and C.J.M. analysed the data. All authors contributed critically to the drafts and gave final approval for publication.

DATA AVAILABILITY STATEMENT

The data presented in this paper are available via the Manaaki Whenua—Landcare Research online DataStore: Stakeholder survey data (MacLeod et al., 2021) and Biodiversity expert scores (Brandt et al., 2022).

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