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Improving wetland health and condition monitoring through applying quantitative and qualitative methodologies: A pilot study from Northland, New Zealand

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Original Research Article: Wetlands

Improving wetland health and condition monitoring through applying quantitative and qualitative methodologies: A pilot study from Northland, New Zealand

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Abstract

Despite extensive losses over the past 200 years, New Zealand's freshwater wetlands continue to provide disproportionately high societal value through ecosystem services such as water attenuation and purification. Incentivising the restoration and protection of remnant wetlands, particularly on private land, could therefore deliver substantial benefits. Payment for Ecosystem Services (PES), through which landowners are rewarded for maintaining or improving ecological condition, provides one mechanism for encouraging such action. However, credible PES outcomes require robust monitoring, reporting and verification. Most approaches for assessing wetland condition in New Zealand typically rely on qualitative indices, yet few studies have tested their performance against quantitative ecological indicators. This pilot study examined whether a qualitative framework could accurately assess wetland condition, pressures and management responses when evaluated against quantitative metrics. We developed: (a) a qualitative Rapid Assessment Framework (RAF), structured around a pressure - state - response model, for monitoring and reporting; and (b) a quantitative composite formula, the Wetland Quality Index (*WQI*), as a verification tool. We then assessed the relationship between these methods across wetlands in the Northland region spanning a broad gradient of ecological quality. RAF Part B (wetland condition) was strongly and significantly related to the *WQI* ($R^2 = 0.92$), demonstrating the potential of the RAF as a robust monitoring and reporting tool for landowners and the *WQI* as an independent verification tool for wetland ecologists. Together, these methods provide a practical foundation for measuring ecological improvements and supporting monetary incentives for wetland restoration and protection on private land.

Keywords

Freshwater wetlands; Northland; rapid assessment framework; quantitative metrics; ecological quality.

Introduction

New Zealand's freshwater wetlands provide a wide range of critically important ecosystem services including flood attenuation, habitat provision, and water purification such as the removal of nitrogen, phosphorus, and pathogens such as *E. coli* (Clarkson et al. 2013; Goeller et al. 2019; Tanner et al. 2010). The monetary value of these services is disproportionate to spatial extent, with one study estimating that whilst comprising only 0.97% of the country's total land area they nevertheless provide an estimated 11.3% of net total economic value based on ecosystem service provision (NZD $14, 208 ha^{-1} $year^{-1}$. Cameron et al. 2020). However, approximately 90% of the former extent of New Zealand's wetlands has been lost over the last 200 years since European settlement (Cromarty and Scott, 1996; Ministry for the Environment, 1997) with remanent palustrine wetlands highly fragmented and a substantial proportion (estimated at ~60-65%) subject to formal protection within the protected area network (Ausseil et al. 2011a; Ministry for the Environment and Stats NZ, 2024). However, protection is biased toward nutrient-poor upland bogs and fens, while highly productive lowland swamps and floodplain wetlands - historically the wetland types most affected by drainage and land conversion - remain under-represented and disproportionately vulnerable to degradation. Consequently, only ~6–7% of the country's original wetland extent is deemed to be currently protected (Walker et al. 2006; Ausseil et al. 2011b; Clarkson et al. 2013).

The remaining extent of wetlands (~37%) are found on private land, and the ability to protect remnants through policy or regulation is often a contentious arena. For example, the identification of wetlands within regional plans, or their designation as Significant Natural Areas (SNAs), has occasionally generated conflict between regulatory efforts to protect ecologically significant areas and the imperative to maintain land for productive agricultural use (Clarkson et al. 2013). This conflict reflects a broader legacy of land-use change in New Zealand, where widespread wetland drainage and conversion for agriculture have been dominant drivers of wetland loss (Johnson and Gerbeaux, 2004; Clarkson et al. 2013), mirroring patterns observed across other ecosystems where agricultural development has been a primary cause of habitat loss and biodiversity decline (Foley et al. 2005). Remaining wetlands are frequently embedded within highly modified pastoral landscapes, where ongoing agricultural intensification and associated land-use expectations can be at odds with protection and restoration objectives (Myers, 2013; Lawson, 2021). At the same time, regulatory mechanisms intended to protect wetlands - including the Resource Management (*National Environmental Standards for Freshwater*) Regulations 2020 (NES-F) - may impose economic and operational constraints on landowners through wetland identification, land-use restrictions and nutrient-management requirements, reinforcing perceived trade-offs between environmental protection and agricultural productivity (Lawson 2021; Ministry for the Environment and Stats NZ 2024).

Subsequently, wetland protection within planning instruments often sits at the interface of competing land-use priorities and can contribute to contestation over land management decisions in rural landscapes. Wetland restoration efforts are similarly complex, as the re-establishment of appropriate

hydrological regimes and other conditions necessary for wetland structure and function - often through earthworks and other forms of physical site modification – are resource intensive, requiring investment of time, effort, and funds (Mitsch and Gosselink, 2015).

One means to incentivise the preservation and restoration of wetlands, particularly remnants on private land, is through Payment for Ecosystem Service (PES) initiatives where landowners are rewarded for activities that improve ecological health and condition, thereby providing a ‘carrot’ to counterbalance the ‘stick’ of policy and regulation. In application, this can potentially be realised in the context of green or sustainable lending where interest rates on debt may be reduced based on the assurance that on-farm activities improve wetland health and make a fundamental contribution to the provision of ecosystem services. Although green lending products may involve reduced margins at the individual loan level, lenders can be incentivised to offer preferential terms because such products attract additional borrowers and increase overall lending activity. In competitive financial markets, institutions offering more attractive green loan conditions are likely to secure a greater share of rural clients, generating income through increased loan volumes while enhancing their environmental and sustainability credentials.

Internationally, result-based agri-environmental PES schemes increasingly link payments to measured biodiversity outcomes rather than prescribed management actions (Herzon et al. 2018; Elmiger et al. 2023). Their effectiveness depends on indicators that are ecologically meaningful, responsive to management and sufficiently straightforward for landowners to apply. For example, farm-based visual indicators developed for Portuguese silvopastoral systems were applied by farmers and independently validated by technical specialists (Pinto-Correia et al. 2022). This outcome-oriented approach is also consistent with the emerging IUCN Green Status of Ecosystems framework which is being developed to assess past, current and potential future ecosystem status and the contribution of conservation actions to recovery (IUCN 2026).

Credible PES schemes require monitoring, reporting and verification (MRV) methods that can detect changes in ecological condition and provide confidence that funded actions are effective (Cameron et al. 2025). Rapid assessments offer a practical means of involving landowners in this process, but their simplified, observer-based scores require validation against independent ecological measurements. Although such tools are widely used across ecosystem types (Bouget et al. 2014; Davis et al. 2016; Dorney et al. 2018; Medeiros and Torezan 2013), few wetland studies have directly compared qualitative scores with quantitative field data collected concurrently (Stein et al. 2009; Faber-Langendoen et al. 2016; Yang et al. 2021).

We tested whether a landowner-oriented assessment could reliably represent wetland condition, pressures and management responses. A pressure-state-response Rapid Assessment Form (RAF) was developed for Northland wetlands and applied at nine sites spanning a broad ecological gradient. Quantitative measurements of wetland extent, vegetation composition, diversity, hydrology and other

ecological attributes were collected concurrently and integrated into a new Wetland Quality Index (*WQI*). Agreement between RAF and *WQI* scores was then evaluated to determine their respective potential for PES monitoring and independent verification. Research presented here is part of a wider project that also incorporates RAFs for native forests (Cameron et al. 2025) and streams. An online version of all three the RAFs - the *Nature Benefits Calculator* (https://ves.tanestrees.org.nz) – is planned which also incorporates Key Performance Indicators (KPI's) to measure progress as the response framework is implemented, a tool to estimate management-costs, and the monetary value added through improving the provision of ecosystem services.

Materials and Methods

Study locations and site descriptions

This study was conducted across fragmented, remnant wetlands on two Pāmu Landcorp farms, Kapiro and Omamari, in Northland, and two private farms in the vicinity of Omamari (Fig. 1, Online Resource 1). Maitahi Wetland Scientific Reserve, administered by the Department of Conservation, was also assessed and served as a reference wetland to Omamari sites given similar environmental conditions (e.g., potential species composition, climate, and edaphic profile (Durbecq et al. 2020)). Northland has a mean annual temperature of 15.5 - 16°C and receives approximately 1,500 - 2,200 mm of rain annually (NRC 2011). Fig. 2 presents representative photographs of each site.

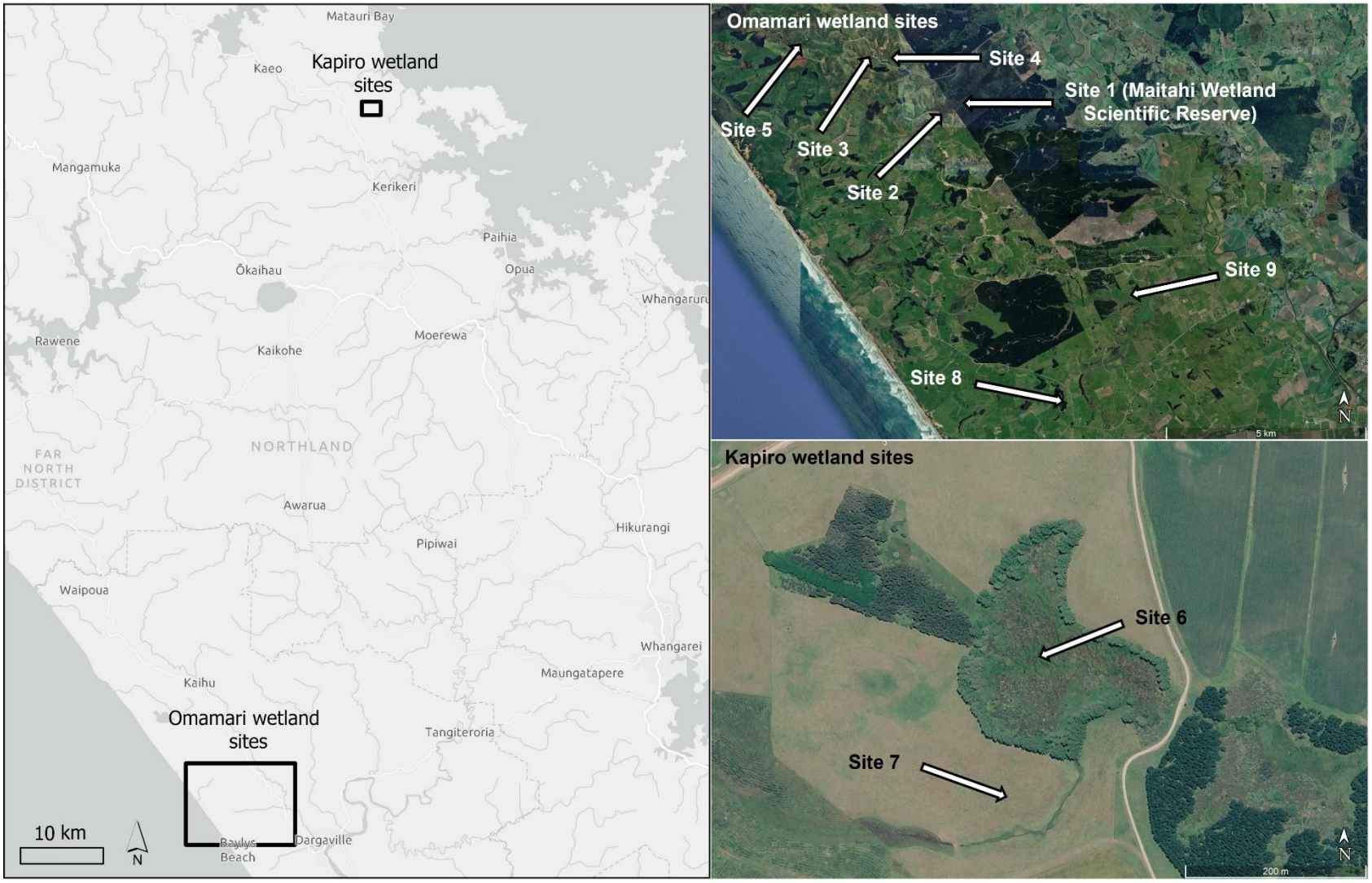


Fig. 1 Location of wetland study sites assessed in Northland, New Zealand. Eagle Technology, LINZ, StatsNZ, NIWA, Natural Earth, © OpenStreetMap contributors.


(1)
(2)
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Fig. 2 Photographs representative of all sites assessed in this study. From top left: (1) Site 1, Maitahi Wetland Scientific Reserve. Note the dominance of raupō and Machaerina rubiginosa, as well as scattered harakeke/flax, Mānuka, tī kōuka/cabbage tree and pampas grass. (2) Site 2, Ōmāmari Pāmu Farm, with a similar dominance of raupō and scattered tī kōuka/cabbage tree and pampas grass. (3) Site 3, Ōmāmari Pāmu Farm, which is primarily composed of raupō. Note pampas thickets in the background. (4) Site 4, Ōmāmari Pāmu Farm, with raupō and occasional tī kōuka/cabbage tree and pampas grass. (5) Site 5, Ōmāmari Pāmu Farm, composed of raupō and bordered by extensive pampas grass thickets. (6) Site 6, Kapiro Pāmu Farm, note the dominance of *Machaerina rubiginosa*, as well as common harakeke/flax and scattered tree ferns. (7) Site 7, Kapiro Pāmu Farm, note the dominance of swamp millet in the plot as well as other exotic grasses, sedges and rushes. (8) Site 8, Ōmāmari, with tall (~3m high) raupō in the centre of the wetland and boarded by tree fern and pine trees in buffer margins. (9) Site 9, Ōmāmari, composed almost exclusively of willow weed with various other exotic grasses present.

Importantly, in this study a 'wetland' was defined by whether it exhibited at least some of the prerequisite enabling conditions to facilitate wetland formation (e.g., located in a shallow depression with waterlogged soils). This contrasts with a stricter legal definition such as that contained in the Resource Management Act (1991) where the key denominator is whether a wetland already supports '*a natural ecosystem of plants and animals that are adapted to wet conditions*' (RMA 1991). This allowed for the inclusion of (a) existing poor-quality sites under wet soil conditions but which did not currently support a natural ecosystem of plants and animals, (b) sites which could naturally revert to wetlands in the absence of pressures, or (c) sites which could be actively managed to become wetlands (i.e., constructed wetlands) - a key consideration given the scale and extent of wetland loss in New Zealand and the need to encourage restoration.

Rapid Assessment Form development

The wetland Rapid Assessment Form (RAF) was developed by synthesising elements from established national and international assessment methods, including the Wetlands Monitoring and Assessment Kit (Denyer and Peters 2012), Wetland Condition Index and Wetland Pressure Index (Clarkson et al. 2003), Condition Index (Ausseil et al. 2011a), National Wetland Condition Assessment (US EPA 2016), Ecological Recovery Wheel (Gann et al. 2019), Kahikatea Forest Green Wheel (Denyer and Deng 2019), and the review by Fennessy et al. (2007). Collectively, these sources address key indicators of wetland condition including hydrology, water quality, vegetation, native fauna, introduced animals, wetland extent and surrounding land use, and provide approaches for evaluating sites against a desired reference condition. The organisational structure was also adapted from the forest RAF described by Cameron et al. (2026), although the indicators and scoring criteria were developed specifically for wetlands. Synthesising these sources provided a scientifically grounded foundation while enabling the RAF to address monitoring requirements not integrated within any single existing method (Wildland Consultants 2024).

The resulting RAF comprises 38 indicators arranged within a DPSIR-informed structure (Burkhard and Müller 2008), an approach also advocated by Denyer and Deng (2019). Part A records site metadata, while Parts B, C and D respectively evaluate ecological state, environmental pressures and management responses. This structure allows observed condition to be considered alongside its likely causes and the actions undertaken to address them.

Indicators in Parts B - D were assigned importance weightings through expert consensus. High-priority indicators receive a multiplier of 1.5, medium-priority indicators remain unadjusted, and lower-priority indicators receive a multiplier of 0.5. Part D additionally assesses management effort relative to site-specific need. Accordingly, limited action where intervention is required receives a low score, whereas an action that is unnecessary - for example, planting within a wetland already dominated by indigenous vegetation - receives a high score. Response scoring also recognises that coordinated interventions, such as stock exclusion, pest control and restoration planting, can produce greater ecological benefits than isolated actions. Scores for Parts B - D are combined and expressed as a percentage of the

maximum available weighted score, enabling comparison among wetlands and repeated assessment through time.

Evidence used to score pressures and responses was obtained from landowners, Pāmu Landcorp farm managers, pest-control contractors and the Department of Conservation. The complete RAF, site-level results, and rationale for indicator selection are provided in Online Resources 2 - 4 respectively.

Field assessments

Detailed vegetation sampling was conducted within 5 × 5 m plots at the same nine wetland sites assessed using the RAF, with both surveys undertaken by the same assessors (Sites 1-5 July 2024, Sites 6-7 August 2024, and Sites 8-9, March 2026). By assigning plant species to categories that reflect their fidelity to wetlands (e.g., obligate, facultative), vegetation sampling is useful for delineating and monitoring hydrological changes in a wetland and is used to develop the wetland Prevalence Index (Clarkson et al. 2013). Vegetation data was included as plant communities are often a reflection, at least in part, of the condition of ecosystems (Part B of the wetland RAF form) and the pressures under which they exist (Part C of the wetland RAF). Each plot was aligned due north, east, south and west, and all vascular plant species within the plot were identified to species level. For each species, the overall percentage cover was estimated and the maximum and average heights of each species recorded. Any species that were not encountered in the plot but found in the vicinity were listed. Depth to the water table was measured in each plot, although only once, given field sampling time constraints, and representative photographs were taken from each corner. The presence and extent of wetland buffer margin was also recorded.

As with vegetation, bird communities may be associated with changes in ecosystem health and other impacts. One five-minute bird count was conducted at each site following the method of Hartley (2012). Birds seen or heard to a maximum of 100 metres from the observer were recorded. Following the five-minute bird count, call playbacks for spotless crake (*Zapornia tabuensis*; At Risk–Declining), fernbird (*Poodytes punctatus*; At Risk–Declining) and banded rail (*Gallirallus philippensis*; At Risk–Declining) were undertaken (3 sets of 30 second playbacks followed by 30 second silences with one-minute silences pre and post playback). Birds seen or heard within 100 metres of the observer were recorded.

Development of the Wetland Quality Index as a quantitative metric for assessing wetland health and condition

We formulated the *WQI* with reference to several quantitative metrics developed (albeit for different end purposes) both within New Zealand (e.g. Ausseil et al. 2011a) and overseas (e.g., US EPA, 2016) as an alternative, but complementary measure, to the WCI and WPI. These two indices, while extremely useful and mainstays of wetland monitoring in the country, do not use quantitative data. The WCI (developed to assess wetland condition or quality) is instead reliant on qualitative pressure indicators evaluated as ‘change over time’, with the assumption that in the absence of stressors, quality or condition is optimised. The rationale underlying the WCI - that wetlands respond predictably to stressors

- may not always be the case and it does not enable a direct measure of the compositional components or inherent characteristics that define quality or condition itself (cf. the *WQI* and Part B of the RAF). The WPI essentially uses an inverse scale of most of the WCI components to record the presence and degree of pressures, representing a duplication in approach.

The *WQI*, in contrast, is a composite index derived from 15 quantitative metrics and developed as an empirical means to assess relative condition and compare wetlands within the same hydrological and vegetative classification. It is intended to be undertaken by trained wetland ecologists, with the underlying assumption being that the higher the value, the more that individual sites have retained most of their constituent components of ecosystems quality such as biodiversity content and ecosystem functionality (Ausseil et al., 2011a). The formula is:

$$WQI = \ln(WA) \times \ln(BM) \times \ln(R) \times Hleft \times \ln(rVPI \times 4) \times \ln(N) \times H' \times J' \times \ln(D\% \times 4) \times \ln(FC\% \times 4) \times \ln(BS) \times \ln(BS\%) \times \ln(H \times 4) \times \ln(SC \times 4) \times \ln(WTD\text{-}SD \times 4) \quad \text{(Equation 1)}$$

The individual components of the *WQI* are explained in Table 1. A number of indicators used in Part B of the RAF and the *WQI* measure similar attributes (e.g., S-1 and *WA*, S-2 and *WTD-SD*, S-4 and *N*, S-13 and *H*, S-14 and *SC*), albeit on differing scales and using qualitative and quantitative scoring systems respectively. This was designed to ensure complementarity, particularly given their intended use as monitoring and reporting tools by landowners (RAF) and verification metrics by wetland ecologists (*WQI*).

Each indicator is log transformed (apart from *Hleft*, *H*' and *J*', which are already transformed) using the natural logarithm (ln) to improve normality and reduce the range between different units (e.g., %, ha, or m), similar to the approach advocated by Ausseil et al. (2011) and Cameron et al. (2025). The *rVPI*, *D%*, *FC%*, *H*, *SC*, and *WTD-SD* are also given greater weighting and multiplied by four given these indicators are considered more temporally and spatially stable than other indicators (e.g., 5-minute bird counts which need repeated, consistent assessments) and, therefore, more important in reducing index variability. Testing at varying weighting scales was undertaken, with the intent to ensure that these six indicators (out of a total of 15) comprised at least half of the composite *WQI* score.

274 **Table 1.** *WQI* component indicators, formulas, and rationale for use.

***WQI* component**				**Explanation**
Abbreviation	**Full name**	**Unit/value**	**Formula/how calculated**	
WA	Wetland area	Ha	Delineated through GIS analysis (Google Earth Pro)	A critical hypothesis is that the greater the wetland area, buffer margin extent, and ratio of buffer to wetland, the higher the ecological quality of the site. Ecological quality in wetlands is strongly influenced by landscape configuration, with larger wetland area, greater buffer extent, and higher buffer-to-wetland ratios generally associated with improved biodiversity, water quality, and ecosystem function due to reduced edge effects, enhanced habitat availability, and increased protection from external disturbances (Semlitsch and Bodie, 2003; Mitsch and Gosselink, 2015). To avoid skewering results, the maximum width of the buffer margin (and by extension, buffer margin area) is capped at 30 m following Ausseil et al. (2011a) which avoids scenarios such as small wetlands surrounded by a fully forested, large catchment and buffer margin.
BM	Buffer margin	Ha		
R	Ratio of wetland area to buffer margin	%, or ratio of wetland area to buffer margin	$R = BM \div WA$	
Hleft	Hectares left	Ha	*Hleft* = *WA* × (current extent ÷ historic extent)$^{0.4}$	*Hleft* is modified from Ausseil et al. (2011a) where the hydrological class (e.g. swamp, marsh, bog, fen, or Pakihi / gumland) of each individual wetland is assigned following the classification system of Johnson and Gerbeaux (2004) and the extent of historical loss relative to contemporary extent of each class calculated. Results are log transformed to base 10 to avoid negative numbers. The 0.4 power function, as used by Ausseil et al. (2011a), was taken after Connor and McCoy (2001) to express the increasing relative value of an additional hectare as less remains.
rVPI	reversed Vegetation Prevalence Index	Index value	*rVPI* = $\sum(w_i \times p_i) / \sum p_i$	The *rVPI* is calculated where: • w_i= numerical wetland indicator value assigned to species *i* • p_i= proportional abundance of species *i* (e.g. % cover, relative cover, or frequency)

<table>
<tr>
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<td>The original prevalence index (PI) formula (Clarkson et al. 2013) classes wetlands with values <3 as higher quality by exhibiting a greater propensity for hydrophytic vegetation which is a key criterion for wetland delineation. Reversing the formula so that higher values (e.g. >3) are indicative of higher quality wetlands provides consistency with the interpretative function of other metrics in following a low-to-high quality scoring gradient, as shown below.

<table>
<tr><th>Category</th><th>Description</th><th>Original PI Value</th><th>rVPI value</th></tr>
<tr><td>OBL</td><td>Obligate wetland species</td><td>1</td><td>5</td></tr>
<tr><td>FACW</td><td>Facultative wetland species</td><td>2</td><td>4</td></tr>
<tr><td>FAC</td><td>Facultative species</td><td>3</td><td>3</td></tr>
<tr><td>FACU</td><td>Facultative upland species</td><td>4</td><td>2</td></tr>
<tr><td>UPL</td><td>Upland species</td><td>5</td><td>1</td></tr>
</table>
</td>
</tr>
<tr>
<td>N</td>
<td>Species richness</td>
<td>No. species</td>
<td>N = no. native species recorded from a wetland site.</td>
<td>The number of native flora species, recorded in plot assessments and within the general vicinity of the wetland and buffer margin.</td>
</tr>
<tr>
<td>H’</td>
<td>Shannon-Weiner Species Diversity Index</td>
<td>Index value</td>
<td>$\mathrm{H}' = -\sum_{i=1}^{S} Pi\ In(Pi)$</td>
<td>H’ was calculated from the number of native flora species recorded within plots for each wetland site, the proportion of species relative to the total number of individuals, and sums the proportion multiplied by the natural log of the proportion for each species as follows:

S= total number of species
p_i= proportion of individuals (or relative abundance) of species i</td>
</tr>
</table>

				$p_i = \frac{n_i}{N}$ n_i= number (or abundance/cover) of individuals of species *i* N= total number (or total abundance/cover) of all species.
J'	Pielou Evenness Index	Index value	$J' = \frac{H'}{In(S)}$	*J'* is log transformed and a measure of how evenly species are distributed in a community, calculated using *H'* and the total number of species (S) in a sample (Pielou 1966).
D%	Dominance of native flora	Percentage	$D\% = \left(\frac{N_{native}}{N_{total}}\right) \times 100$	*D%* is the relative abundance of native flora species recorded from sites and expressed as a percentage of all species recorded both within plots and the general vicinity. This variable is a measure of species richness and, by default, also accounts for the relative abundance of exotic and/or invasive plant species and therefore values wetlands with a higher native species abundance as exhibiting greater quality.
FC%	Percent cover native flora	Percentage	$FC\% = \left(\frac{\%_{native}}{\%_{total}}\right) \times 100$	*FC%* is the percentage cover of native flora recorded within plots and used as means to assess relative dominance (as opposed to abundance). Similar to *D%*, this is intended to favour wetlands which exhibit more native flora coverage higher on the quality continuum.
BS	Native bird species richness	No. native bird species	*BS* = no. native bird species recorded from a wetland site.	*BS* is the number of native bird species recorded during 5-minute bird counts for each site. For wetland Sites 8 and 9, 5-minute bird counts were unable to be conducted and instead we use an average from the Omamari Farm sites (1-5) as a proxy given similarity in hydrological and vegetative classes and associated habitat suitability.
BS%	Percentage of native bird species	Percentage	$BS\% = \left(\frac{\%_{native}}{\%_{total}}\right) \times 100$	BS% is the percentage of all bird species recorded that are native.
H	Height	Meters	*H* = tallest vegetation	*H* is the height, in metres, of the tallest native species within the wetland or buffer margin. Vegetation height can serve as an indicator of wetland maturity and structural development (Stefanik et al. 2012).
SC	Structural classes	No. structural classes	*SC* = no. wetland structural classes	*SC* is the number of wetland structural classes present following Johnson and Gerbeaux (2004). Greater representation of vertical and horizontal vegetation classes

				(i.e. increased structural complexity) is associated with enhanced habitat heterogeneity, supporting higher biodiversity and improved ecosystem functioning (Mitsch and Gosselink, 2015; Tews et al., 2004).
WTD-SD	Water Table Depth – Standard Deviation	Cm	$WTD - SD = 100 - SD(WTD_{mean})$	Water table depth (*WTD*) is calibrated to a desired baseline and calculated as 100 minus the standard deviation (*SD*) around the mean (WTD_{mean}). Water table depth in wetlands (measured in cm) is the vertical distance between the ground surface and the groundwater level, typically determined using installed dip wells or piezometers and recorded relative to a fixed surface datum. This is an important indicator of hydrological integrity (Ausseil et al. 2011a, Clarkson et al. 2013) and relevant for permanent (as opposed to ephemeral) wetlands but requires repeat measurements over the course of at least a year to capture seasonal fluctuations and derive a mean. Wetlands exhibiting low variability in water table depth around a natural mean tend to maintain stable hydrological regimes that support ecological integrity, whereas large or artificial fluctuations - such as those caused by drainage, impoundment, or altered surface runoff - can degrade wetland condition and facilitate shifts in vegetation and ecosystem function (Clarkson et al., 2003). By subtracting *SD* from a base unit of 100, it rewards lower fluctuations in the water table with a higher score in line with the other indicators composing the *WQI*. However, the timeframe for conducting the current pilot study precluded the ability to conduct repeat measurements to derive an estimate of relative water table fluctuation. While water table depth was recorded during field assessments, a one off measurement provides no intraspecific interpretative comparability and it was deemed illogical to compare water table depth between wetland sites as a quality indicator given *WTD* would vary as a function of variables such as catchment position, edaphic profile, and hydrological flow.

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Statistical analysis

Separate ordinary least-squares (OLS) regressions were used to examine relationships between: (a) *WQI* as the response variable and RAF Overall, RAF Part B and WCI as individual predictors; and (b) RAF Part C as the response variable and WPI, *WQI* and WCI as individual predictors (Greene 2018). Model performance and statistical significance were evaluated using R^2 and the F-statistic, respectively. Residual normality was assessed using the Jarque - Bera test, while the Breusch - Pagan test was used to detect heteroskedasticity (Breusch and Pagan 1979; Jarque and Bera 1980). Huber - White robust standard errors were applied to account for potential heteroskedasticity (Huber 1967; White 1980). All analyses were conducted using EViews 13 (IHS Markit 2022).

Principal components analysis (PCA) was used to examine multivariate relationships among the RAF components, WCI, WPI, *WQI* and individual *WQI* indicators, including their relative contributions to observed variation. Variables were standardised to zero mean and unit variance, and PCA was performed using a correlation matrix. True zero values, such as the absence of a wetland buffer or native plant records, were retained. Site scores and variable loadings were displayed in biplots to show similarities among wetlands and the direction and strength of variable contributions. The first two principal components were interpreted according to the variance explained and their representation of ecological gradients in wetland condition and pressures.

Limitations in implementation of the RAF and *WQI*

The accessibility of the RAF necessarily involves less taxonomic and technical precision than an expert-led assessment (Cameron et al. 2025). This is particularly relevant to PES monitoring, where evidence must demonstrate that management has produced additional ecological benefits (Maseyk et al. 2019). Wetland delineation can also require specialist knowledge because vegetation and soil characteristics reflect long-term hydrological conditions rather than water levels observed during a single assessment. Distinguishing wetland from dryland vegetation may be difficult for non-specialists, especially where sedges and grasses have similar morphology or diagnostic features are seasonal. Identifying hydric soils presents a comparable challenge.

Observer judgement introduces further uncertainty. Estimates of wetland extent, buffer width or fauna abundance may differ among users despite standardised questions. The RAF addresses this by providing descriptive guidance and allowing indicators to remain unscored where reliable information is unavailable (Wildland Consultants 2024). Nevertheless, training and calibration among assessors will be important for consistent application.

The study also treats the *WQI* as a quantitative representation of the ecological condition measured qualitatively by the RAF Part B. Like any composite index, however, the *WQI* simplifies ecological complexity and cannot fully represent species interactions, regeneration processes, temporal variability or three-dimensional habitat structure (Seidel et al. 2019; Ehbrecht et al. 2021). A wetland with a comparatively low *WQI* score may nevertheless have high conservation significance, for example by

supporting a threatened species. The *WQI* should therefore be interpreted as a comparative measure of condition rather than a complete assessment of ecological value. Its principal benefit is the integration of multiple field measurements into a repeatable metric that can support comparisons among similar wetlands and detect change through time.

Results

Rapid Assessment Form results

The RAF results for Part B (wetland condition) strongly reflected the continuum of ecological quality displayed by wetland sites – from exotic pasture dominated by hydrophytic adapted grasses (Site 7 and 9) through to protected wetlands exhibiting a high diversity and dominance of native flora (Site 1 and 6). Site 1 (Maitahi Wetland Scientific Reserve) and Site 6 (protected under a QE II Covenant) scored highest (10.25 and 8.92 respectively, or 79.4% and 69.1% of the maximum obtainable score), while the other Omamari wetlands (Sites 2, 3, 4, 5, and 8) can be considered intermediate in terms of ecological quality (Table 2). These sites are typically fragmented, smaller wetland remnants (S-1, median size 4.48 ha) that lack an extensive buffer margin (S-6, S-7) and exhibit moderate flora diversity (S-4) with Raupō (*Typha orientalis*) predominant, and pampas (*Cortaderia selloana*) thickets occurring in patches.

Across all wetland sites, the mean number of indigenous plant species recorded was slightly higher than the mean for exotic species. However, native species dominance was much higher at Site 1 (80%), Site 6 (66.7%), and Site 4 (53.3%), whereas exotic species were more prevalent at all other sites (range 25%-48%) where species such as pampas and hydrophytic grasses (e.g., Willow weed, *Persicaria maculosa*) dominated.

The RAF Part C (pressures) results are consistent with the trend observed in Part B, with Sites 1 and 2 again scoring highest (87.6% and 77.5%) and Sites 7 and 9 lowest (33.1% and 22.5%). Response (RAF Part D) results, however, differed considerably, with the wetlands on Pāmu farms (aside from Sites 5, 7 and 9) all scoring higher (range 67% - 89.6%) than Sites 1 and 8. In terms of overall results, Site 6 topped the site rankings at 87.04%, while Sites 7 and 9 – like they did for all individual Parts – ranked lowest with 10.2% and 7.2% respectively. Site 1, which had the highest ecological quality score, was ranked 4th (66.6%) largely as a result of a lower scoring response section (RAF Part D).

WQI results and relationship to the RAF, WCI, and WPI

Model fit statistics estimation (Fig. 3 [a], [b] and [c]) indicates positive relationships between the *WQI* and all independent variables, with varying strength across models (Online Resource 6). The *WQI* exhibited a strong, statistically significant correlation with Part B of the RAF (adjusted R^2 = 0.921, Fig. 3 [a]) and showed consistent patterns in terms of site rankings with Site 1 again the highest and Sites 7 and 9 the lowest scoring. The other sites, despite differences in positional rankings with Part B (Table 2), showed limited variation and were clustered largely around the median (median *WQI* = 31.6; range among Sites 2, 3, 4, 5, 6 and 8 = 8.7). The RAF Overall model, when specified in log-log form, captures

a nonlinear relationship (adjusted $R^2 = 0.61$) characterised by diminishing marginal improvements with the estimated elasticity coefficient ($\beta = 0.26$) indicating that a 1 unit increase in RAF Overall scores is associated with a 0.26 unit increase in *WQI* (Fig. 3 [b]). The WCI also displayed a strong relationship to the *WQI* (adjusted $R^2 = 0.74$), with the coefficient for WCI indicating that a one-unit increase in WCI is associated with a 1.40 unit increase in *WQI* (Fig. 3 [c]).

Unsurprisingly, RAF Part C (wetland pressures) and the WCI and WPI were strongly correlated ($R^2 = 0.837$ and 0.823 respectively. Fig. 3 [d]) given the similarity in measured qualitative variables, while a notable result was the strong correlation between RAF Part B and Part C ($R^2 = 0.796$) indicative that wetland state is strongly influenced by pressures (Online Resource 7). Jarque-Bera tests retain the null hypothesis of normally distributed residuals across all models. Similarly, Breusch-Pagan tests also retain the null hypothesis of homoskedasticity, indicating no strong evidence of heteroskedasticity.

When the *WQI*, RAF (all parts and overall), WCI and WPI were assessed through PCA results showed that Sites 7 and 9, the lowest ranking Part B and *WQI* wetland sites, were clear outliers and exerted a disproportionate influence on the ordination space (Fig. 4). In terms of the relative influence of component indicators for the *WQI*, Axis 1 is driven primarily by the number of structural wetland classes present (*SC*) and the relative dominance of native species (*D%*), with substantial contributions from *Hleft*, *N*, and *WA*, confirming that the primary ordination axis reflects a composite gradient integrating structural and compositional aspects of wetland condition (Fig. 5). In contrast, Axis 2 is dominated by *BS*, *BS%*, and *J'*, with additional contributions from *FC%*, *rVPI*, and *H*, indicating that the secondary axis captures variation associated with vegetation structure and diversity-related metrics rather than overall *WQI* condition. Of interest, results for the *rVPI* showed marginal relative variation (range 4.0-4.98) with all sites (including Sites 7 and 9) passing the >3 threshold for exhibiting wetland-specific vegetation. When the relative contribution of individual indicators to the *WQI* is assessed, the five weighted variables (*rVPI*, *D%*, *FC%*, *H* and *SC*) comprised on average 60.5% of site scores and therefore surpassed the 50% threshold the study design required.

384 **Table 2**. Summation RAF results and *WQI* parameters (not log transformed) from wetland sites assessed in this study.

Location	**Omamari**					**Kapiro**		**Omamari**	
Site #	**Site 1**	**Site 2**	**Site 3**	**Site 4**	**Site 5**	**Site 6**	**Site 7**	**Site 8**	**Site 9**
Wetland hydrology/vegetation classification	Swamp/Reedland	Swamp/Reedland	Swamp/Reedland	Swamp/Reedland	Swamp/Reedland	Swamp/Sedgeland	Swamp/Grassland	Swamp/Reedland	Swamp/Grassland
RAF score: Part B wetland state and % of max. (rank)	10.25, 79.72% (1)	7.42, 57.69% (4)	6.13, 47.64% (6)	6.96, 54.12% (5)	5.38, 41.81% (7)	8.92, 69.40% (2)	3.08, 23.93% (8)	7.76, 60.33% (3)	2.35, 18.25% (9)
RAF score: Part C wetland pressures and % of max. (rank)	11.27, 87.6% (1)	8.86, 68.9% (3)	7.14, 55.6% (7)	7.64. 59.4% (5)	7.21, 56.1% (6)	9.96, 77.5% (2)	4.25, 33.1% (8)	7.73, 60.1% (4)	2.89, 22.5% (9)
RAF score: Part D wetland responses and % of max. (rank)	85.50, 63.3% (5)	115.00, 85.2% (2)	90.50, 67% (4)	101.00, 74.8% (3)	84.50, 62.6% (6)	121.00, 89.6% (1)	9.00, 6.7% (8)	30.80, 22.8% (7)	6.40, 4.7% (9)
RAF score: Overall and % of max. (rank)	107.4, 66.59% (4)	131.7, 81.68% (2)	104.0, 64.57% (5)	115.8, 71.93% (3)	97.5, 60.41% (6)	140.2, 87.04% (1)	16.6, 10.16% (8)	46.6, 28.80% (7)	11.8, 7.24% (9)
WQI score (rank)	39.26 (1)	31.55 (5)	26.67 (7)	32.91 (3)	27.72 (6)	31.81 (4)	22.02 (8)	35.34 (2)	19.75 (9)
***WQI* indicators**									

WA: Wetland area (ha)	33.40	22.76	2.81	5.53	1.50	2.78	0.60	4.48	3
BM: Buffer margin area (ha)	13.68	2.84	0.67	3.14	1.96	0.56	0	9.64	0
R: Ratio of wetland area to buffer margin area (%)	0.41	0.12	0.24	0.57	1.31	0.20	0	2.15	0
Hleft: Hectares left	2.18	1.92	1.05	1.45	1.05	1.03	0.29	1.661	0.9881
rVPI: Reversed Vegetation Prevalence Index	4.84	4.97	4.98	4.57	4.56	4.63	4.87	4.49	4.0000
N: Species richness (no. native flora species)	20.00	6.00	13.00	8.00	4.00	12.00	7.00	10.00	1
H': Shannon-Weiner Species Diversity Index	1.47	0.04	0.22	1.04	0.36	1.19	1.04	0.45	0
J': Pielou Evenness Index	0.59	0.03	0.16	0.64	0.52	0.61	0.75	0.549	0

D%: % Native species dominance (wetland site)	80.00	48.00	43.33	53.33	36.36	66.67	38.89	43.50	25.0
FC%: % Native flora cover (within plots)	99.22	98.28	97.69	82.20	62.09	99.56	93.33	77.44	0
BS: No. native bird species recorded	4.00	5.00	6.00	8.00	8.00	9.00	1.00	6.20	6
BS%: % bird species recorded native	57.14	50.00	60.00	66.67	61.54	75.00	20.00	59.07	59.1
H: Height of tallest native species within wetland/buffer margin (m)	12.00	8.00	1.00	5.00	1.00	12.00	0.30	9.83	5.0
SC: No. wetland structural classes present	4	3	2	2	2	4	1	3	1

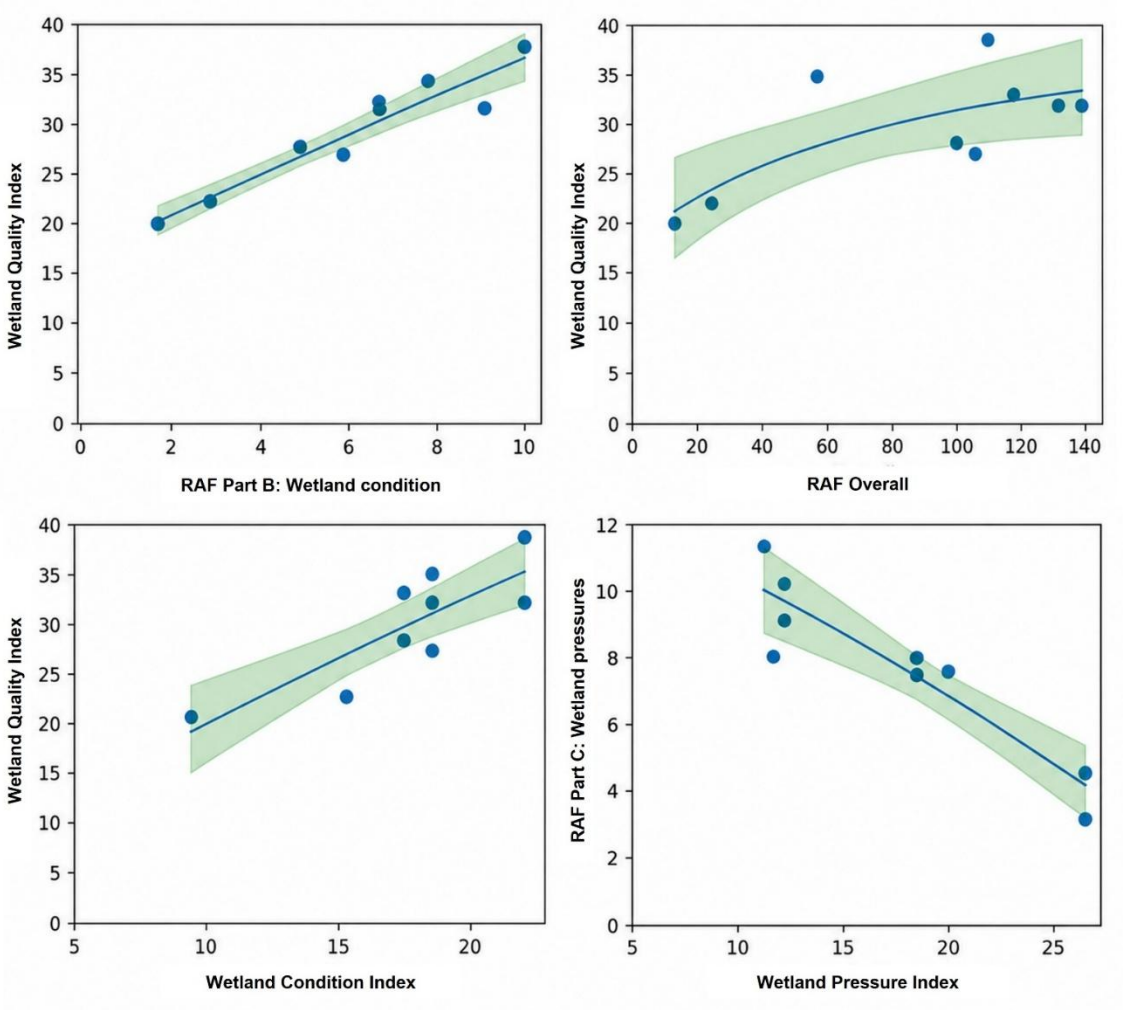
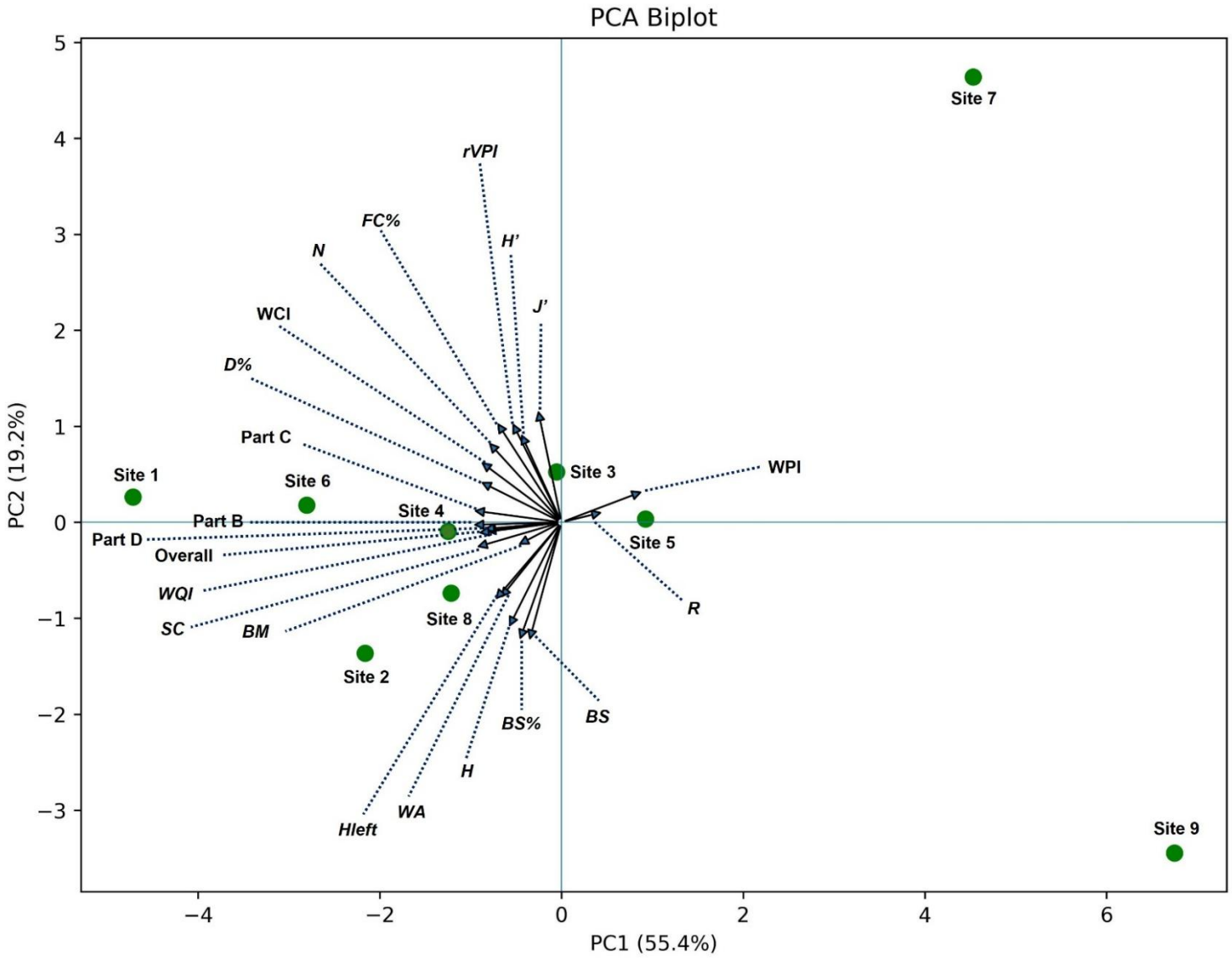
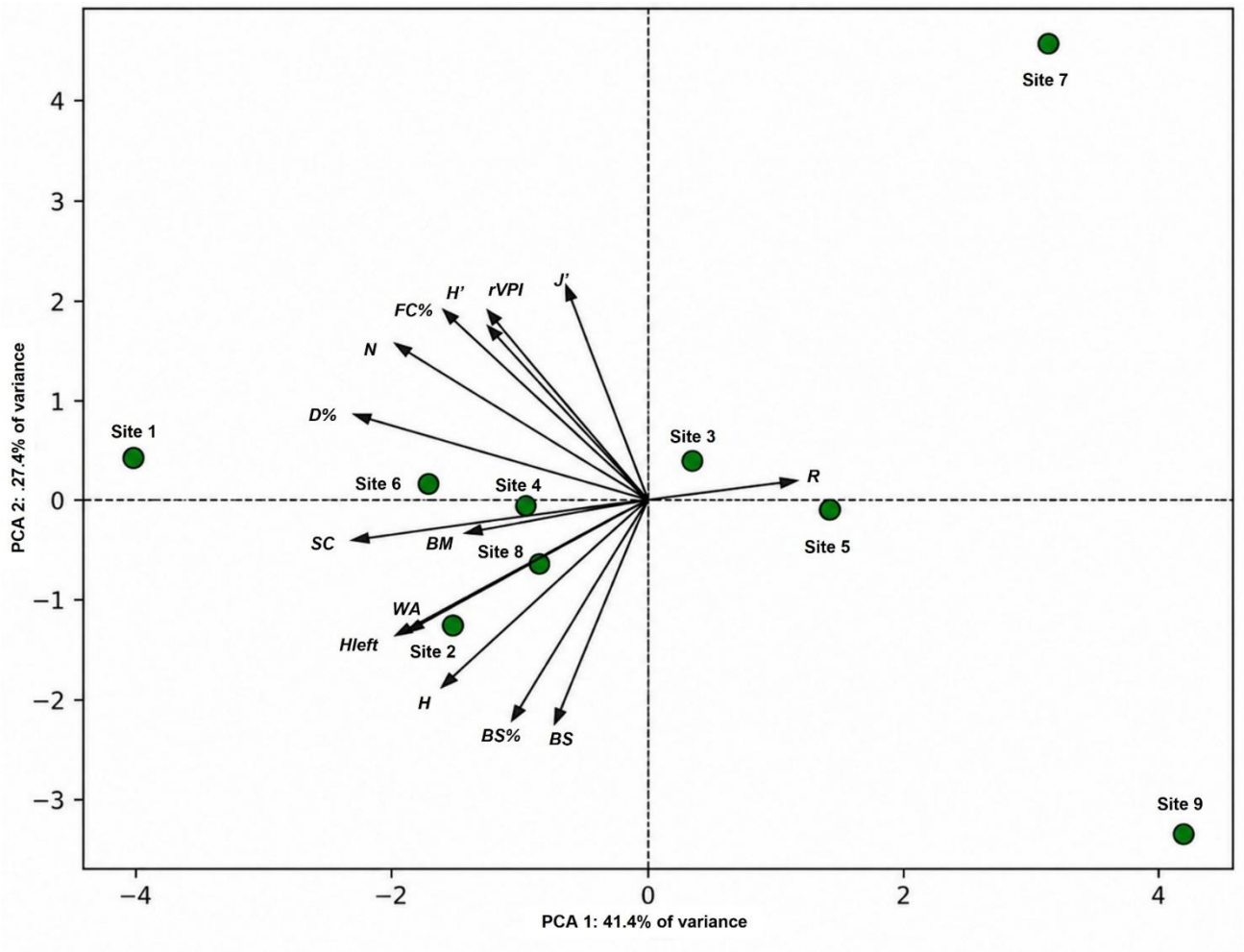


Fig. 3 Relationships between the Wetland Quality Index (*WQI*) and RAF Part B (top left), RAF Overall (top right), and Wetland Condition Index (bottom left), together with the relationship between RAF Part C and the Wetland Pressure Index (bottom right). Lines represent fitted regression models and shaded areas denote 95% confidence intervals.


PCA Biplot
PC2 (19.2%)
PC1 (55.4%)
Site 7
rVPI
FC%
H'
N
J'
WCI
D%
Part C
Site 1
Site 6
Site 3
WPI
Site 4
Part B
Site 5
Part D
Overall
WQI
R
SC
BM
Site 8
Site 2
BS%
BS
H
WA
Hleft
Site 9

391

392 **Fig. 4** Principal components analysis (PCA) ordination of wetland sites based on standardised variables (log transformed) using a correlation matrix. Arrows
393 represent variable loadings, with length indicating the strength of contribution to the ordination axes. All variables contributed to axis construction. WPI = Wetland
394 Pressure Index; *R* = ratio of wetland to buffer margin area (*WQI* variable); *BS* = No. native bird species (*WQI* variable); *BS%* = % native bird species (*WQI*
395 variable); *H* = height of tallest native vegetation (*WQI* variable); *WA* = Wetland area (*WQI* variable); *Hleft* = Hectares left (*WQI* variable); *BM* = Buffer margin
396 area (*WQI* variable); *SC* = No. vegetation structural classes (*WQI* variable); *WQI* = Wetland Quality Index; Overall = RAF score overall; Part D = RAF Part D
397 (management responses); Part B = RAF Part B (wetland state); Part C = RAF Part C (wetland pressures); *D%* = Native flora species dominance (*WQI* variable);
398 WCI = Wetland Condition Index; *N* = No. native flora species (*WQI* variable); *FC%* = % native flora cover (*WQI* variable); *rVPI* = Reversed Vegetation Prevalence
399 Index (*WQI* variable); *H'* = Shannon-Weiner Species Diversity Index (*WQI* variable); *J'* = Pielou Evenness Index (*WQI* variable).
400



401

402 **Fig. 5** Principal components analysis (PCA) ordination of wetland sites based on standardised *WQI* variables (log transformed) using a correlation matrix.
403 Arrows represent variable loadings, with length indicating the strength of contribution to the ordination axes. All variables contributed to axis construction.

404

Discussion

Performance and future development of the wetland RAF

The RAF differentiated wetlands across marked gradients in condition, pressure and management response, indicating that its indicator set is sufficiently sensitive for comparative assessment. Its applicability beyond the wetland types represented here nevertheless requires wider testing. The approach extends the forest framework of Cameron et al. (2025), but uses criteria and scoring calibrated specifically for wetland ecosystems.

While the strong correlation between Part C (wetland pressures) and the WPI and WCI is not particularly surprising given they measure similar attributes (Online Resource 5), the tight coupling between Part C and Part B (as well as the *WQI*) is an important result. For example, the Omamari wetland sites (Sites 1, 2, 3, 4, 5, and 8) assessed in this study were largely Raupō-dominated reedlands with little variance in vegetative structure and all classed within the same hydrological category (swamp), yet the RAF system was still able to pick up subtle nuances in indicators such as relative species dominance between what seem like intuitively similar sites. Parts B and C of the wetland RAF are tightly correlated with measures of indigenous (S-4, S-10, S-11) and exotic vegetation dominance (P-7, P-8, P-9) respectively, corroborating assertions that weed invasion is one of the primary threats to wetlands both in New Zealand and around the world (DeFerrari and Naiman, 1994; Zedler and Kercher, 2004). Exotic weeds can adversely affect multiple wetland functions such as the hydrological and nutrient regimes, as well as altering the balance of food and habitat resources for indigenous fauna (Zedler and Kercher, 2004; Champion and Reeves, 2009). The ripple effect of wetland invasion by weeds can therefore be significant, resulting in the displacement of indigenous plant and animal species.

The variation in Part D scores is an accurate reflection of response relative to effort and need, with wetlands on Pāmu managed farms generally scoring higher due to more substantive environmental management efforts being implemented. Management efforts are particularly apparent in terms of ensuring stock are adequately excluded from wetlands (R-1) and the fencing of wetlands sites to enhance buffer margin size relative to wetland extent (R-8). Pāmu also has a dedicated pest animal control program in place across both Kapiro and Omamari farms for herbivores, omnivores (R-2) and predators (R-3). In contrast, interviews with Department of Conservation (DOC) staff revealed that Site 1 – despite being a large, prominent wetland reserve of significant regional value and ranking #6 on the list of the top 255 freshwater wetlands in Northland (Wildland Consultants 2010) - had no formal management plan in place (R-10), no pest animal control for herbivores undertaken despite the occurrence of feral pigs (R-2), and minimal pest plant control apart from periodic wilding pine control (R-4). This should not be interpreted as a lack of commitment to statutory obligations or management requirements, but rather reflects the constraint imposed by a sustained decline in long-term funding (Predator Free NZ Trust, 2018) and the consequent limitation on the DOC's capacity to deliver conservation outcomes across its mandate.

While trial results indicate that the scope, coverage and suitability of indicators used across all parts of the RAF are adequate, there are areas for further improvement and trialling. For example, while Part C was designed to ensure that most pressure variables contained in both the WPI and WCI were captured, the Von Post Index (as used in the WCI) is currently excluded. The Von Post Index is a semi-quantitative scale (H1–H10) that classifies peat (more typically associated with bogs and fens) according to its degree of decomposition, based on the colour of the water and the quantity and character of organic material released when a peat sample is manually squeezed (Biester et al. 2013). It therefore cannot be used to assign relative condition based on a one-off measurement unless benchmarked to a desired reference level over time, which presents a similar temporal restriction as S-2 (changes to water table relative to the wetlands natural state) and *WTD-SD* used in the *WQI*. However, further iterations could include the Von Post Index as an important indicator of wetland soil health to be tracked and monitored over time.

A practical benefit of the RAF is its capacity to translate assessment results into measurable, site-specific key performance indicators (KPIs). Landowners can use low-scoring condition indicators and their associated pressures to prioritise interventions, estimate implementation costs and track subsequent change. This creates a transparent basis for reporting PES outcomes and evaluating whether financial incentives reflect the cost of restoration and maintenance (Cameron et al. 2025). For example, Sites 7 and 9 could adopt a three-year target to increase RAF Part B by eight points through stock exclusion, indigenous planting and herbivore and weed control, with annual progress assessed using the corresponding vegetation and browse indicators. Site 8 would require a different target because its principal pressures were stock access, possum browse and localised infestations of pampas and Arum lily. Its management plan could therefore prioritise fencing, sustained animal control and targeted weed removal, with progress evaluated against water-quality, browse and vegetation-naturalness scores. These examples demonstrate how the same framework can generate interventions proportionate to the condition and pressures recorded at each wetland.

The RAF is an evolving tool, and further refinement will be required to validate use and applicability across a wider range of hydrological (e.g., ombrotrophic and minerotrophic peatland wetlands) and vegetative (e.g., rushland, sedgeland) typologies to fully represent the diversity of wetlands that characterise New Zealand.

Verification potential and future development of the *WQI*

Results – while preliminary – indicate that the *WQI* provides a useful, accurate, and practical means to capture the desired components of ecosystem quality within a single unit for quantitively assessing the health and condition of wetlands. The *WQI* also provides the ability for comparative analysis between wetlands that fall within the same hydrological and vegetative classification. In this regard, it provides trained wetland ecologists with a unified ‘quality’ metric that has significant interpretive advantages over more common comparative measures such as multiple linear regression analysis (Cameron et al. 2025). Results can then be compared against the qualitative descriptors of the RAF (particularly Part B

and C), which are intended to be used by landowners as monitoring and reporting tools, as well as complement use of the WCI and WPI. Although RAF Part B was assessed by the same observers who conducted the vegetation surveys underpinning most *WQI* indicators, similarly strong correlations are expected from landowner assessments, consistent with results from volunteer stream monitoring (Storey et al. 2016).

Several *WQI* components require further evaluation. Wetland area (WA), buffer area (BM) and their ratio (R) favour larger sites and may undervalue small wetlands that provide connectivity between habitat remnants or protect downstream waters (Ausseil et al. 2011b). Hleft partly accounts for historical loss within each hydrological class, but it does not capture every dimension of ecological importance. Incorporating the occurrence of threatened or locally uncommon species could improve representation of rarity and conservation significance, although doing so would require appropriate weighting to prevent a single record dominating the index. Vegetation richness, diversity, evenness, native dominance and cover remain appropriate measures of condition because they collectively describe community composition and also reveal undesirable exotic dominance (Clarkson et al. 2003; Ortiz-Burgos 2016; Pielou 1966). The limited variation in rVPI was nevertheless valid: even Sites 7 and 9, which were dominated by exotic vegetation, contained a high proportion of obligate or facultative wetland species that receive higher scores under that metric. Testing across additional wetland classes and degradation gradients will be needed to determine whether these variables and their current weightings consistently distinguish ecological condition.

Wetland flora abundance and species composition may change seasonally, particularly in ephemeral wetlands where hydroperiod fluctuates and surface water is transient, which means associated plant assemblages are often temporary and highly dynamic (Johnson and Rogers, 2003; Bedford, 1996). However, they are considered in general to be more consistent, stable indicators of health than the presence or abundance of native birds (*BS*, *BS%*), which may be more seasonal and require repeat temporal measurements to accurately gauge diversity. Thus, the higher weighting accorded in the *WQI* for these indicators, along with *rVPI*, *SC* and *H*, are relevant and prudent considerations. While we were unable to measure and report *WTD-SD* given study timeframe constraints, this variable remains pertinent and should be included in long-term wetland monitoring.

Future versions of the WQI could incorporate benchmarked soil and water-quality measurements, including nitrogen, phosphorus, bulk density, soil carbon, water content and, where standing water is present, microcystins. Interpretation would need to account for natural differences among wetland classes: swamps and marshes receive surface-water and organic-matter inputs, ombrotrophic bogs are naturally nutrient-poor, and fens are influenced by groundwater (Clarkson et al. 2003). Nutrient enrichment can nevertheless alter vegetation, nutrient cycling and algal productivity across these systems (Clarkson et al. 2003; Sorrell 2010). Direct measurement could capture fertiliser inputs as well as nutrients introduced through livestock urine and erosion, pathways that are difficult to quantify through a rapid assessment. Bulk density, moisture and soil carbon may further indicate whether

hydrological conditions sustain the anaerobic processes required by hydrophytic vegetation (Reddy and DeLaune 2008). Microcystins could provide an additional warning of cyanobacterial proliferation in nutrient-enriched wetlands with standing water (US EPA 2016). These measurements would strengthen long-term verification, but they would also increase field, laboratory and interpretation requirements. Variables should therefore be added only where they provide information not already represented by existing indicators and can be evaluated against wetland-type-specific reference conditions, including relevant ANZG (2018) guidelines.

The combined RAF - *WQI* approach addresses a recognised challenge in results-based agri-environmental schemes which requires balancing scientific credibility against the cost of monitoring and usability. International experience indicates that overly complex indicators can discourage participation, whereas simple indicators may inadequately represent biodiversity outcomes (Herzon et al. 2018; Elmiger et al. 2023). The close agreement between RAF Part B and *WQI* scores, on the other hand, demonstrates their complementary potential: the RAF for landowner-led monitoring and the *WQI* for independent ecological verification. Together, they provide a structured means of linking management actions with measurable changes in wetland condition, along with a practical division of monitoring responsibilities. The costs involved, consistency and acceptability to landowners will require further testing across wetland classes, seasons and assessors before either method is adopted for verifying PES outcomes.

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