Bioeconomy Science Institute
Maiangi Taiao

Defining and assessing ecosystem collapse

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Contents

Summary

Project and client

- In April 2026 the Ministry for the Environment (MfE) contracted the Bioeconomy Science Institute to undertake a literature review on the definitions and approaches used to assess ecosystem collapse.
- The review was required to critically evaluate how state and trajectory of change in ecosystem condition are assessed across terrestrial, marine, and freshwater domains.

Objectives

The objectives were to:

- provide a definition of 'ecosystem collapse' based on the published literature
- describe the prerequisites for understanding the trajectory towards, and risk of, ecosystem collapse
- evaluate potential criteria for assessing the risk of ecosystem collapse, and determine their relevance across terrestrial, freshwater, and coastal domains
- identify and evaluate illustrative case studies as examples of how collapse criteria, variables, indicators or measures are used to determine thresholds of ecosystem condition and risk of collapse
- develop an illustrative ecosystem collapse case study for a terrestrial ecosystem using expert input, and assessing data availability, ecosystem collapse risk, and data gaps.

Methods

- We identified key literature on assessing ecosystem collapse risk using standardised, repeatable search terms. Key publications were identified through a structured search of peer-reviewed scientific publications, foundational publications, and reviews from the International Union for the Conservation of Nature (IUCN) Red List of Ecosystems (RLE), and targeted searches for key concepts of ecosystem condition, thresholds, and tipping points.
- To examine how ecosystem collapse risk has been evaluated in practice, we identified case studies that reported on ecosystem collapse or degradation for three domains: terrestrial, coastal and estuarine, and freshwater ecosystems. We determined for each case study which criteria, indicators, and measures were used to determine collapse risk. We extracted information on how each criterion was evaluated, including the thresholds applied, and provide examples of how each criterion was assessed.

Literature review

- Definitions of *ecosystem collapse* are provided from the literature.
- The IUCN definition of ecosystem collapse describes the end point of ecosystem decline, when all occurrences or locations of an ecosystem are considered collapsed throughout the geographical range of an ecosystem, either at a global, sub-global, or national scale. Tipping points, thresholds, and early warning signals are useful ecological concepts derived from a rich body of theory, but the translation of these concepts into practice has been variable, partly due to the complexity of ecosystems themselves, but also to confusion over terminology.

- The RLE framework has been designed to evaluate ecosystem collapse risk using defined criteria and thresholds, but has the flexibility to incorporate threats and thresholds relevant to the ecosystem being assessed.
- To underpin RLE assessments, the IUCN guideline historical baseline year, against which changes in ecosystem condition and extent are evaluated, is 1750. Because of the unique history of human arrival in New Zealand, a baseline date of 1250 also has merit. A consensus on baseline dates to underpin RLE assessments in New Zealand is needed.
- Virtually all (26 out of 27) case studies we reviewed considered several RLE criteria to assess collapse risk. Spatial criteria (A and B) contributed to the overall collapse risk in 24 out of 27 case studies.
- For coastal and estuarine ecosystem case studies, reduced extent was most often used to determine collapse risk. Environmental degradation was highlighted as a major consideration, typically using indicators specific to the ecosystem of interest (e.g. water quality indicators, sedimentation rates). Disruption to biotic processes in coastal or estuarine ecosystems was often evaluated based on trends in native biota or harvest rates.
- For freshwater ecosystems, environmental degradation and restricted distribution most often determined collapse risk. Environmental degradation was a major consideration, and, when assessed, used quantitative indicators specific to the ecosystem (e.g. water flow in streams and rivers, water quality indicators in lakes). Disruption to biotic processes was less often evaluated, but when assessed was usually based on trends in the abundance of native biota.
- For terrestrial ecosystems, reduced and restricted distributions were by far most commonly used to determine collapse risk. Environmental degradation, when assessed, used indicators specific to the ecosystem, often based on previous research (e.g. the effects of climate change, changes to the fire regime). Disruption to biotic processes, when assessed, was based on the adverse effects of weeds, disease, or reduced habitat suitability (e.g. reduced structural complexity).

Conclusions

- A definition of ecosystem collapse is provided that has been assessed and adopted internationally. The definition provides useful overarching principles for assessing ecosystems across marine, freshwater, and terrestrial domains to ensure consistency in approach.
- The RLE framework is the most advanced in practice for appraising ecosystem threat status and collapse. This framework includes defined thresholds determined pragmatically using global expertise for each of four criteria: A (declining distribution), B (reduced distribution), C (degradation of abiotic environment), and D (altered biotic processes and interactions).
- Ecosystem distributions (criteria A and B) were most commonly used to set thresholds for collapse risk across all domains because they are relatively easy to measure and evaluate. These are clearly most informative for ecosystems that have undergone the most extreme reductions in extent, or those that are naturally restricted.
- The indicators used and thresholds set in criteria C (degradation of abiotic environment), and D (altered biotic processes and interactions) were highly ecosystem-specific across the studies, resulting in a broad range of approaches or methods used. Defensible ecosystem-specific thresholds require a detailed understating of the ecosystem in question, and of threats to the ecosystem’s biota or functioning.

- Using the RLE framework to assess the risk of ecosystem collapse has value in New Zealand in all ecosystems across all domains. It is a repeatable, defensible process to identify the ecosystems most at risk of collapse, and this information can inform prioritisation for their protection and restoration.
- Data deficiencies are likely for many ecosystems, which may limit the robustness of initial RLE assessments, but could be used to prioritise data collection or fill knowledge gaps.
- Partnering with mana whenua to include mātauranga Māori and te ao Māori in New Zealand’s RLE implementation would strengthen New Zealand’s approach to RLE assessments and could provide additional knowledge or complementary approaches to understanding condition and trends in ecosystems.

Recommendations

- Commission case studies to evaluate the risk of ecosystem collapse in freshwater and marine ecosystems, including the utility of indicators and collapse thresholds in these systems, and compare and contrast results with those from terrestrial ecosystems.
- Implementing the RLE process requires:
  - continued investment in a unified ecosystem typology and mapping of ecosystems, because these underpin assessments of ecosystem collapse risk
  - achieving a consensus on baseline dates to underpin RLE assessments, with peer review, by evaluating the advantages and disadvantages of using historical baselines of 1250 vs 1750 in determining collapse risk
  - formalising international peer review processes (e.g. drawing on international expertise).

1 Introduction

Humanity's growing need for food, energy, water, and materials is causing a crisis for the Earth's biodiversity and ecosystems, and the services they provide (IPBES 2019). Seventy-five percent of the Earth's land surface is significantly altered, with 66% of the ocean area affected and over 85% of the global area of wetlands lost (IPBES 2019). Global environmental change and increasing human demands on ecosystems have generated widespread concerns about the current state or ongoing degradation of ecosystems and whether these declines are irreversible.

New Zealand mirrors these global trends and concerns. It has lost 71% of its original forest cover since human settlement, and the loss of original forest and regenerating forest continues (Ewers et al. 2006). The rate of loss of wetlands in the last 30 years in one New Zealand region (0.5% of the area per year) matches the global average rate (Robertson et al. 2019). New Zealand's 71 naturally uncommon terrestrial ecosystems provide the major habitat for many endemic plants and animals, even though they only constitute 0.5% of the country's land area (Williams et al. 2007), yet 41% of them have been assessed as either endangered or critically endangered (Holdaway et al. 2012). Currently we do not have a clear understanding of which changes in the state and condition of ecosystems might represent irreversible tipping points or, ultimately, ecosystem collapse.

For New Zealand to prevent the collapse of ecosystems and the services they provide, information is required on the trajectories of change in the extent and condition of its ecosystems across the land, freshwater, and marine domains. Understanding what constitutes ecosystem collapse, and which ecosystems are most at risk of collapse, will identify those most in need of protection or management so that degradation can be averted or reversed. Once an ecosystem has collapsed it will have lost its biotic and abiotic features, and the organisms it contains will be beyond the range of natural spatial and temporal variability they have evolved with (Bland et al. 2017). Collapsed ecosystems are unlikely to be recovered without intensive and expensive restoration efforts (Newton 2021). Although we focus here on the definition of ecosystem *collapse* and its application, understanding *declines* in ecosystem condition should be used to identify and prevent ongoing degradation and collapse.

1.1 Context and scope

This report reviews the definitions, concepts, and approaches related to the assessment of risk of ecosystem collapse, evaluates New Zealand and international examples, and presents relevant case studies. As part of the approach to developing a defensible, evidence-based approach to assessing whether ecosystems are on a trajectory towards collapse, the report also assesses related concepts such as 'thresholds' and 'tipping points', beyond which ecosystem collapse may be inevitable.

The report gives recommendations on the best definitions of 'ecosystem collapse' based on a literature review and case studies, and also provides recommendations for future work. The work presented here is highly relevant and aligned with New Zealand's global obligations to report on the UN Convention on Biodiversity. The Kunming–Montreal Global Biodiversity Framework views use of the International Union for Conservation of Nature (IUCN) Red List of Ecosystems (RLE) as a headline indicator for 2030 Target 1 ('Plan and manage all areas to reduce biodiversity loss') and as providing direct metrics for Targets 2 ('Restore 30% of all degraded ecosystems') and 3 ('Conserve 30% of land, waters and seas'). The process of red listing of ecosystems requires a systematic evaluation following international guidelines of the likelihood of ecosystem collapse.

We evaluate both New Zealand and international evidence from the academic literature, key government publications, and other credible sources of evidence across three domains: marine coastal, freshwater, and terrestrial ecosystems. This evaluation is not a systematic review of the literature, but was designed to rigorously evaluate how ecosystem collapse has been defined and used in practice, based on recent, illustrative exemplar studies or systems.

1.2 Objectives

The objectives were to:

- provide a definition of 'ecosystem collapse' based on the published literature
- describe the prerequisites for understanding the trajectory towards, and risk of, ecosystem collapse
- evaluate potential criteria for assessing the risk of ecosystem collapse, and determine their relevance across terrestrial, freshwater, and coastal domains
- identify and evaluate illustrative case studies as examples of how collapse criteria, variables, indicators or measures are used to determine thresholds of ecosystem condition and risk of collapse
- develop an illustrative ecosystem collapse case study for a New Zealand terrestrial ecosystem using expert input, and assessing data availability, ecosystem collapse risk, and data gaps.

The case study should assess whether current data can be used to assess the risk of ecosystem collapse, and if so, quantify the risk of collapse and identify where there are gaps and challenges. This case study could form a scalable blueprint for assessment in other ecosystems or domains.

2 Methods

2.1 Definitions, concepts, and case studies

To identify relevant information from a large body of literature, we first reviewed a comprehensive text on ecosystem collapse and recovery (Newton 2021), which identified key literature on collapse definitions and related concepts, and we then sourced these directly.

We searched for case studies using Web of Science for publications, using the search phrase ("ecosystem collapse risk" OR "risk of ecosystem collapse"), and the search phrase ("ecosystem collapse" OR "collapse risk") (the latter for studies published since 2021 only). These searches returned 35 and 499 records, respectively. We refined the latter by selecting only records for the Web of Science categories environmental science, ecology, biodiversity conservation, environmental studies, marine freshwater biology, and forestry. We excluded records focused on socioeconomic issues, prehistoric ecosystem collapse, review articles (many of which we had already identified), and geological studies. Across these searches, after removing duplicates and irrelevant records, 37 publications were identified. We then assessed the relevance of selected articles to ecosystem collapse risk assessment, and, where possible, extracted information on indicators or measures used to infer ecosystem collapse or collapse risk.

Because the search above identified many irrelevant records that did not explicitly evaluate ecosystem collapse risk, we also searched for further examples that used the RLE framework.

These included relevant articles published in the special *Austral Ecology* edition (Volume 40, Issue 4, 2015), ‘Ecosystem Risk Assessment’. Ten case studies for individual ecosystems were published in this edition; we excluded one on the basis that it was dissimilar to New Zealand ecosystems (hot desert woodland).

We searched for additional examples using Google Scholar and Web of Science, for publications since 2015, using the search phrase “("IUCN Red List of Ecosystems" OR "RLE") AND ("ecosystem collapse" OR "collapse risk")”. This search returned 420 and 37 records, respectively. Records included review papers, especially on the overall RLE approach and its relevance to specific issues (e.g. climate change, restoration and management, policy). We reviewed the first 50 publications and identified 14 case studies that we had not already included and several review articles. We excluded two publications for coral ecosystems (which we deemed less relevant to New Zealand), two publications that summarised assessments for entire countries or continents (Africa and South America), and one that we could not access via our institutional subscription.

To boost the number of case studies for freshwater ecosystems, we searched using the phrase “("IUCN Red List of Ecosystems" OR "RLE") AND ("ecosystem collapse" OR "collapse risk") AND (“freshwater”)”, which returned 256 results. We identified no additional freshwater case studies in a review of the top 100 results, so we selected four case studies from the founding document describing the RLE framework (Keith et al. 2013), and one from a compilation of assessments of ecosystems from Myanmar (Murray et al. 2020).

Few of the case studies identified in the searches above had quantitatively assessed collapse risk (i.e. using RLE criterion E). We searched the IUCN Database to identify additional examples and added one (‘Meso-American Reef’, included in Bland et al. 2017). We also searched on the IUCN Database for additional case studies for ecosystems considered universally collapsed, and added a study on European oyster ecosystems (zu Ermgassen et al. 2024).

To search for RLE case studies completed for New Zealand ecosystems, we searched using the phrase “("IUCN Red List of Ecosystems" OR "RLE") AND ("New Zealand")”, which revealed an assessment for New Zealand mangrove ecosystems (McNeill et al. 2024).

We expect there to be a selection bias in our approach: many ecosystem collapse risk assessments have been undertaken for ecosystems worldwide, and given the scope above we considered only a subset of case studies derived primarily from the peer-reviewed published literature (>90%) rather than the grey literature, and most were undertaken as RLE assessments. Our case studies also favoured publications that considered a single ecosystem. Our search terms (above) also probably missed the body of theoretical and empirical research considering regime shifts, particularly in aquatic ecosystems.

The RLE highlights specific criteria formulated to capture important aspects of ecosystem extent and condition that are considered indicative of collapse risk, and we based our literature summaries on these criteria. For each case study we determined which aspects were determining factors for the Red List outcome and summarised findings across coastal and marine, freshwater, and terrestrial domains. We grouped ecosystems with the IUCN Global Ecosystem Typology (GET) realm ‘Freshwater/Marine’ with the freshwater domain and ‘Marine’, ‘Marine/Freshwater/Terrestrial’ and ‘Marine/Terrestrial’ with coastal or estuarine.

For literature that did not follow the RLE framework, we assessed whether the indicators or measures used aligned with any of the RLE criteria. Where ecosystem collapse risk had been explicitly assessed (i.e. as in RLE case studies), we determined which of the criteria were informative of the overall outcome and summarised these findings for each domain. We extracted information on which indicators or measures were used to evaluate collapse risk, including which thresholds were applied (where available).

2.2 Terrestrial case study: Mackenzie District widespread terrestrial ecosystems

A more detailed case study of ecosystem trajectories of change and collapse was developed to illustrate the approach and process involved, and to reveal what steps are required in practice for evaluating RLE. Based on our expertise in terrestrial ecosystems, we selected vegetation of the Mackenzie District for this exercise.

Our approach was to, first, identify and critically evaluate the primary published literature on the vegetation history and condition of the Mackenzie District, eastern South Island (Appendix 2). This district was selected because of the large body of research describing three ecosystems that occur there and the availability of evidence for historical change in these ecosystems. This information allows for qualitative evaluation of the factors driving changes in the state and condition of ecosystems from discrete historical baselines to provide a preliminary assessment of collapse risk. We identified three widespread ecosystems of interest: natural (native) forest, tall tussock grassland, and degraded short tussock grassland. These ecosystems align with typologies at Levels 6–7 of the GET, once those typologies are developed (McCarthy & Wiser 2024).

For each ecosystem we provide information on historical and current ecosystem extent, abiotic threats, and relevant biotic processes or disruptions that could inform future quantitative analyses to estimate the probability of collapse of the three ecosystems. A complete quantitative assessment of ecosystem trajectories or collapse was not feasible and was out of scope for this project, but we use this case study to identify the quality and sources of information and data that are required.

Summary: Multiple search techniques were used to distil the literature on ecosystem collapse risk and the IUCN Red List framework to identify a subset of the literature and case studies most relevant to ecosystem collapse risk definitions and assessment.

3 Ecosystem collapse: conceptual definitions

Although the term ‘ecosystem collapse‘ has been in use for decades, attempts at a precise definition are relatively recent. Lindenmeyer and Sato (2018) emphasise that ecosystem collapse is irreversible, or at least very difficult to reverse, as well as being both widespread and undesirable in terms of the erosion of ecosystem services, ecosystem processes or biodiversity. The IUCN has since formalised a definition of ecosystem collapse as:

> a transformation of identity, a loss of defining features, and/or replacement by a different ecosystem. An ecosystem is collapsed when all occurrences lose defining biotic or abiotic features, no longer sustain the characteristic native biota, and have moved outside their natural range of spatial and temporal variability in composition, structure and/or function (IUCN 2024, p 17).

This definition describes the end point of ecosystem decline, when *all* occurrences or locations of an ecosystem are considered collapsed throughout its geographical range at a defined spatial scale (e.g. regional/national). By considering the state of an ecosystem across its range, this definition allows ecosystem collapse risk assessments to inform national-scale environmental management or conservation priorities.

Ecosystem collapse can also be considered at a local scale – the scale at which conservation management decisions and interventions are often made. The observation that across the distribution of an ecosystem there may be examples spanning the full spectrum of condition and trajectory towards collapse (Bergstrom et al. 2021) has prompted an alternative definition of collapse for use at a local scale (Newton et al. 2021):

> A **degraded ecosystem state** that results from the abrupt decline and loss of biodiversity, ecosystem functions and/or services, where these losses are both substantial and persistent, such that they cannot fully recover unaided within decadal timescales.

This definition emphasises that collapse cannot easily be reversed, whereas the IUCN definition does not consider reversibility.

The most widespread definitions used for collapse are summarised in Table 1; many of these have emerged from development of the RLE framework. Most definitions require knowledge of ecosystem ‘identity’, including defining features and characteristic biota (e.g. Bland et al. 2017; IUCN 2024). Explicit definition and description of what an ecosystem constitutes is thus an important first requirement when considering collapse risk. In common across all definitions is an emphasis on ecosystem condition or state – specifically the maintenance of biodiversity, functions or processes.

Table 1. Definitions of ecosystem collapse in the literature

Definition	Source
An abrupt and undesirable change in ecosystem state.	Lindenmayer et al. 2016
Major changes in ecosystem conditions are widespread and are either irreversible or very time- and energy-consuming to reverse.	Lindenmeyer & Sato 2018
A theoretical threshold, beyond which an ecosystem no longer sustains most of its characteristic biota or no longer sustains the abundance of biota that have a key role in ecosystem organisation (e.g. trophic or structural dominants, unique functional groups, ecosystem engineers, etc.). Collapse has occurred when all occurrences of an ecosystem have moved outside the natural range of spatial and temporal variability in composition, structure and function. Some or many of the pre-collapse elements of the system may remain within a collapsed ecosystem, but their abundances may differ and they may be organised and interact in different ways with a new set of operating rules.	Keith et al. 2013
A transformation of identity, a loss of defining features and a replacement by a different ecosystem type. An ecosystem is collapsed when all occurrences lose defining biotic or abiotic features, no longer sustain the characteristic native biota, and have moved outside their natural range of spatial and temporal variability in composition, structure and/or function.	Bland et al. 2017
A transition beyond a bounded threshold in one or more indicators that define the identity and natural variability of the ecosystem. Collapse involves a transformation of identity, loss of defining features, and/or replacement by a novel ecosystem. It occurs when all ecosystem occurrences (i.e. patches) lose defining biotic or abiotic features, and characteristic native biota are no longer sustained.	Bland et al. 2018
A **change from a baseline state** beyond the point where an ecosystem has lost key defining features and functions, and is characterised by declining spatial extent, increased environmental degradation, decreases in, or loss of, key species, disruption of biotic processes, and ultimately loss of ecosystem services and functions.	Bergstrom et al. 2021
A **degraded ecosystem state** that results from the abrupt decline and loss of biodiversity, ecosystem functions and/or services, where these losses are both substantial and persistent, such that they cannot fully recover unaided within decadal timescales.	Newton et al. 2021
A **transformation of identity**, a loss of defining features, and/or replacement by a different ecosystem. An ecosystem is collapsed when all occurrences lose defining biotic or abiotic features, no longer sustain the characteristic native biota, and have moved outside their natural range of spatial and temporal variability in composition, structure and/or function.	IUCN 2024: follows Bland et al 2017, though does not require replacement by a different ecosystem type.

Source: adapted from Newton 2021 and Newton et al. 2021

Summary: The IUCN RLE ecosystem collapse definition describes the end point of ecosystem decline, when all occurrences or locations of an ecosystem have lost defining features, throughout the geographical range of an ecosystem, either at a global, sub-global, or national scale. Ecosystem collapse can also be defined at a local scale.

4 Tipping points, thresholds, and early warning signals

The theoretical basis of tipping points and ecological thresholds is found in long-established ecological resilience theory (see Holling 1973; Gunderson et al. 2012). Resilience theory considers the capacity of a system to absorb disturbance and reorganise while undergoing change in a way that maintains critical functions, structures, and feedbacks. In practice, resilience can be either helpful (i.e. by maintaining an ecosystem state so that it does not cross a threshold) or unhelpful (i.e. by maintaining an ecosystem in a degraded state; Standish et al. 2014).

Here we don't attempt to synthesise this body of theory or the large literature on complex systems that underpins the concepts of tipping points, thresholds, and early warning signals. However, we do note the following.

- Ecosystems are complex adaptive systems in which interactions between both biological and physical components can generate feedbacks and emergent properties or behaviours.
- Emergent properties of ecosystems can include tipping points, thresholds, critical transitions or catastrophic regime shifts. These terms are often used imprecisely in the literature, and can arise from multiple phenomena, sometimes creating confusion in their application and interpretation (e.g. van Nes et al. 2016).
- Resilience theory has been extended to consider social–ecological systems, processes and feedbacks (e.g. see Folke 2006; Cote & Nightengale 2012), but we do not evaluate social or cultural resilience or collapse in social–ecological systems further here.
- Global interest in the state of ecosystems, and the reliance of human well-being on them, has driven concern about ecosystem changes to avoid catastrophic degradation; these broad concerns are considered out of scope to discuss in detail here, but highlight the growing need to understand and manage ecosystems generally.

Below we briefly introduce some key concepts or terms (in bold) and how we adopt and apply them in this report. A fuller treatment of these concepts would require additional literature review and synthesis.

Tipping points and ecological thresholds, although grounded in technical theory, are widely used to describe ecological changes or trajectories in ecosystem condition and functioning. Tipping points (van Nes et al. 2016; Yletyinen et al. 2019) are a common feature of complex systems, occurring when an ecosystem crosses a critical threshold (see below), characterised by an abrupt ('catastrophic') shift to an **alternative (stable) state** (Scheffer et al. 2001). Put another way, tipping points are often considered as rapid, sometimes unexpected, transitions where small additional changes in environmental conditions, threats or system drivers produce disproportionately large or abrupt changes to an alternative state: called variously regime shifts, state shifts or catastrophic changes (Scheffer et al. 2009; Munson et al. 2018).

Tipping points can also result from critical changes in the ways the components of the system interact with each other (Walker et al. 2012). In theory, ecological tipping points represent the shift between stable states due either to changes in variables (external drivers) or to parameters (conditions of the complex system itself; Beisner et al. 2003). Applying tipping point theory to ecosystem management is challenging, because 'tipping point' can describe a variety of distinct phenomena (Thrush et al. 2009; Milkoreit et al. 2018). Moreover, the behaviour of ecosystems is far more diverse and complex than can be captured with the bifurcation models or simulations, or ball-and-saucer illustrations, typically used to demonstrate tipping points conceptually (Bowman et al. 2015).

Examples of real-world ecosystem tipping points include lakes ‘flipping’ from a high to low condition due to eutrophication, and the sudden collapse of animal populations that have been under gradual stress (Scheffer et al. 2001, Lindenmayer et al. 2005). Tipping points have been documented in many ecological systems (e.g. Lindenmayer & Luck 2005) but have seldom been systematically examined across different systems (but see Rocha et al. 2015; Benton et al. 2017). Although anthropogenic drivers are not required for biophysical tipping points, many tipping points or other non-linear dynamics are triggered by human action (Lade et al. 2013; Selkoe et al. 2015).

Critical thresholds are boundary points or conditions beyond which an ecosystem shifts from one state, structure, or functional regime to another (Groffman et al. 2006; Bergstrom et al. 2021). Thresholds do not necessarily imply a transition to a collapsed state but represent one or more critical points where state changes (either positive or negative), or a shift between alternative stable states, can occur. In the RLE framework (see below), the term ‘threshold’ has specific uses and meaning (see section 5.2.3). Thresholds are central to the measurement of resilience. Direct experimental data on thresholds are difficult to obtain in most cases, but observational data from ecosystems in different states of degradation can be used to direct management decisions and priorities.

Thresholds and tipping points underpin how trajectories of change occur for ecosystem condition, functions and, in some cases, collapse. Four distinct profiles of ecosystem collapse are recognised by Bergstrom et al. (2021) based on ecological data for 19 ecosystems worldwide: abrupt, smooth, stepped, and fluctuating (Figure 1):

- **Abrupt collapse** occurs when an ecosystem crosses a critical threshold and undergoes rapid changes within a short time scale. Abrupt collapse profiles are often associated with abiotic pressures. For example, abrupt mass dieback of mangroves in northern Australia was associated with a temporary 20 cm drop in sea level brought on by a severe El Niño weather pattern (Duke et al. 2017). Abrupt collapse may be particularly challenging to address through management, because there may be little warning and limited opportunity for intervention.
- **Smooth collapse** implies a gradual, progressive degradation over an extended period that may not demonstrate clear tipping points or thresholds. This trajectory may be detected through monitoring but may still be difficult to reverse if degradation has accumulated over decades. Smooth collapse has been demonstrated across multiple ecosystems as the result of long-term abiotic changes (e.g. changes in precipitation or temperature; Bergstrom et al. 2021).
- **Stepped collapse** may occur via a series of discrete transitions, often triggered by pulse disturbances (e.g. flooding, fire, cyclones) superimposed on systems with chronic pressures (e.g. habitat loss, invasive species; Bergstrom et al. 2021). For example, Burns et al (2015) demonstrate stepped collapse of mountain ash forests through intermittent logging.
- **Fluctuating collapse** involves oscillations between degraded and partially recovered states, which may reflect variable environmental conditions or management interventions. It was suggested by Bergstrom et al. (2021) that where fluctuating collapse patterns occur there may be windows of opportunity within which to optimise management outcomes.

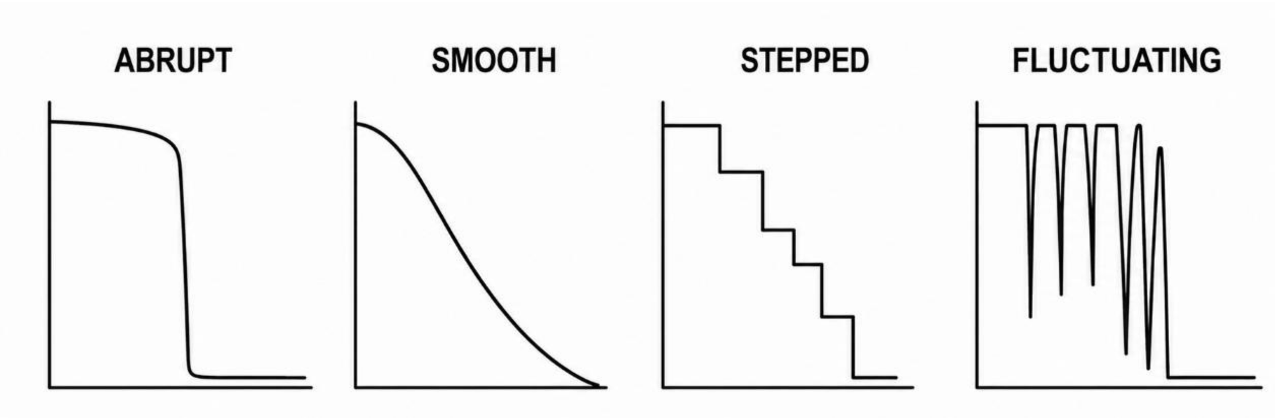

Figure 1. Four collapse profiles. The y axis represents a desirable indicatior specific to an ecosystem (e.g. percentage cover of native species, number of functional groups represented, population size of a keystone species, spatial extent of an ecosystem); the x axis is time. (Source: Bergstrom et a. 2021)

These profiles or trajectories of change reflect different underlying mechanisms or drivers, and these can differ both within and among ecosystem types. Regardless of the processes involved, the profiles have important implications for detection, prediction or management. For example, more than one collapse profile or threshold for an ecosystem may be relevant if there is variability in responses to external drivers. It may be difficult to set discrete quantitative thresholds for an indicator variable when it follows a smooth collapse profile. What this emphasises is a need to predict or anticipate change, and to quantify trajectories with sufficient sensitivity to avoid collapse. A final consideration is whether thresholds result in hysteresis, thought of as a breaking point where the system switches to an alternative stable state that is difficult to reverse (Litzow & Hunsicker 2016).

Early warning signals are measurable statistical or ecological indicators that provide advance notice of declining resilience and an increased likelihood that a system is approaching a tipping point or critical transition that may include ecosystem collapse. Recent reviews of attempts to predict regime shifts, tipping points, and thresholds suggest only modest progress for a few leading indicators and systems.

Early indicators can include increased variance, spatial or temporal autocorrelation, and slowing recovery from disturbance (Scheffer et al. 2009; Moore 2018). In general, system-level changes are well characterised in theory and models but more difficult to demonstrate empirically (Dakos et al. 2024). For example, real-world tipping points are demonstrated for some systems such as lakes, in which relatively high-resolution data for abiotic conditions and trophic interactions are sometimes available. For freshwater lakes, increased variability in key trophic interactions, sometimes termed ‘flickering’, can presage rapid transitions to food web topologies and ecosystem processes. Far fewer examples of tipping points are known for terrestrial ecosystems; fire traps created by positive feedbacks between fire regimes and invasive grasses are one of the best examples (e.g. D’Antonio & Vitousek 1992; Fusco et al. 2021).

Early warning signals, thresholds, and tipping points are sometimes difficult to determine or quantify with confidence (Hemraj & Carstensen 2025). Robustly quantifying thresholds or tipping points for specific drivers or interventions is complicated because collapse can result from interactions among multiple drivers or pressures acting in combination that typify many ecosystems (Bergstrom et al. 2021; Newton 2021).

Sato and Lindenmeyer (2018) reviewed the practicalities of predicting ecosystem collapse using early warning indicators (including in the context of RLE criterion E, which requires a quantitative prediction of ecosystem collapse risk).They concluded that while there is evidence to suggest that numerically predicting collapse is possible, early warning indicators do not yet predict collapse reliably, and they need refinement before they can be practically applied. As a consequence, ecosystem collapse is more commonly recognised retrospectively rather than being robustly predicted in advance or detected while the transition is still occurring (Newton 2021).

Despite the complexity involved in quantifying ecosystem collapse, the application of resilience theory, tipping points, thresholds, and early warning signals holds promise to understand ecosystem trajectories of change and intervene before collapse. However, putting these concepts into practice is an area of active research, debate, and development (Thrush et al. 2016). For example, functional redundancy and response diversity show promise as proxy measures of resilience and are amenable to management, such as maintaining assemblages of both rare and common species that should have a diversity of responses to environmental changes or threats (Standish et al. 2014). Another emerging idea that has bearing in practice is that connectivity and scale contribute to resilience, and both can be improved through management. For example, dispersal limitation is the primary driver of catastrophic shifts resulting from environmental change in trophic models, suggesting that spatial structure is an important consideration for rapid ecosystem change (Arumugam et al. 2024).

Ultimately, understanding and preventing ecosystem collapse will depend on the quality of data available to evaluate ecosystem condition and change – either degradation or recovery – and the effectiveness of interventions. Overall, the use of collapse profiles (Figure 1) can be thought of in practice as a way of summarising data and models to identify thresholds of concern, and to guide evidence-based responses or interventions to prevent ecosystem collapse.

Summary: The identification of tipping points, thresholds, and early warning signals, and their application to management interventions, is an active area of research and development. Collapse profiles can be used to conceptualise and convey trajectories of change in ecosystems.

5 Frameworks for evaluating ecosystem collapse risk

Research on ecosystem collapse is undertaken using approaches ranging from purely theoretical (which may include models based on empirical data) to observational (which aim to predict ecosystem collapse in real-world scenarios) (Table 2). Across both empirical and theoretical approaches, ecosystem collapse appears to be system-specific, creating inconsistencies in both the definition and use of the term (e.g. Boitoni et al. 2015). Moreover, there are few studies that clearly define ecosystem collapse, even among studies focused on prediction (e.g. Sato & Lindenmayer 2018). To address this challenge, multiple approaches are needed, combining ecosystem-specific research (e.g. to identify thresholds of collapse and an end point where collapse is considered to have occurred) and ecological monitoring (e.g. to understand ecosystem condition and trajectories of change against a baseline).

Prospective ecological monitoring frameworks for understanding ecosystem condition include essential biodiversity variables (Proença et al. 2017), assessment of ecological integrity, ecosystem health, ecosystem services (including IPBES; Díaz et al. 2015), and conceptual frameworks such as socio-ecological systems or multilayer network modelling (Young et al. 2006; Pilosof et al. 2017). Each of these frameworks or approaches can contribute to our understanding of ecosystem condition and, with temporal data, the trajectories of change in ecosystems. Using such frameworks can contribute the data and knowledge needed to understand the major drivers, dynamics, and stressors of ecosystems, as well as emergent ecosystem behaviours (see section 4).

The RLE is a framework that can draw on these multiple sources of information or data to evaluate the risk of ecosystem collapse. Empirical research, whether experimental or observational, can provide key data and models to inform the selection of indicators and quantitative thresholds for RLE assessments. Ecological monitoring data are required to assess the state of ecosystems relative to these thresholds. The robustness and quality of RLE evaluations depend on both comprehensive knowledge of the system being considered and adequate data on the threats to and condition of ecosystems.

New Zealand has multiple monitoring programmes across ecosystems that can inform ecosystem risk assessments. For example, the NZ Biodiversity Assessment Framework provides eight national outcome objectives, which collectively are intended to give a comprehensive overview of the state of ecological integrity across New Zealand's ecosystems (McGlone et al. 2020). A stocktake of existing knowledge of a range of environmental attributes for New Zealand showed that, across domains, the state of knowledge for many was rated as 'good' or 'excellent', though for many attributes data were incomplete (Lohrer et al. 2024). There may generally be a poorer understanding of ecosystem extent in the marine and coastal domain than for freshwater and terrestrial domains due to greater difficulty sampling estuarine and coastal areas (Townsend et al. 2018).

Across all domains, the data needed to assess the severity of threats, environmental degradation, and disruptions to biotic processes is patchy among sites and ecosystem types, making assessment of collapse risk across the entire extent of an ecosystem difficult. However, standard indicators and measures defined in New Zealand's Outcome Monitoring Framework (Department of Conservation 2024) could provide many of the data required to assess these attributes.

Table 2. Frameworks for evaluating ecosystem collapse risk

Approach	Key features of approach
Theoretical ecosystem collapse research	Conceptual, mathematical or simulation methods that may be parameterised with real data May inform appropriate indicators and critical thresholds for empirical research
Empirical ecosystem collapse research	Includes observational, temporal and experimental research (4). Often no clear ecosystem description (1) Often conducted for a specific location (rather than entire ecosystem extent) Ecosystem collapse is often implied, but not clearly conceptualised or defined (1) Experimental tests of theory are rare (1) Predictions of ecosystem collapse are seldom made in time to prevent collapse (1) May provide data on relevant indicators and critical thresholds Multivariate analyses frequently used (2) Predicting collapse is more feasible in aquatic than terrestrial ecosystems (1, 4)
Ecological or environmental monitoring frameworks	Ideally systematically implemented, with agreed and consistent standards (5) Provide temporal and spatial data on trends in ecological integrity, environmental degradation, or other attributes (5)
Red List of Ecosystem assessments (3)	Detailed ecosystem descriptions required Conceptualisations required of what constitutes ecosystem collapse Considers entire extent of ecosystem Ecosystem conditions compared to reference states (baselines) Collapse risk assigned based on qualitative and/or quantitative thresholds Multivariate analyses not recommended (2, 3)

References: 1: Sato & Lindenmeyer 2018; 2: Rowland et al. 2018; 3: IUCN 2024; 4: Newton 2021; 5: McGlone et al. 2020.

5.1 The RLE framework for assessing ecosystem collapse risk

The RLE framework is currently the only formal operational assessment protocol designed to enable ecosystem collapse risk to be assigned to ecosystems. In this framework, ecosystems are assessed against the following five criteria (IUCN 2024):

A: reduction in geographic extent
B: restricted geographic distribution
C: environmental (abiotic) degradation
D: disruption of biotic processes or interactions
E: quantitative analysis that estimates the probability of ecosystem collapse.

In this framework each criterion, A–E, uses decision thresholds against which to judge changes in the extent or severity of degradation (refer to IUCN 2024 for full descriptions). For criteria C–E, indicators, thresholds, and methods are tailored to the specific ecosystem.

There can be multiple end points to a degraded ecosystem and not necessarily a consensus on what is desirable or undesirable (Newton 2021). It is therefore important to consider both what the desirable state of an ecosystem would be (which may be captured, to some degree, by conceptualising the baseline ecosystem condition or extent) as well as potential undesirable states. Undesirable states of ecosystems are partially captured in the RLE assessments as the ‘collapse description’ (section 5.2.3), which is specific to the ecosystem in question, describing what collapse would look like in that ecosystem (IUCN 2024).

The RLE definition of ecosystem collapse and the overall approach have been subject to criticism. Boitoni et al. (2015) highlighted areas of ambiguity, which include difficulty defining ecosystems consistently in the first place. The collapse definition and criteria applied are criticised as being vague; specifically, because the RLE framework requires ecosystem-specific attributes or indicators to be used when defining collapse, they argue that it will be difficult to compare risk levels assigned to different ecosystems (Boitoni et al. 2015). Indeed, it is generally considered that trade-offs are necessary to achieve the generality and flexibility required of a globally applicable risk assessment protocol (Keith et al. 2013).

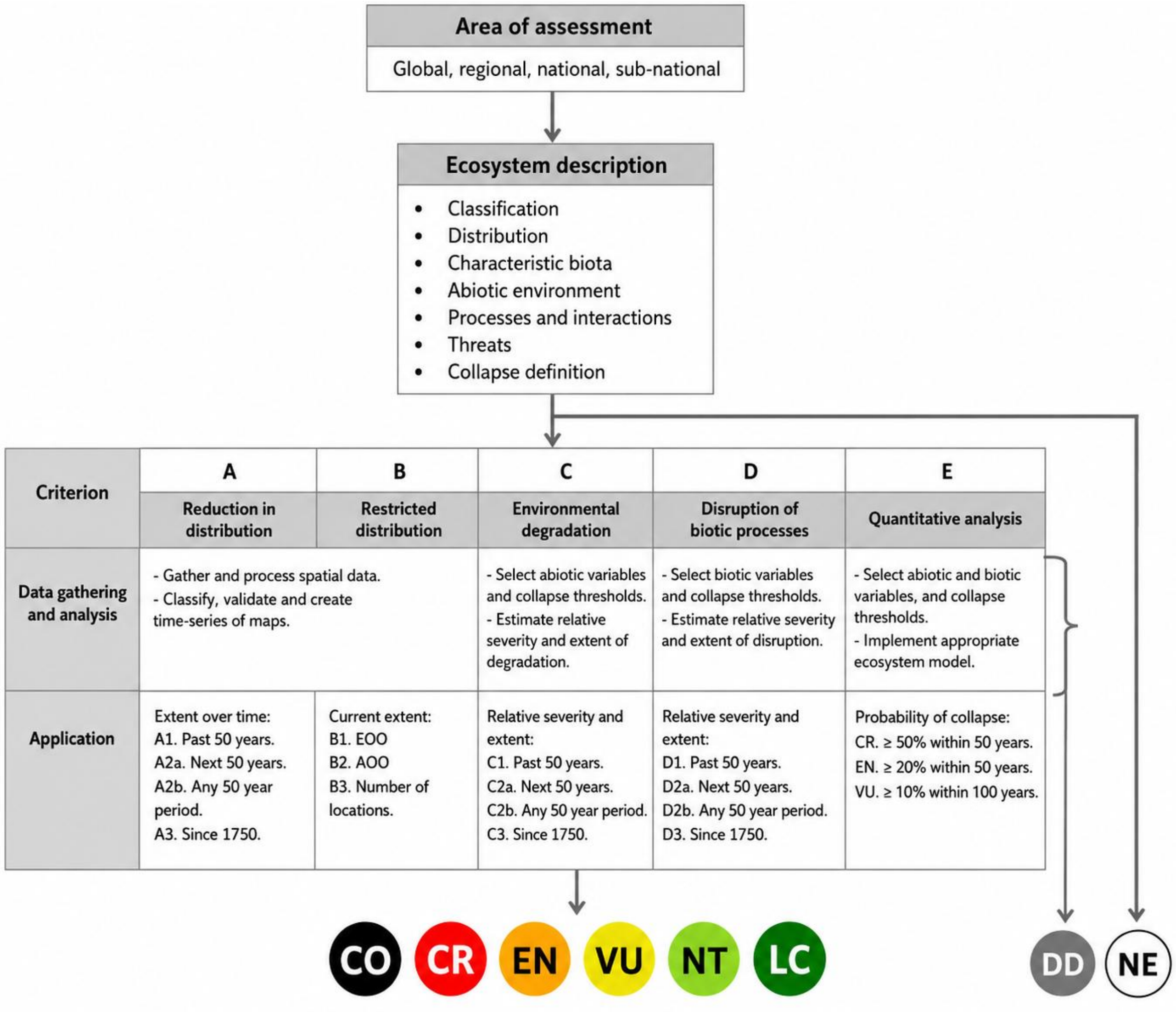

Figure 2. RLE framework, including the ecosystem description (see prerequisites for assessing ecosystem collapse risk, section 5.2), criteria, and sub-criteria. An ecosystem collapse risk is assigned to each criterion (and sub-criterion) which determine the overall Red List outcome.

Notes: AOO = area of occupancy; EOO = extent of occurrence. Ecosystem collapse risk (Red list outcome) categories are ‘collapsed’ (CO), ‘critically endangered’ (CR), ‘endangered’ (EN), ‘vulnerable’ (VU), ‘not threatened’ (NT) and ‘least concern’ (LC). No assignment is made for criteria deemed data deficient (DD), or otherwise not evaluated (NE).

Summary: The RLE framework has been designed to evaluate ecosystem collapse risk using defined criteria and thresholds, but with the flexibility to incorporate threats and thresholds relevant to the ecosystem being assessed.

5.2 Prerequisites for evaluating ecosystem collapse risk

Definitions of ecosystem collapse (section 3) emphasise understanding and describing the characteristics of ecosystems themselves, so this is an important first step before assessing ecosystem collapse risk (e.g. 'Ecosystem description', Figure 2). The ecosystem description includes information on the ecosystem classification, distribution, characteristic biota, and abiotic environment and threats (IUCN 2024).

5.2.1 Ecosystem classification and distribution

For RLE assessments, ecosystem assessment units must be cross-referenced to the IUCN GET.[1] This is a hierarchical classification with six levels; RLE assessments may be carried out on ecosystem units defined at a range of thematic scales but are recommended for units classified in Levels 4–6.

Ecosystem classifications (or typologies) provide a framework within which to define the dominant biota of ecosystems and, if successfully mapped, will contribute to understanding ecosystem distribution. Where assessments are to be conducted across an entire ecosystem extent, it is important that ecosystems be well described and that their extent can be determined. Continued work to develop national unified typologies will provide the foundation for RLE assessments at national and regional scales.

New Zealand has multiple marine and estuarine classifications that could be used to define ecosystems. Although these may have their limitations (Lundquist et al. 2024), they will inform the development of future marine typologies. New Zealand marine classifications have been primarily based on abiotic features (e.g. water depth, topography, substrate type), with some consideration of benthic habitat but limited consideration of pelagic components (Lundquist 2024). Spatial proxies for ecosystem distribution based on abiotic features may be used to describe the extent (criterion A or B) of an ecosystem in RLE assessments, but only if justified on the basis that they validly represent the distribution of ecosystem biota (IUCN 2024).

Marine typologies need to have both benthic and pelagic components (Lundquist et al. 2024; Sprague & Wiser 2024). Statistical classifications have been developed to provide proxies for deepwater ecosystems based on environmental characteristics (the Seafloor Community Classification, Stephenson et al. 2022), because limited sampling had been done for most of New Zealand's deep ocean. A hierarchical classification has also been developed for coastal hydrosystems and estuaries (Hume et al. 2016), and classifications are available for seamounts and rocky reefs (Shears & Babcock 2004; Rowden et al. 2005; see also Rowden et al. 2018).

A unified, ecologically grounded typology for New Zealand's freshwater ecosystems needs to be developed and tested. It will build on existing statistical classifications, such as the River Environment Classification (Snelder & Biggs 2002), the Freshwater Environments of New Zealand database (Leathwick et al. 2010), and the NZ Rivers Map interactive mapping tool (Whitehead & Booker 2019), although each of these has been criticised for their inability to distinguish waterway ecosystems (Collins Consulting 2025).

Terrestrial typologies used in New Zealand currently include an expert-based system (Singers & Rogers 2014) and a quantitative, plot-based system (Wiser & De Cáceres 2018), as well as a

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[1] (https://global-ecosystems.org/).

typology for terrestrial naturally uncommon ecosystems (Williams et al. 2007). Work is underway to unify terrestrial typologies and produce a workable typology suitable for national and international reporting. For terrestrial ecosystems this will provide a catalogue of units at the vegetation association scale, Level 8 of the International Vegetation Classification, and at the finest unit (Level 6) of the IUCN GET (Faber-Langendoen et al. 2025), which could be used for RLE assessments in the future.

5.2.2 Ecosystem description

The process of RLE assessment first requires an ecosystem description, which should ideally be linked to a formal ecosystem typology. The description includes documentation of characteristic biota and their abundance, key interactions, and relevance to ecosystem processes, dynamics, and functions (IUCN 2024). It includes a description of any abiotic influences on ecosystem distribution or function that explain an ecosystem's natural range of variability and ability to sustain characteristic native biota.

Often, dominant or indicator species for an ecosystem form part of the description, and are used to distinguish between ecosystem types in both qualitative and quantitative topologies. This ecosystem description includes a conceptual model that describes key interactions between biota, and between biota and the environment, along with a summary of the major threats (past, present and future), their severity, extent and trends (IUCN 2024).

5.2.3 Ecosystem-specific conceptualisation of collapse

Once an ecosystem has been clearly defined, a **collapse description** is developed, based on the information summarised above and the conceptual model (IUCN 2024):

> If multiple states of collapse are possible (e.g. due to different threats), all of these should be described with a similar level of detail. **Collapse thresholds** for the application of criteria A and B are typically defined as 100% loss of spatial distribution of the ecosystem type. The use of alternative thresholds of collapse for criterion A or B must be thoroughly justified. Collapse thresholds for the application of criteria C, D and E should be identified as part of the assessment of those criteria. Examples of locally collapsed occurrences of the ecosystem type help support descriptions of collapsed states. (IUCN 2024)

The term 'threshold' is used in RLE assessments in two ways. First, criteria have decision thresholds that are common across assessments (e.g. for criteria A and B). Second, for criteria C–E, quantitative indicators and collapse thresholds *specific to the ecosystem* may be specified. It is from the ecosystem and collapse descriptions that key indicators of ecosystem condition are identified that might inform its position along a trajectory towards collapse. Criteria C and D include two components to be evaluated for each subcriterion (severity and extent). The relative severity is the proportional change in a chosen indicator, scaled between two values: one describing the initial state of the system (0%), and one describing a collapsed state (100%; the collapse threshold); the extent is the degree to which the pressure occurs across the ecosystems distribution (IUCN 2024).

5.2.4 Defined baseline for assessing change

It is important to define reference states (baselines) against which the magnitude of change to an ecosystem can be evaluated and interpreted. Baselines are also essential for setting meaningful targets for restoration, management or ecosystem recovery. Baselines are most often considered in the context of historical baselines; that is, baselines that describe ecosystem condition and extent at some point in time before human impacts intensified. Setting historical baselines can counter the phenomenon of ‘shifting baselines’ (Pauly 1995), which is the tendency for people at any given time to regard as ‘normal’ the state of an ecosystem as they first encounter it. Accordingly, successive generations of people regard the same ecosystem as ‘normal’ even as it degrades, with attendant underestimation of the severity of biodiversity loss, changes in ecosystem structure and function, and collapse risk (Sáenz-Arroyo et al. 2005; Lotze & Worm 2009).

The RLE framework uses baselines to assess recent (past 50 years) and historical change in ecosystems to evaluate whether there has been a reduced extent (criterion A), environmental degradation (criterion C), or a disruption of biotic processes (criterion D). In the standard RLE approach, historical change is assessed from a baseline year of 1750 (Figure 2), which is intended to capture trends since the onset of industrialisation (when pressures on ecosystems intensified) to the present day (IUCN 2024). In practice, evidence of historical ecosystem extent and condition is usually incomplete, so RLE case studies draw on a range of evidence, often qualitative rather than quantitative.

Some assessments of ecosystems in New Zealand have used 1250 as a baseline against which to assess changes in state and condition; for example, across New Zealand’s forests (Ewers et al. 2006) and palustrine and inland saline wetlands (Ausseil et al. 2011). The case for using a 1250 baseline is that it represents a state with no direct human influence (i.e. before first human settlement in c. 1280; Wilmshurst et al. 2008), and that the composition and extent of ecosystems in 1250 is possible to estimate based on multiple sources of data, such as fossil records, palaeontological evidence, or species distribution models (e.g. for forests, Leathwick 2001). New Zealand is not unique in its relatively recent arrival of humans: other island nations such as Hawaii and Iceland were settled about the same time, and Cabo Verde, Mauritius, and La Réunion about 300 years later. So a 1250 baseline may be more appropriate in some cases, but this is a departure from the current international guideline of a 1750 baseline.

It may also be appropriate to consider baselines in the context of ecosystem recovery (*sensu* Newton 2021). The standards to assess ecosystem recovery are being developed by the IUCN.[2] Large-scale extinctions of terrestrial vertebrates and invertebrates occurred between 1250 and 1750, and others occurred after 1750 (e.g. Gill 1991). The choice of baselines will also determine how ecosystem recovery is assessed or achieved.

Summary: Ecosystem collapse risk assessments require clear ecosystem descriptions (including a robust ecosystem typology at GET Levels 4–6), and ecosystem-specific collapse descriptions. A historical reference baseline is needed to understand changes in ecosystem condition and extent. The IUCN guideline historical baseline year is 1750, but a baseline of 1250 also has merit for New Zealand ecosystems. A consensus is needed on baseline dates, for all ecosystems, to underpin New Zealand ecosystem collapse risk assessments.

[2] https://iucn.org/resources/conservation-tool/iucn-green-list-protected-and-conserved-areas.

6 Criteria for ecosystem collapse risk assessment

This section is based on our literature search on ecosystem collapse risk for coastal and estuarine, freshwater, and terrestrial domains. Most case studies that explicitly evaluate ecosystem collapse risk, and include all elements in section 5.2, are RLE assessments. For each domain the following summary is structured using the main criteria for RLE assessments (i.e. reduced and restricted distribution, environmental degradation, and disruption to biotic processes).

6.1 Coastal and estuarine ecosystems

The drivers of ecosystem collapse in coastal and estuarine ecosystems were reviewed by Jackson (2008), who concluded that the 'cumulative effects of exploitation, habitat destruction, and pollution are more severe in estuaries and coastal seas than anywhere else in the ocean except for coral reefs'. The principal drivers of ecosystem collapse identified included habitat destruction, overfishing, introduced species, ocean warming, acidification, toxins, and eutrophication *(*Jackson 2008).

Historical baselines for coastal and estuarine ecosystems across Europe, North America, and Australia were reconstructed using palaeontological, archaeological, historical, and ecological data (Lotze et al. 2006). Trends were analysed across seven cultural periods, reflecting the stages of cultural and market development rather than calendar dates (Lotze et al. 2006). This work incorporated many attributes of the RLE framework, including assessment of long-term trends in biotic characteristics (e.g. changes in species assemblages and extinctions) and evaluation of abiotic pressures.

Few coastal or marine studies in the ecological literature (aside from RLE studies) identified by our searches explicitly considered ecosystem collapse risk, although collapse was implied based on degradation due to abiotic pressures (e.g. Pérez-Ruzafa et al. 2026*)* and biotic disruption. One study provided an assessment that c. 75% of coastal wetlands of Louisiana would plausibly be drowned by 2070 (Li et al. 2024), inferring collapse, but in this case the ecosystems were not described in any detail.

One experimental study suggested that ecosystem collapse will not result from more frequent marine heatwaves, but that the relative frequency of co-occurring species might change (Bass & Falkenberg 2023). Another study used multiple indicators and multivariate approaches to determine ecosystem condition and the combined effects of different stressors; for example, interactions between insect browsing and eutrophication on saltmarsh dieback (Tomasula et al. 2023).

6.1.1 Reduced and restricted distribution

A reduced or restricted spatial distribution plays a major role in coastal and estuarine ecosystem collapse risk, and arises mostly through past habitat loss, although some ecosystems have a naturally restricted distribution (e.g. estuaries, coastal lagoons; Williams et al. 2007). For ecosystems that have reduced in extent, habitat loss is most often the result of direct anthropogenic effects such as land-use change (Clark et al. 2015; Murray et al. 2015) or harvest (e.g. Gillies et al. 2020; zu Ermgassen et al. 2024).

Reduced distribution was assessed in all RLE case studies, and together with restricted distribution these spatial criteria contributed to Red List outcome in all cases and were the sole determinant in two. For ecosystems with dominant biota that lends itself to direct mapping (e.g. mangroves), case studies typically mapped the current ecosystem extent and often quantitatively determined recent changes. Data were typically derived from existing typology maps, aerial photographs, and/or satellite imagery.

Where detailed spatial data are unavailable proxies can be used (IUCN 2024), and this approach is useful for assessing historical ecosystem extent. For European oyster reef ecosystems, an impressive array of historical accounts, mostly between 1800 and 1930, were gathered and used to assess historical change (zu Ermgassen et al. 2024). These data were then used to establish, for different locations, whether there is high or low confidence that oysters were historically present in sufficient abundance to be reef-forming.

Likewise, no maps were available to assess either the past or current extent of oyster reefs of southern and eastern Australia (Gillies et al. 2020). This study used historical first-person accounts, place names, cultural histories, archaeological excavations (aboriginal middens), sediment cores, government fisheries reports, commercial surveys, and sediment cores. Very few of the accounts provided information on ecosystem distribution within a known location, so reduction in ecosystem extent was assessed using current presence/absence of oyster reefs at each recorded historical location.

In ecosystems where direct mapping is not possible, models can sometimes be appropriate to predict ecosystem extent. For example the distribution of suitable light conditions for under-ice marine benthic invertebrate communities in Antarctica was predicted using bathymetric data and remote-sensing data on sea-ice concentration (Clark et al. 2015).

Among New Zealand coastal or estuarine ecosystems, estuaries and lagoons are considered naturally uncommon and have restricted distributions (Williams et al. 2007). The ecological functioning of New Zealand estuaries and lagoons is thought to be declining, and has been considered vulnerable and endangered, respectively (Holdaway et al. 2012; note that an earlier iteration of RLE criteria was used). The mangrove ecosystem of New Zealand (Level 4 unit of the IUCN GET) was assigned to the category 'Least Concern' based on its increased extent since 1970 (McNeill et al. 2024). Reduced or restricted distributions are likely to be relevant for inferring the collapse risk of other New Zealand biogenic habitat (seagrass, kelp forests), although there may be limitations in the existing spatial data (Anderson et al. 2019).

Table 3. Summary of ecosystem collapse criteria for the coastal and estuarine domain

Criterion	Threats identified	Indicators or measures	Relevance
Reduced distribution	Loss of habitat (1, 2, 3, 4, 5)	Historical reduction in extent (since 1750) (2, 13) Recent reduction in extent (last 50 years) (1) Future reduction in extent (next 50 years) (11)	**Major consideration** – driven by abiotic and biotic threats Always assessed in RLE examples (9/10 cases). Almost always determined the Red List outcome (8/9 cases) Recent change– mostly quantified Historical change – mostly qualitatively assessed, often data deficient
Restricted distribution	Some ecosystems may be naturally restricted; e.g. estuaries and lagoons (6), mangroves (7)	Area of occupancy and extent of occurrence (12, 13)	**Minor importance** Always assessed in RLE examples (10/10 cases) Often determined the Red List outcome (6/10 cases) Assessed quantitatively
Environmental degradation	Eutrophication and pollution (2, 8*, 9*, 10) Sedimentation rate / erosion (5, 11) Land-use change / coastal development (3, 10, 12) Substrate simplification (2) Climate change (3, 7, 10, 11, 12, 13) Salinity changes (1, 5) Changes to water flow (1) Episodic flooding / hurricanes (2, 3) Deforestation (mangroves, 7)	Dissolved oxygen (below 2 mg/L significant mortality of fauna is expected in seagrass beds; 13) Nitrogen concentration (>0.026 mg/L ammoniacal or 0.025 mg/L oxidised will cause mortality of seagrass, 1) Nitrogen:phosphorus balance (no thresholds were specified; 17*) Secchi depth (13) Sediment discharge (collapse certain if sediment discharge declines to zero; 10) Mapped land use / land-use change or shoreline modification as a proxy for sediment load (>50% land intensively used will determine collapse of oyster reef; 2) Sea-level rise (1.8 m rise will cause 100% collapse of saltmarshes 13; 75% of coastal wetlands predicted to drown by 2070, 19*) Expansion of sea urchin range due to sea warming, leading to kelp overgrazing (14*) Responses of seagrass species to marine heatwaves (16*) Change in sea-ice coverage (qualitative; 12) Coral cover / bleaching rate (3)	**Major consideration** Often assessed in RLE examples (7/10 cases) Sometimes determined the Red List outcome (4/10 cases) Quantitative thresholds often set
Disruption of biotic processes or	Harvest (2, 5, 10, 13) Invasive species (10, 13)	Historical/long-term trends in oyster abundance (using harvest records showing >80% decline; 2)	**Major consideration** Often assessed in RLE examples (6/10 cases)

Criterion	Threats identified	Indicators or measures	Relevance
interactions	Decline in abundance of native biota (5, 7, 10, 13) Trophic diversity alteration/disruption (5, 14*) Disease/pathogens (2, 15)	Catch per effort for key species (13) Shifts in spatial vegetation patterns (extent of mangroves encroaching into seagrass habitat; 13) Trends in key wildlife species – dugong and sea turtles, 13; migratory species, 10; multiple species associated with oyster reefs, 5; 30–80 species across multiple taxonomic groups and ecological guilds, 8* Compositional change (8*) Trophic modification (shift in oystercatcher diet; 5) Reduced structural complexity (5) Urchin density (>2.2 m^2 leads to urchin barren, 14*) Size of grazed patches on macrophyte-dominated substrates to estimate urchin effects (18*)	Often determined the Red List outcome (5/10 cases) Qualitative thresholds most often used (e.g. 80% decline in abundance of indicator species confers 'Critically Endangered' status)
Quantitative assessment of ecosystem collapse risk	Models predicting impacts of abiotic and biotic threats on coral reefs (3)	Stochastic model to compare scenarios using interactions between functional groups and hurricane disturbance (3)	**Seldom possible** – one case study

1: Keith et al. 2013; 2: Gillies et al. 2020; 3: Bland et al. 2017; 4: Mahoney & Bishop 2017; 5: zu Ermgassen et al. 2024; 6: Holdaway et al. 2012; 7: Marshall et al. 2018; 8*: Lotze et al. 2006; 9*: Jackson 2008; 10: Murray et al. 2015; 11: McNeill et al. 2024; 12: Clark et al. 2015; 13: Sievers et al. 2020; 14*: Ling & Keane 2024; 15*: Camp et al. 2015; 16*: Bass & Falkenberg 2023; 17*: Pérez-Ruzafa et al. 2026; 18*: Minguito-Frutos et al. 2025; 19*: Li et al. 2024.

* Indicates not an RLE case study.

6.1.2 Environmental degradation

Environmental degradation is a major threat to coastal and estuarine ecosystems, reducing their capacity to function as nutrient and sediment sinks and provide coastal protection. Eutrophication has been claimed to be the main source of degradation in coastal systems worldwide since the 1950s, especially in estuaries (Jackson 2008). Water eutrophication indicators (e.g. sediment, nitrogen, and phosphorus load) increased consistently across all the ecosystems considered by Lotze et al. (2006), who described widespread declines in seagrass meadows and other coastal wetlands across Europe, North America, and Australia due to reclamation, habitat destruction, and eutrophication.

Assessment of environmental degradation is typically quantitative, using indicators and thresholds specific to the ecosystem (Table 3), and in RLE examples is often based on previous research. Likewise, collapse thresholds, where specified, are specific to the study ecosystem and typically use quantitative data (Table 3). Examples include limits on sediment discharge (e.g. where sedimentation is crucial for maintaining the tidal flats ecosystem for the Yellow Sea), and collapse was considered likely if sediment discharge declined to zero due to land-use change (Murray et al 2015).

Several indicators and thresholds were used for seagrass communities, depending on the study. At Moreton Bay previous research identified a collapse threshold for dissolved oxygen levels of 2.0 mg/L; below this level significant mortality and sub-lethal responses of animals were expected (Sievers et al. 2020). In South Australia a collapse threshold for annual mean ammoniacal (0.026 mg/L) and oxidised (0.025 mg/L) nitrogen concentration was used; higher levels were predicted to cause high or total mortality of seagrass (Keith et al. 2013).

Because data for indicators of environmental degradation may be unavailable, proxies or indirect indicators are sometimes used. For the oyster reef ecosystem of southern and eastern Australia, indicators to assess environmental degradation included the relative severity of adjacent catchment modification, which was estimated using percentage of land-use change as a proxy for sedimentation. Likewise, in this study the extent of estuary shoreline modification was used as a proxy for substrate simplification (Gillies et al. 2020).

Climate change is a pervasive threat to coastal ecosystems and may exert direct or indirect effects. For saltmarshes at Moreton Bay, tidal inundation was identified as the key threat and a 1.8 m rise in sea level was predicted to lead to a 98% reduction in saltmarsh extent, considered close enough to 0% extent remaining to use 1.8 m as a collapse threshold (Sievers et al. 2020). Climate change is indirectly triggering collapse in kelp ecosystems as sea warming enables poleward range expansion of sea urchins, causing a catastrophic phase shift from a kelp-dominated to an urchin-dominated state, with projections suggesting that half of kelp beds in the region affected will collapse by 2030 (Ling & Keane 2024).

Environmental degradation in New Zealand estuarine and coastal ecosystems is likely to be an important consideration in RLE assessments. Data and models for the ecosystem-level consequences of sedimentation rates in estuaries are likely to exist for New Zealand (e.g. Thrush et al. 2003), but we were not able to undertake a systematic review.

6.1.3 Disruption of biotic processes or interactions

In coastal and marine ecosystems, trophic interactions are important for ecosystem stability and functioning, and when food webs are disrupted through harvest it can trigger cascading effects that contribute to ecosystem collapse. In the review by Lotze et al. (2006), exploitation through harvesting was considered the most significant driver of declines in native biota in coastal and estuarine ecosystems. Temporal changes were assessed for 30 to 80 species across each ecosystem, spanning major taxonomic groups and ecological guilds (Lotze et al. 2016). Despite the wide geographical distribution and unique historical context for the ecosystems included, all showed similar trajectories; specifically, a long, slow period of decline, followed by a rapid acceleration in decline in populations of most native biota over the last 150–300 years (Lotze et al 2006).

Across all ecosystems considered, the most commercially desirable species were depleted first, which were replaced by smaller, less economically valuable species. Among fish species the earliest declines occurred in diadromous species (salmon, sturgeon) that were both easily fished and highly desired, followed by large pelagic fish (tuna, sharks, cod, halibut) and small pelagic fish (herring, sardines; Lotze et al 2006). Across multiple ecosystem case studies in Lotze et al. 2006, habitat-forming filter-feeders such as oysters suffered substantial declines through dredging, which reduced the habitat complexity of estuaries and coastal waters.

Multiple drivers of collapse can reinforce each other through positive feedbacks; for example, fishing, habitat destruction, introduced species, and eutrophication (Lotze et al. 2006; Jackson 2008). For example, oysters have been nearly eliminated in many locations by overfishing, but recruitment and survival are later constrained by eutrophication, disease, introduced species, and formerly uncommon predators that were previously kept in check by now-overfished species (e.g. Camp et al. 2015; Jackson 2008).

The literature highlights that the disruption of biotic processes is a major consideration for coastal and estuarine ecosystems, but this is perhaps not obvious from the small number of RLE case studies we reviewed: only six of ten case studies assessed this criterion, which was only informative in four cases (Table 3). Indicators and collapse thresholds were specific to the study ecosystem. In practice a range of semi-quantitative and qualitative indicators and thresholds were used.

For the oyster reef ecosystem of southern and eastern Australia, harvest records were used to estimate the extent of decline in oyster reefs and assess historical change. Although loss could not be quantified for current oyster reef localities, the case was made that the relative severity of oyster biomass decline was more than 90%, and the extent of threat was more than 90% of past extent, classifying the ecosystem as ‘Critically Endangered’ with a medium degree of confidence (Gillies et al. 2020). Several other examples cited declining abundance of key ecosystem biota as a contributor to ecosystem Red List outcome; for example, in sea turtles (Sievers et al. 2020) and migratory birds (Murray et al. 2015).

In the European native oyster reefs described earlier, oysters now occur mostly as scattered individuals and there are no known locations where oysters persist at reef-forming densities (zu Ermgassen et al. 2024). Oysters were a keystone species, and overharvesting eliminated the reef structure and its associated biodiversity. To assess change, comparisons were made between species recorded in historical data and recent samples: eight species historically associated with oyster reefs were notable for their rarity or complete absence in the modern samples, while some other species increased. Also based on historical observations, it was noted that Eurasian oystercatchers in Europe now predominantly feed on mussels, clams, and worms, as opposed to oysters (zu Ermgassen et al. 2024).

For New Zealand, the current state of knowledge of biogenic habitats has been reviewed by Anderson et al (2019) and Lohrer et al. (2024). Large-scale historical losses of biogenic habitats have been documented, although some examples may be local. It is not clear whether there are examples of ecosystem collapse across an entire ecosystem range (as per the RLE definition of collapse), but ecosystem collapse has been inferred at a localised scale. Kelp forests may be subject to collapse across parts of their range (e.g. Spyksma 2025), and kelp forests provide three-dimensional canopy cover to a wide range of flora and fauna, including invertebrates and fish. Loss of kelp forests in New Zealand is associated with dramatic shifts in ecological states and a loss of functioning and integrity (Tait & D'Archino 2024). Removal of predatory fish and crayfish causes a trophic cascade, as urchins increase in density, affect the kelp forest, and form an 'urchin barren'; up to 30% of fished reef habitats in northeastern New Zealand are degraded and now dominated by urchin barrens (Tait & D'Archino 2024).

For kelp forests, tipping points have been suggested for key parameters; for example, urchin densities above two urchins per square metre were considered sufficient to turn kelp forests into urchin barrens, though fewer urchins may be required to maintain barrens (Ling & Keane 2024; Tait & D'Archino 2024). Light intensity less than 1 mol/m^2/d is also a key limit for healthy and productive kelp forest ecosystems (Tait & D'Archino 2024). The best current evidence suggests that kelp forests are not currently collapsed across their entire range and may be in good condition at some localities (Anderson et al. 2019).

Another New Zealand example described as ecosystem collapse is provided by historical overharvesting of mussels causing the local collapse of once-extensive beds (e.g. green-lipped mussels at Firth of Thames, Kenepuru Sounds, and Okiwa Harbour; horse mussel beds in outer Marlborough Sounds). These collapses, initiated by harvesting, also coincided with declines in water quality and increased sedimentation due to coastal development and changing land use (Anderson et al. 2019).

Biotic processes and species assemblages can also be disrupted by biological invasions, though none were mentioned in the case studies we reviewed. Non-native *Caulerpa* spp. (*Caulerpa brachypus* and *Caulerpa parvifolia,* aquatic weeds) have been detected and are now spreading in New Zealand coastal areas (e.g. Aotea/Great Barrier Island; Lam-Gordillo & Keeler 2025). Experience overseas indicates that invasive *Caulerpa* spp. can form extensive intertidal and subtidal meadows, sometimes extending to depths of c. 50 m, with potential to alter benthic habitats and displace indigenous biodiversity (Williams 2007).

6.1.4 Quantitative assessment of ecosystem collapse risk

We identified one case study for a marine environment that quantitatively assessed the risk of ecosystem collapse risk. Although coral reefs do not occur in New Zealand waters, we include this example here to demonstrate the approach.

The Mesoamerican coral reef is considered to be at high risk from mass bleaching, ocean acidification, hurricanes, pollution, and fishing (Bland et al. 2017). It was considered that the reef would almost certainly be collapsed when future live coral cover declined to less than 1% throughout the mapped ecosystem distribution, as estimated using expected rates of mass bleaching, aragonite saturation, and hurricane frequency.

Fish are key components of the reef ecosystem, so as well as a threshold for coral cover (above), separate collapse thresholds for herbivorous fish and piscivorous fish were defined: 5 g/m^2 for herbivorous and 2 g/m^2 for piscivorous. To assess the probability of ecosystem collapse (criterion E), a stochastic model was used to backcast past ecosystem dynamics and forecast future dynamics under 11 scenarios of threat. Scenarios tested showed a wide range of collapse probabilities across time frames and indicators, but four out of seven likely scenarios led to an assessment as 'Endangered' based on coral cover in the next 50 years (Bland et al. 2017). Overall, however, the ecosystem was categorised as 'Critically Endangered' based on assessments of criteria C and D (Bland et al. 2017).

Summary: For coastal and estuarine ecosystems, all case studies we reviewed used more than one criterion to assess ecosystem collapse risk. Reduced extent most often determined the Red List outcome. Environmental degradation was highlighted by case studies as a major consideration. When assessed, environmental degradation used indicators and thresholds specific to the ecosystem, often based on previous research. Disruption to biotic processes in coastal or estuarine ecosystems was typically evaluated based on trends in native biota or harvest rates.

6.2 Freshwater ecosystems

Freshwater ecosystems have been heavily altered by anthropogenic activity worldwide. Major threats include the overexploitation of species, pollution, fragmentation, degradation of habitat, flow modification, water extraction, and invasive species (Geist 2011).

6.2.1 Reduced and restricted distribution

A reduced or restricted ecosystem extent can be a contributing factor in ecosystem collapse in freshwater ecosystems, but its importance is variable and ecosystem-specific. It is likely to be most informative and useful for identifying ecosystems affected by anthropogenic activity and/or conversion to other ecosystem types (Table 4). Where an ecosystems extent is largely geographically defined (e.g. lakes, rivers, streams), reduced extent is less likely to drive risk assessment, unless extreme habitat modification or hydrological changes have occurred.

Unsurprisingly, restricted extent (Criterion B) is a critical factor determining the potential collapse risk of ecosystems that are naturally restricted (e.g. New Zealand's Naturally Uncommon Ecosystems; Holdaway et al. 2012), but also those ecosystems that have had large reductions in extent. One or both of reduced ecosystem extent and restricted extent contributed to the overall Red List outcome in virtually all the cases we reviewed (Table 4).

Most RLE case studies quantitatively estimated current ecosystem extent and sometimes recent changes. Data were derived from existing ecosystem maps, historical or recent aerial photographs, and/or satellite imagery (e.g. Keith et al. 2013; Murray et al. 2015). Vegetation or habitat surveys can also contribute (Keith et al. 2013). Some studies relied on qualitative assessments where no quantitative data were available. Unsurprisingly, qualitative assessments were most prevalent when assessing historical changes.

Reduced extent in the RLE studies we reviewed was primarily driven by environmental degradation (see next section); for example, through land-use change. The raised bogs of Germany have declined in extent by 98% since 1750, and by 58% within 50 years, when assessed against estimates of prior extent from the literature (i.e. rather than mapped areas; Keith et al. 2013).

Restricted ecosystem extent for the Coorong lagoons and Murray Mouth inverse estuary determined the Red List outcome of 'Critically Endangered': total extent of occurrence was less than 2,000 km^2. Furthermore, the ecosystem occurs at only one location, satisfying the criterion thresholds for restricted extent (Keith et al. 2013). At the less concerning end of collapse risk, connected wetlands of the Lake Eyre Basin have a status of 'Least Concern'. Flooding data for wetlands across the Lake Eyre Basin over 30 years (1983–2012) detected no significant reduction in wetland area, and the wetland area exceeded thresholds in extent and area of occurrence (Pisanu et al. 2015).

The threat status of several naturally uncommon ecosystems was evaluated using an earlier iteration of the RLE framework (Holdaway et al. 2012). These ecosystems have naturally restricted distributions. Tarns, although naturally restricted, were not considered to be threatened, based on the fact that no immediate environmental or biotic threats were identified (Holdaway et al. 2012).

6.2.2 Environmental degradation

Environmental degradation is a major consideration in freshwater ecosystems, which often face multiple pressures, including water extraction and hydrological changes, eutrophication and pollution, climate change, and mining (Table 4). Almost all case studies quantitatively assessed environmental degradation, which often determined the overall ecosystem Red List outcome. Most often, quantitative data were available for just one or a few indicators for each ecosystem. Examples of thresholds identified for specific indicators are provided in Table 4.

For Karst Rising Springs wetlands, in South Australia, recent environmental degradation was assessed using trends in the daily spring discharge rate, with a collapse threshold assumed to be 30–38 megalitres per day based on earlier research. There were insufficient data to assess trends in discharge across all locations, but a model was used to extrapolate trends for the entire ecosystem (Keith et al. 2013). Predictions of sea-level rise and saltwater intrusion resulting from climate change are expected to affect most of the remaining ecosystem by 2070 (Keith et al. 2013).

At Chesapeake Bay, eastern USA, previous research informed two indicators that were used to set thresholds: changes in water clarity (represented by a median Secchi depth during the local growing season of April to November, which is less than 0.82 m in regions where salinity is less than 5% and 1.10 m where salinity is more than 5%), and dissolved oxygen in bottom waters (less than 2 mg O_2/L; Mahoney & Bishop 2017).

The influence of future environmental degradation can be assessed using models. For Coorong lagoons and Murray Mouth inverse estuary, South Australia, a climate projection estimated that the ecosystem will undergo an environmental degradation of 100% severity over more than 80% of its extent in the next 50 years; this would cause climate variability, and barrage volume and salinity are expected to exceed thresholds; for example, salinity thresholds were set based on previous work (117 g/L for an upper limit and 100 g/L for a lower limit, beyond which various biota would be affected; Keith et al. 2013). Models were used to determine future climate suitability of coastal sandstone upland swamps in New South Wales. The severity of environmental degradation was based on the reduction in ecosystem extent from the present day, and the collapse threshold was when suitability reaches zero across the entire area (Keith et al. 2013).

One of the most extreme examples of environmental degradation is provided by the collapse of the Aral Sea on the border of Kazakhstan and Uzbekistan, once the fourth largest lake in the world. The major threat was water extraction from the mid-1900s to support irrigated cotton production. The sea dried up throughout most of its area, fragmented into several lakes, and salinity increased.

The hydrological changes precipitated a dramatic loss of invertebrate fauna and fish. Reedbeds of the shorelines disappeared, and woodlands and reed beds of the contributing river deltas contracted, leading to a major loss of mammals and avifauna dependent on the vegetation (Keith et al. 2013). This ecosystem is one of few assessed using the RLE framework that is deemed to have completely collapsed (but see Micklin et al. 2020)

The impact of land use on the degradation of New Zealand waterways is well documented (Ministry for the Environment & Stats NZ 2026), so assessment of environmental degradation is likely to be a major consideration in RLE assessments of New Zealand freshwater ecosystems.

Table 4. Summary of ecosystem collapse criteria for the freshwater domain

Criterion	Threats identified	Indicators or measures	Relevance
Reduced distribution	Driven by environmental degradation, agriculture, residential development (1) Important consideration for some wetlands (1,2)	Historical reduction in extent (since 1750; 2, 4) Recent reduction in extent (last 50 years; 6) Future reduction in extent (next 50 years; 4)	**Variable** – ecosystem specific and minor consideration for most rivers, streams, lakes Almost always assessed in RLE examples (6/7 cases) Sometimes determined the Red List outcome (3/6 cases) Recent change – mostly quantified Historical change – can be qualitatively assessed, but often data deficient
Restricted distribution	Some ecosystems are naturally restricted; e.g. Naturally Uncommon Ecosystems of NZ (1) Can be a driven by environmental degradation; e.g. some lakes (3), wetlands (2)	Area of occupancy and extent of occurrence (2, 3, 5, 6, 7)	**Variable** – ecosystem specific Always assessed in RLE examples (7/7 cases) Often determined the Red List outcome (4/7 cases)
Environmental degradation	Water flow /hydrological modification (4,5) Mining (4,6) Climate change (3, 4, 7) Fire regime (7) Eutrophication and pollution (1, 4, 8) Sedimentation (9) Land-use change (8) Structural simplification Episodic disturbance	Annual flows (creeks) (4) Groundwater discharge rate (collapse assumed to occur if spring discharge <30–38 mm per day; 5) Average annual salinity (biota affected if beyond a range of 100–117 g/L; 2) Sea-level rise (most of the wetland extent will be affected by 2070; 5) Climate suitability (using models to determine future extent; 6, 7) Seasonal fluctuations in dissolved O, pH (*10) Dissolved oxygen (<2 mg/L; 9) Secchi depth (proxy for water clarity, thresholds set vary with salinity; 9) Nitrogen concentration (2)	**Major consideration** Almost always assessed in RLE examples (6/7 cases) Often determined the Red List outcome (4/7 cases) Quantitative thresholds often specified. More than 1 indicator often assessed.

Criterion	Threats identified	Indicators or measures	Relevance
Disruption of biotic processes or interactions	Invasive species (1, 4, 10*, 11*, 12*) Loss of key/native species (1, 4) Trophic alteration (11*)	Waterbird diversity (4) Presence of invasive species (Nile perch eliminated many other species, 10*; invasive goldfish triggered a regime shift, 12*; invasive eels reduced fish species richness and prey biomass for key wading bird species,13*; invasive eels reduced amphibian populations, 14*) Trends in woody re-sprouters (in upland swamp affected by fire; 6) Spatial coverage of keystone species (7, 9) Distribution of key species (threshold defined as 99% reduction in key biogenic habitat relative to baseline, 9) Species richness (6, 9) Mortality rates of fish in a lake (driven by hypoxia, 10*)	**Minor consideration** Often used in RLE examples (4/7 cases) Sometimes determined the Red List outcome (3/7 cases) Qualitative thresholds often used for abundance of key ecosystem biota
Quantitative assessment of ecosystem collapse risk	Climate change and water extraction effects on wetland (2)	Quantitative simulation model using scenarios based on future climate and different water extraction levels (2)	**Seldom possible** – one case study

References for case studies: 1: Holdaway et al. 2012; 2: raised bogs of Germany, Keith et al. 2013; 3: Murray et al. 2020; 4: Pisanu et al. 2015; 5: Karst Rising Springs wetland, Keith et al. 2013; 6: coastal sandstone upland swamps, Keith et al. 2013; 7; Coorong lagoons and Murray Mouth inverse estuary, Keith et al. 2013; 8; Keith et al. 2015; 9: Mahoney & Bishop 2017; 10*: Goldschmidt et al. 1993; 11*: Pintar & Dorn 2025; 12*: Hintz et al. 2026; 13*: Radar & Rackcliffe 2025; 14*: Howell et al. 2026;

* Indicates not an RLE case study IUCN criterion

6.2.3 Disruption of biotic processes or interactions

Compared with spatial extent and environmental degradation, disruption to biotic processes or interactions was less often assessed among the case studies we reviewed (Table 4). However, this seems likely to be driven by data availability rather than the relative importance of biotic processes. This criterion was assessed in just four of the seven case studies we reviewed, but for three of these it drove the overall Red List outcome.

Indicators and thresholds used to assess biotic disruption are ecosystem-specific. Among the case studies we reviewed, indicators included the distribution of key biogenic species (e.g. seagrasses; Mahoney & Bishop 2017; *Ruppia* spp., Keith et al. 2013); species richness for a functional group of top predators (Mahoney & Bishop 2017); re-sprouting plants (promoted by fire; Keith et al. 2013); and waterbirds (Pisanu et al. 2015).

Biological invasions of fish species have been implicated in freshwater ecosystem collapse, although we did not identify a case study that used the RLE framework. An example from the literature is that of Lake Victoria, East Africa, once considered to hold one of the most diverse fish assemblages in the world. The introduction of Nile perch (*Lates niloticus*) caused c. 200 species to be eradicated in less than a decade, disrupting the food web and driving a permanent change in community structure and composition (Goldschmidt et al. 1993). With few surviving native species, the former ecosystem is considered to have collapsed (Goldschmidt et al. 1993).

Assessment of biotic disruption will be an important component of RLE assessments for New Zealand freshwater ecosystems. There is a large body of work showing that invasion of streams and lakes by introduced fish species disrupts ecosystem processes, causes regime shifts, affects populations of native fish and invertebrates, and trophic interactions, and causes localised extinctions of some native taxa (Townsend & Simon 2006; Kelly & Schallenberg 2019; Schallenberg & Sorrell 2009). Across 95 New Zealand lakes, the prevalence of regime shifts was correlated with the introduced macrophyte *Egeria densa* and the number of introduced fish present, including *Ameiurus nebulosus* (catfish), *Carassius auratus* (goldfish), *Scardinius erythrophthalmus* (rudd), *Cyprinus carpio* (koi carp), and *Tinca tinca* (tench) (Schallenberg & Sorrell 2009).

Regime shifts and tipping points can have long-term consequences for ecosystem biota. An example is provided by Lake Ellesmere / Te Waihora. Historically, littoral lake regions had clear water that was maintained by banks of dense macrophytes (Kelly & Jellyman 2007). Cyclone Giselle in 1968 (the 'Wahine storm') is now seen as a tipping point: much of the macrophyte vegetation was uprooted (Gerbeaux, 1993), driving a major shift in benthic communities and trophic links to shortfin eels (Kelly & Jellyman 2007).

The threat status of several naturally uncommon New Zealand freshwater ecosystems was evaluated by Holdaway et al. (2012) using an earlier iteration of the RLE framework. These assessments incorporated aspects of biotic disruption by considering the inclusion of indicators such as native dominance and species occupancy (Holdaway et al. 2012).

6.2.4 Quantitative assessment of ecosystem collapse risk

We identified just one case study that quantitatively assessed the risk of ecosystem collapse risk among freshwater case studies. As noted above, the Coorong lagoons and Murray Mouth inverse estuary in South Australia are under threat from climate change and water extraction (Keith et al. 2013). Six scenarios were used to quantify the likelihood of ecological collapse using combinations of potential future climate projections and different water extraction levels. Ecological collapse was simulated to occur in four out of the six scenarios, and a dry future climate projection with the current water extraction level was always likely to result in ecological collapse. The modelled scenarios provided insight into the amount of degradation that is inherent due to climate change compared with that caused by extraction levels. The ecosystem was categorised as ‘Critically Endangered’.

Summary: All freshwater case studies we reviewed used more than one RLE criterion to assess collapse risk. Environmental degradation and restricted distribution most often determined Red List outcome. Environmental degradation was a major consideration, and, when assessed, used quantitative indicators specific to the ecosystem, often based on previous research (e.g. water flow, water quality indicators). Disruption to biotic processes was less often evaluated, but, when assessed, was usually based on trends in the abundance of native biota.

6.3 Terrestrial ecosystems

6.3.1 Reduced and restricted distribution

A reduced or restricted ecosystem extent was a major contributing factor in ecosystem collapse in the terrestrial case studies we reviewed, but its importance was variable and ecosystem-specific (Table 5). It is most informative for ecosystems affected by land-use change and/or conversion to other ecosystem types. This especially includes forests, which have been extensively cleared for agriculture and timber. All terrestrial case studies we reviewed used either reduced or restricted extent, and these criteria contributed to the overall Red List outcome in virtually all cases, and were the sole determinants of overall ecosystem Red List outcome in five out of nine cases (Table 5).

Restricted distribution (criterion B) most often determined the overall Red List outcome, especially for ecosystems considered collapsed, critically endangered or endangered. Where an ecosystem’s extent is largely geographically defined (e.g. due to abiotic conditions, such as for some of New Zealand’s naturally uncommon ecosystems; Holdaway et al. 2012), restricted extent is likely to be the most important consideration. However, in the RLE framework this only applies when there are also either very few known locations, evidence of environmental degradation, or disruption of biotic processes (IUCN 2024).

Reduced and restricted distribution were usually quantitatively assessed using maps, satellite imagery, historical vegetation surveys, and reports or other historical data (e.g. logging records). Expert opinion was used to estimate the extent of recent or historical decline, as was done for many naturally uncommon ecosystems by Holdaway et al. (2012). Spatial data on ecosystem extent were sometimes only available across part of the ecosystem distribution. Reductions in extent were still able to be assessed when existing data were considered representative of changes occurring overall (e.g. reductions in the extent of alpine patch herbfields; Williams et al. 2015).

Where boundaries of ecosystem types are difficult to distinguish from adjacent habitat, rules can be developed. For example, in a woodland ecosystem, Tozer et al. (2015) judged areas with less than 5% tree cover to have been cleared, so the sensitivity of the risk assessment to varying methods of defining ecosystem boundaries could be assessed (Tozer et al. 2015).

Future changes in extent (criterion A2) could be projected assuming that current rates of change will continue into the future, given current policy and land management practices. For example, woodland clearance on the Cumberland Plain, Australia, is driven by agricultural conversion and urban spread. Tozer et al (2015) estimated the lower and upper bounds of projected woodland clearance based on recent rates of clearance (e.g. 1997–2007) while considering policy changes that might affect clearance rates.

Reduced or restricted extent is likely to be a major determinant of RLE collapse risk in some New Zealand terrestrial ecosystems. New Zealand has been estimated to have lost 90% of its wetlands, although this varies across regions and wetland type (Ausseil et al. 2011). A historically reduced extent (since 1250) is emerging as the predominant determinant of Red List outcome in assessments currently being undertaken for New Zealand palustrine wetlands (O. Burge, pers. comm.).

Among forests, lowland forest types have been most affected by land-use change (Mark 1980). Although there is no accepted definition of what constitutes ‘lowland’ forest in New Zealand, spatial databases show that 90% of pre-human indigenous forests below 100 m elevation have been lost due to land clearance (decreasing to 80% below 400 m elevation), with the majority of lowland forest remaining in the west and south of the South Island in areas historically less favoured for habitation or exploitation (Walker 2024).

Holdaway et al. (2012) considered approximately a third of New Zealand’s naturally uncommon terrestrial ecosystems to be threatened, based on their very small extent or their extent in combination with known threats. These assessments were made qualitatively and used an earlier version of the RLE criteria. For each ecosystem, extent was usually estimated as a range, and in the assessment the upper limit was used as a conservative estimate.

6.3.2 Environmental degradation

Environmental degradation was of moderate concern in terrestrial case studies and was usually a consequence of climate change or land-use change (Table 5). Environmental degradation was quantitatively assessed in five cases but determined the overall Red List outcome for none. Examples in case studies included models to assess the impacts of climate change; for example, to predict the future persistence of snow in alpine patch herbfields (e.g. Williams et al. 2015), environmental suitability models for a dryland ecosystem (Chen & Ma 2026), and changes in the number of cloud days and precipitation in cloud forests (Auld & Leishman 2015).

Environmental degradation is a major concern in New Zealand palustrine wetlands, where drainage is the major form of reduction in wetland extent (McGlone 2009; Denyer & Peters 2020). Climate change is also likely to affect many New Zealand wetlands due to shifts in rainfall or the frequency of extreme weather events (affecting sedimentation; Burge 2024). New Zealand has several naturally uncommon geothermal ecosystems, and these may be affected by geothermal fluid extraction for energy, which affects surface activity, causes subsidence, alters the water table and soil temperatures, and consequently affects the plant communities present (Beadel et al. 2018). Many New Zealand terrestrial ecosystems are affected infrequently by fire, especially early successional woody vegetation, wetlands, and grasslands (Perry et al. 2014), but impacts may increase in some regions with climate change (McGlone & Walker 2011).

Although deforestation of native forest is much reduced from historical levels, it may still be a concern in some regions or forest classes. Deforestation mapping shows that over 48,000 ha of ‘natural’ pre-1990 forest has been cleared since 1990 (Ministry for the Environment 2026), although this figure may include clearance of exotic species that form naturally regenerating forest. Walker (2024) has suggested that for native forest, the status quo might be about 7,000–8,000 ha cleared nationally per decade. Climate change may also increase rates of forest loss through increased flooding and slips; for example, sizeable damage occurred in West Coast forests due to Cyclone Ita (Walker 2024).

Table 5. Summary of ecosystem collapse criteria for the terrestrial domain

Criterion	Threats identified	Indicators or measures	Relevance
Reduced distribution	Driven by land clearance/logging/harvest (1, 2, 3, 4, 14) Driven by fire (16*)	Historical reduction in extent (since 1750; 1, 2) Recent reduction in extent (last 50 years; 2) Future reduction in extent (next 50 years; 7, 9*) Future extent predicted in a simulation model (16*)	**Major consideration**, ecosystem-specific- especially important for forests Assessed in almost all RLE examples (8/9) Recent reductions – mostly quantified / mapped Historical reductions – mostly qualitative, often data deficient
Restricted distribution	Driven by land clearance/development (2, 3, 5)	Area of occupancy and extent of occurrence (2, 5, 7, 12)	**Moderate consideration** Assessed in almost all RLE examples (8/9) Often quantified; occasionally assessed qualitatively (e.g. 12)
Environmental degradation	Harvest/logging (4, 6) Climate change (3, 5, 7, 9) Habitat suitability (4, 7) Fire (1, 6, 16*) Drought (10*) Mining (11*) Hydrology (13*) Soil type (14)	Logging rate (collapse assumed if all remaining forest is fragmented; 4) Snow cover (used climate predictions to determine when snow cover will decline to zero; 5) Habitat suitability (using climate models; 3, 9) Fire return interval (8) Rainfall (collapse assumed at annual rainfall of 900 mm; 7) Cloud cover (collapse assumed when cloud days decline by 50%; 7) Land suitable for future development (for assessing potential future impact; 3) Hydrological trends (changes in cover of open water, surface water flow, ground water depth; 13*) Soil type via anthropogenic impacts (14)	**Moderate consideration** Often data deficient Assessed in c. 50% RLE examples Seldom contributed to the Red List outcome
Disruption of biotic processes or interactions	Recruitment failure (7, 17*) Species composition (5, 13*) Biological invasions (1, 2, 5, 7) Herbivory (13*) Disease/pathogens (8) Loss of structural complexity (6, 15*) Trophic alterations (no example in case studies)	Rat predation on seeds/seedlings of important tree species (affecting survival and recruitment, qualitative assessment; 7) Spatial extent of tree regeneration failure (and predictors of regeneration failure; 17*) Species composition change (increased shrub cover, collapse of herbfield assumed when shrub cover is 50%; 5) Regime shift from forest to grassland (13*) Weed cover (1) Decline in key species (susceptible to pathogen; 8) Loss of structural complexity (collapse assumed when there is less than 1 large, hollow-bearing tree/ha to provide habitat; 6)	**Major consideration** Often data deficient Assessed in c. 50% RLE examples Seldom contributed to the Red List outcome

Criterion	Threats identified	Indicators or measures	Relevance
Quantitative assessment of ecosystem collapse risk	Simulation of logging / forest fragmentation (4) Simulation of logging/fire, causing decline in hollow trees (6)	Historical logging rate was used to estimate the proportion of unfragmented forest in 50 years' time. The collapse assumption is that if only forest fragments remain, functional forest collapse would be imminent (4). Model of forest growth to predict likelihood of collapse given current logging and fire regimes. Collapsed state defined as when there is less than 1 large hollow tree/ha (6).	**Seldom possible** – two case studies Seldom contributed to the Red List outcome

1: Tozer et al. 2015; 2: Metcalfe & Lawson 2015; 3: Chen & Ma 2026; 4: DellaSala et al. 2021; 5: Williams et al. 2015; 6: Burns et al. 2015; 7: Auld & Leishman 2015; 8: Barrett & Yates 2015; 9: Chen et al. 2024; 10, Au et al. 2023; 11*: Cairns et al. 2025; 12: Holdaway et al. 2012; 13*: Cooper et al. 2025; 14: Crespin & Simonetti 2015; 15*: Lindenmeyer & Sato 2018; 16*: Jones et al. 2025; 17*: Taylor et al. 2025;

* Indicates not an RLE case study

6.3.3 Disruption to biotic processes

Disruption of biotic process is a major consideration in terrestrial ecosystems but was rarely assessed in the case studies we evaluated. Just five out of nine case studies assessed disruptions to biotic processes, which was only informative for the overall Red List outcome in one case study. Indicators and collapse thresholds were specific to the study ecosystem and required quantitative data. Indicators and measures examined across case studies we examined included trends in species composition (Williams et al. 2015), the impacts of disease or pathogens (Barrett & Yates 2015), biological invasions (Tozer et al. 2015), loss of structural complexity (Burns et al. 2015), and predation of seeds or seedlings causing reduced recruitment and survival (Auld & Leishman 2015; Table 5). Examples from the general ecological literature included an examination of the extent and predictors of tree regeneration failure (Taylor et al. 2025).

Examples of biotic disruptions to New Zealand terrestrial ecosystems are numerous and include biological invasions (herbivory or predation of native biota from mammals; Clout 2006; competitive exclusion by invasive weeds, Hulme 2020), pathogens or disease (e.g. myrtle rust, McCarthy et al. 2024; kauri dieback, Simpkins et al. 2025), disruptions to mutualisms (e.g. loss of native birds or insects reducing pollination or seed dispersal; Anderson et al. 2011; Wotton & Kelly 2012), and altered fire regimes (an abiotic threat, but promoted by fire-adapted non-native plant species; Standish et al. 2008; Perry et al. 2014).

6.3.4 Quantitative assessment of ecosystem collapse risk

We identified two case studies that quantitatively assessed the risk of ecosystem collapse risk, both of them for forest ecosystems. First, for the mountain ash forests of southeast Australia, Lindenmeyer and Sato (2018) describe the effects of logging and wildfire that drive 'hidden collapse' of these mountain ash forests. The quantitative assessment classified the ecosystem as 'Critically Endangered' (Burns et al. 2015). This used a probabilistic model of tree-growth stages, combined with the frequency of hollow-bearing trees (essential wildlife habitat) to model the probability of ecosystem collapse 50 years into the future, given current logging and fire regimes. This approach used 10,000 simulations generated by varying the input parameters. All scenarios indicated a greater or equal than 92% chance of reaching a collapsed state (which was defined as less than one hollow-bearing tree per hectare) by 2067 (Burns et al. 2015). The collapse appears 'hidden' because the forest may appear intact while undergoing a prolonged period of decline and very slow recovery times, making collapse almost inevitable.

Quantitative assessments of collapse risk were also undertaken for interior wetbelt and inland temperate rainforest in British Columbia, Canada. Assessment was based on a model using historical logging rates to estimate the proportion of unfragmented forest that might remain after 50 years. The assessment relied on an assumption that if only small patches of forest remained, their ecosystem functions would be so disrupted that functional collapse would be imminent (DellaSala et al. 2021).

Summary: All terrestrial case studies we reviewed used more than one RLE criterion to assess collapse risk. Reduced and restricted distribution most often determined the Red List outcome. Environmental degradation, when assessed, used indicators specific to the ecosystem, often based on previous research (e.g. effects of climate change, changes to fire regime). Disruption to biotic processes, when assessed, was based on the adverse effects of weeds, disease, or reduced habitat suitability (e.g. reduced structural complexity).

7 Case study: Mackenzie District widespread terrestrial ecosystems

The widespread terrestrial ecosystems of the Mackenzie District are suitable candidates to evaluate in the context of ecosystem collapse, both because of their transformation over time and because they have been well studied (Appendix 2). This case study does not consider the naturally uncommon terrestrial ecosystems of the Mackenzie District, or its freshwater ecosystems (rivers and lakes), which have also been highly transformed over time.

This case study allows evaluation of baselines set at 1250, 1750, and 50 years ago (1975), providing different ways to assess and interpret collapse in the context of different ecosystems that were dominant at each baseline. This case study also allows evaluation of collapse in the novel, modified ecosystems that have replaced natural ecosystems (1975 baseline).

The predominant ecosystems of the inland outwash surfaces of the Mackenzie District, defined in terms of the dominant vegetation cover (as used widely to define ecosystems; e.g. Tierney 2022; Lötter et al. 2023), of each period were as follows.

- 1250: natural forests (dominated by *Halocarpus bidwillii*, *Phyllocladus alpinus*, *Podocarpus laetus*, with some beech). This is the bog pine, mountain celery pine scrub/forest of Singers and Rogers (2014); see also Wardle (1991, p. 196).
- 1750: tall tussock grassland (dominated by *Chionochloa* spp.), which colonised the previously forested area after its transformation by fire. These are predominantly subalpine–subnival snow tussock grasslands (specifically the *Chionochloa rigida / Poa colensoi – Festuca novae-zelandiae / Hypochaeris radicata* tussockland alliance of Wiser et al. 2016; see also Connor 1964), and also cool temperate – low alpine, tall red/copper tussock grasslands (specifically the *Chionochloa rubra* / *Schoenus pauciflorus–Celmisia coriacea–Coprosma cheesemanii* tussockland alliance of Wiser et al. 2016; see also Connor 1964).
- 1975: degraded short tussock grassland (dominated by *Festuca*) and non-native grasses and herbs, which dominated after repeated fire and grazing by sheep and rabbits. This is cool temperate–low alpine, short tussock grassland (the *Festuca novae-zelandiae / Anthoxanthum odoratum–Trifolium repens–Hypochaeris radicata* alliance of Wiser et al. 2016; see also Meurk et al. 2002).

Tall tussock grassland was present in 1250 (McGlone & Moar 1998), but its extent is unclear and likely to have been restricted to high subalpine elevations. **Degraded** short tussock grasslands are a modern artefact that did not exist in either 1250 or 1750.

We evaluate the criteria to assess the risk of ecosystem collapse for each of the three ecosystems that were predominant at the different baselines, based on their current (2026) extent and condition. We do not evaluate change in tall tussock grasslands against a 1250 baseline, although they were present in 1250 (Appendix 2).

7.1 Natural forests (1250 baseline)

7.1.1 Reduction in geographical distribution

There has been an over 90% reduction from the original extent. Natural forest was a widespread ecosystem occupying thousands of hectares in 1250. Natural forests are now reduced to tiny, disconnected fragments, some likely to be in unrepresentative sites (e.g. in those that could not easily transport fire). The drastic reduction in extent of natural forest certainly merits 'Critically Endangered' status in the IUCN Red List at the scale of the Mackenzie District. This is likely to apply more widely, since the same pressures resulting from fires destroying native upland conifer forests of the eastern South Island applied more extensively, with other regions and districts similarly deforested and with few or no remaining fragments. If an alternative baseline of 1750 were used, in terms of reduced geographical distribution the perverse outcome would be that natural forests would be assessed as unchanged and of no concern (i.e. they had already been reduced drastically before 1750, and the remaining fragments were not further diminished). The same would apply if the 1975 baseline were used.

7.1.2 Restricted geographical distribution

This is not assessed in this evaluation, because declining distribution (above) already designates natural forest ecosystems as 'Critically Endangered'. Data to assess restricted distribution are available and would draw on maps of current ecosystem extent, and measures of extent of occurrence and area of occupancy, and the number of threat-defined locations could be determined. Under this criterion, natural forests almost certainly meet the threshold for assignment of a threat category, regardless of whether a 1250 or 1750 baseline is used.

7.1.3 Environmental degradation

(a) *Potentially altered hydrology*. This is likely to have arisen following deforestation, potentially altering the survival of woody plants. Some fragments near modern hydro schemes may have been influenced by altered water-tables upstream of dams, or by diversion of water from braided rivers to hydro channels.

(b) *Loss of soil organic matter and nutrients caused by wind erosion*. After forest fires, loss of nutrient capital is likely at large scales, with local loss affecting natural forest fragments.

(c) *Road and dam construction*. Some fragments may have been destroyed because of these activities.

Variables for these three factors could be either quantified or approximated in a model to assess risk of collapse of the remaining ecosystems within the next 50 years. For example, root-zone moisture capacities could be measured across the remaining forest fragments and incorporated into hydrological models under different climate change scenarios (e.g. Nijzink et al. 2016); and measurements of soil organic matter in fragments could similarly feed into models of likely change with respect to surrounding land use (Poeplau et al. 2011).

7.1.4 Disruption to biotic processes:

(a) *Biological invasions*. Some remaining natural forest fragments have been invaded by non-native herbs and grasses, which may compete with seedlings of the forest trees or alter microsites for their regeneration. There is a high likelihood of invasion of natural forest fragments by non-native conifers with taller stature and much more rapid growth rates, and which are fire-adapted (transporting or promoting fire and regenerating readily after fire, in contrast to native trees). Non-native herbivores (some or all of deer, sheep, brushtail possums, hares, and rabbits) are likely to affect all life-history stages of the resident trees and the likelihood of regeneration. Non-native mammalian predators (mustelids, rodents, cats) are likely to affect native bird, reptile, and invertebrate communities in the forest fragments.

(b) *Extinctions*. It is very unlikely that the full range of forest flora and fauna occurs among the very small remaining fragments, even if they contain most of the dominant tree species, or that the current extent of some patches is sufficient to sustain breeding populations of some flora and fauna. Some species (especially vertebrates and large invertebrates) have become extinct since 1250. It is therefore very likely that ecosystem processes (e.g. pollination, seed dispersal, mutualisms) have been altered, and that some ecosystem processes (e.g. litter decomposition) have been altered as a consequence (Essl et al. 2026).

(c) *Fragmentation effects*. It is likely that the high perimeter:area ratio of the remaining natural forest fragments and their proximity to large, deforested landscapes has resulted in an altered microclimate and exposure to wind (including redispersal of litter), potentially rendering the fragments vulnerable to chronic pressures of climate change (e.g. sustained hot drought periods) or extreme events. The structure of the remaining natural forests has not changed in the last 50 years since their first description, nor have they lost taxa, so a case could be made that the drastic alteration of biotic processes merits 'Critically Endangered' status in the IUCN Red List rather than a collapsed status. Most biological processes could be either quantified (e.g. by reassessments of flora and fauna in individual forest fragments; cf. earlier assessments, such as by Wardle 1991, p. 196, or measurement of some biotic pressures, such as rabbit abundance), or approximated in a model (e.g. models, under different future climate scenarios, using comparative growth rates of native and non-native conifers in forest fragments; Cieraad 2011) to assess risk of collapse of the remaining natural forests within the next 50 years.

7.1.5 Relevance of tipping points and thresholds

The trajectory of change in the natural forests of the Mackenzie District is a strong candidate for an abrupt collapse profile (Figure 1; Bergstrom et al. 2021), with their drastic reduction in distribution resulting from human-caused landscape-level fire in the 14th century (Appendix 2). The natural forests had been subject to fire disturbance before human settlement and had regenerated afterwards (McGlone & Moar 1998), and their failure to do so after human-caused fires suggests that the fires were of a scale and/or intensity that was unprecedented (Ogden et al. 1998; Tepley et al. 2016). The introduction of large-scale fire drastically reduced the distribution of natural forests in the District, and the grasslands that replaced them were resilient to fires and can be viewed as an alternative state to forests (Kitzberger et al. 2016), creating a positive feedback for subsequent fires and reducing the ability of forests to spread and occupy their former extent. Additional pressures that have mostly applied since the 19th century (notably biological invasions that alter biotic processes) make it even less likely that natural forests could expand in distribution or function without significant human investment.

7.2 Tall tussock grassland (1750 baseline)

7.2.1 Declining distribution

Substantially reduced from their 1750 extent (Connor & Macrae 1969; Wardle 1991), tall tussock grasslands are now reduced to fragments, some likely to be in unrepresentative sites (e.g. in those that could not easily transport fire) and other sites from which grazing mammals have been excluded in the long term (e.g. roadside reserves), or where grazing and fire pressure have been least (high-elevation sites). Across much of its former distribution relict individual tall tussocks occur as minor components of contemporary ecosystems (Connor 1964; Connor & Macrae 1969). The major reduction in distribution of tall tussock grasslands could mean that, on this criterion, they might merit 'Endangered' status in the Red List at the scale of the Mackenzie District, subject to quantifying their 1750 distribution. This could apply more widely across the eastern South Island, since the same pressures resulting from fires destroying tall tussock grasslands applied throughout, with other regions and districts similarly subject to ongoing pressures from fire and grazing, and some have few or no remaining fragments.

7.2.2 Restricted distribution

This was not evaluated in our case study assessment. Declining distribution (above) probably already designates tall tussock grassland ecosystems as 'Endangered'. Data to assess restricted distribution is available, and would draw on maps of current ecosystem extent, and measures of extent of occurrence and area of occupancy.

7.2.3 Degradation of abiotic environment

(a) *Potentially altered hydrology*. This is likely to have arisen as a result of the destruction of tall tussocks, which can harvest aerosol moisture, potentially altering the survival of tussocks and co-occurring herbs in fragments. Some fragments near modern hydro schemes may have been influenced by altered water-tables upstream of dams, or by diversion of water from braided rivers to hydro channels.

(b) *Loss of soil organic matter and nutrients caused by wind erosion*. After fires and grazing, loss of nutrient capital was likely at large scales, with local loss affecting tall tussock grassland fragments.

(c) *Road and dam construction*. Some tussock grassland fragments may have been destroyed because of these activities.

These three pressures contribute to an evaluation of risk of tall tussock grassland collapse but are less important (or only locally important) than reduction in distribution or altered biotic processes. Variables for these three factors could be either quantified or approximated in a model to assess risk of collapse of the remaining tall tussock grasslands within the next 50 years.

7.2.4 Altered biotic processes

(a) *Biological invasions*. Remaining areas of tall tussock grasslands have been invaded to varying degrees by non-native woody plants that alter their physiognomy and potentially their function. Most notably, they have been invaded by non-native conifers that alter ecosystem processes and shade out native grasses. Even if removed, the legacy effects of non-native conifers promote the growth of non-native grasses (Green et al. 2025). Other non-native woody plants invade tall tussock grasslands (e.g. *Rosa rubiginosa*), as well as non-native herbs and grasses, which may compete with the seedlings of native grasses and herbs or alter microsites for their regeneration. Non-native herbivores (some or all of deer, sheep, brushtail possums, hares, and rabbits) have affected tall tussock grassland and continue to do so, reducing biomass and drastically altering community composition, and since c. 1850 they have significantly reduced the abundance and biomass of formerly important native herbs (Wardle 1991). Non-native mammalian predators (mustelids, rodents, cats) are likely to affect native bird, reptile, and invertebrate communities in the remaining tall tussock grasslands.

(b) *Extinctions*. Some species (especially vertebrates and large invertebrates) have become extinct since 1250 and others since the 19th century, notably buff weka (*Gallirallus australis hectori*), which was abundant in tall tussock grasslands at the time of European settlement (Wardle 1991). It is very unlikely that the full range of forest flora and fauna occurs in the remaining, unrepresentative fragments of tall tussock grassland, with some species eliminated or greatly reduced in abundance because of biological invasions and repeated fires. Because of this, ecosystem processes (e.g. pollination, seed dispersal, mutualisms) will probably have been altered, as well as ecosystem processes (e.g. litter decomposition). Many biological processes have been drastically altered in these ecosystems, and this could be either quantified (e.g. using repeated measurements of vegetation; Day & Buckley 2013) or approximated in a model to assess risk of collapse of the remaining tall tussock grasslands within the next 50 years.

7.2.5 Relevance of tipping points and thresholds

The trajectory of change in the tall tussock grasslands of the Mackenzie District probably most closely follows a smooth collapse profile (Figure 1; Bergstrom et al. 2021) since their assembly after deforestation in the 14th century and their subsequent reduction in extent and degradation, especially since the 19th century (Appendix 2). Some aspects of the trajectory could approximate a stepped collapse profile since the 1940s, especially when tall tussock grasslands are invaded and replaced by non-native conifers at large scale, completely altering their composition, structure, and function.

7.3 Degraded short tussock grassland (1975 baseline)

7.3.1 Declining distribution

The extent of degraded short tussock grassland in the Mackenzie District in 1975 was likely to be c. 340,000 ha (based on its extent in 1990, and assuming the same rate of loss as over 1990–2001, with c. 20,000 ha excluded as a possible extent of tall tussock grassland: Weeks et al. 2013). Assuming the rate of loss to be the same as for 2001–2008 over the last 17 years from the 2008 extent (303,700 ha; Weeks et al. 2013), then current (2025) extent is c. 266,500 ha, based on conversion of land (including to irrigated dairy pasture) alone, and not including other sources of reduction in distribution (e.g. non-native conifer invasion sufficient to form closed canopies). Based on this conservative estimate of loss, the c. 22% reduction since 1975 is insufficient to meet the >30% threshold to class these grasslands as ‘Vulnerable’ in this criterion of the IUCN Red List.

7.3.2 Restricted distribution

This was not assessed in this evaluation. Data to assess restricted distribution are available and would draw on maps of current ecosystem extent, and measures of extent of occurrence and area of occupancy.

7.3.3 Degradation of abiotic environment

(a) *Potentially altered hydrology*. Some short tussock grasslands near modern hydro schemes may have been influenced by altered water-tables upstream of dams, or by diversion of water from braided rivers to hydro channels. Others close to irrigated pasture may have altered hydrology.

(b) *Loss of soil organic matter and nutrients caused by wind erosion*. The effects of earlier decades of fire and grazing, and continued grazing by sheep, rabbits, hares, etc., probably caused further loss of nutrient capital at large scales over the last 50 years, perhaps constituting a cause of mortality in short tussocks.

(c) *Local high nutrient input*. Application of nitrogen and phosphorus fertilisers in irrigated pasture may have affected the nutrient status of adjacent short tussock grasslands.

(d) *Road construction and property subdivision for housing*. Some short tussock grasslands are likely to have been destroyed because of these activities.

These four pressures contribute to an evaluation of risk of degraded short tussock grassland collapse, but they are only locally important and generally contribute less to the likelihood of ecosystem collapse than altered biotic processes or declining distribution. Variables for these four factors could be either quantified or approximated in a model to assess risk of collapse of the remaining tall tussock grasslands within the next 50 years.

7.3.4 Altered biotic processes

(a) *Biological invasions*. Remaining areas of degraded short tussock grasslands were already been widely invaded by non-native plants in 1975, and this process has continued ever since. Some of these grasslands are overwhelmingly dominated by non-native plants (e.g. the mat-forming herb *Pilosella officinarum*, which alters soil nutrient cycling processes; Scott et al. 2001).

The rate of invasion by non-native conifers, especially *Pinus contorta*, into short tussock grasslands of the Mackenzie District has increased significantly since 1975. These conifers fundamentally alter the physiognomy and function of short tussock grasslands and, on forming closed canopies, shade out most grasslands species. Even if removed, the legacy effects of non-native conifers promote the growth of non-native grasses over native species (Green et al. 2025).

Non-native herbivores (some or all of deer, chamois, sheep, brushtail possums, Bennett's wallaby, hares, and rabbits) caused degradation of short tussock grassland before 1975 and continue to do so, reducing biomass and drastically altering community composition. Non-native mammalian predators (mustelids, rodents, cats, hedgehogs) are likely to affect native bird, reptile, and invertebrate communities throughout degraded short tussock grasslands.

Some biological pressures that affect degraded short tussock grassland are strongly influenced by changing land use; for example, adjacent fertilised pasture supporting livestock (as has occurred in the Makenzie District since 2000) can boost rabbit abundance and associated top predators, reducing the abundance of native lizards and macroinvertebrates (Norbury et al. 2013).

The drastic alteration of biotic processes in degraded short tussock grasslands in the Mackenzie District may be sufficient to merit 'Vulnerable' status in the Red List. Most biological processes could be either quantified or approximated in a model to assess risk of collapse of the remaining natural forests within the next 50–100 years. For example, some biotic invasions that affect these ecosystems (e.g. Bennett's wallaby, *Macropus rufogriseus*, and non-native conifers) are quantified, with data available through publicly available databases (e.g. Wilding Conifer Information System, through LINZ; https://www.wildingpines.nz/assets/Documents/WCIS_User_Guide_v3.0.pdf). There are likely to be sufficient time-series data for several components of the biota of these grasslands (e.g. Meurk et al. 2002; Day & Buckley 2013) to inform a model to evaluate the risk of their collapse.

7.3.5 Relevance of tipping points and thresholds

The trajectory of change in the degraded short tussock grasslands of the Mackenzie District probably most closely follows a smooth collapse profile (Figure 1; Bergstrom et al. 2021), both before and since 1975 (Appendix 2). Some aspects of the trajectory could approximate a stepped collapse profile since 1975, especially when degraded short tussock grasslands are invaded and replaced by non-native conifers at a large scale, or when they are replaced by irrigated dairy pasture wholly comprising non-native grasses and herbs; both processes completely alter the composition, structure, and function of degraded tussock grasslands.

Evidence after the removal of non-native conifers suggests that the effects of their invasion in degraded short tussock grasslands are irreversible, with non-native grasses dominating sites where conifers are eliminated (Green et al. 2025) and below-ground biota and processes fundamentally altered (Sapsford et al. 2022; Dudenhöffer & Hulme 2025). The change in land use to irrigated pasture is probably an irreversible process with respect to re-establishing degraded tussock grasslands or other baseline states (natural forests or tall tussock grasslands); to our knowledge there are no studies on attempts to restore pasture to tussock grassland.

8 Conclusions

This report provides a definition for ecosystem collapse that has been assessed and adopted internationally. The definition provides useful overarching principles for assessing ecosystems across marine, freshwater, and terrestrial domains. We provide definitions and explanations for other related terminology (tipping points and thresholds, early warning signals) and discuss their utility for predicting ecosystem collapse.

8.1 Indicator selection

The international examples assessed in this report across all domains demonstrate that the aspects emphasised in studies of ecosystem collapse, such as key indicators of ecosystem degradation or function, are ecosystem specific (e.g. Bland et al. 2018). Identifying and using indicators in RLE evaluations is a challenge, because under RLE guidelines indicators are used to assess the severity of a detrimental threat or trend across the entire extent of the ecosystem. A lack of spatial or temporal coverage of data can preclude use of a relevant indicator, even if it may be informative for ecosystem collapse at a local scale.

The examples we assessed show that data deficiencies often limit assessment of environmental degradation or disruption to biotic processes. It was also not always explicitly explained why a specific indicator had been selected (see also Rowland et al. 2018), though indicator selection should be transparent so that the quality of Red List assessments can be evaluated. RLE guidelines also stipulate that, wherever possible, several criteria and indicators should be used to avoid underestimating collapse risk (IUCN 2024). However, in practice data deficiencies may bias assessments by forcing the use of relatively easily measured indicators, or indirect indicators that can be derived from spatial data sets (e.g. land-use change as a proxy for sedimentation; Gillies et al. 2020).

Selection of indicators for assessing collapse risk of New Zealand ecosystems should be made using domain-specific expertise. Standard indicators in New Zealand's Outcome Monitoring Framework may provide the data needed to assess a criterion (e.g. Department of Conservation 2024). Assessments should be subject to standards imposed by adequate processes of peer review, using domain-specific expertise.

8.2 Ecosystem spatial distribution

Ecosystems with a small geographical extent, or that are declining in extent, are at greater risk of collapse than those distributed more widely. Spatial extent of ecosystems (i.e. reduced and restricted extent) is the most frequently used criterion to assign collapse risk (24 out of 27 cases overall), probably because this information is relatively easy to measure and evaluate. This criterion alone was often sufficient for assigning a risk outcome to ecosystems that have undergone the most extreme reductions in extent, or those that are naturally restricted. It may be that assessment of the collapse risk of ecosystems with reduced distribution has commanded attention in RLE assessments because they are spatially discrete, occupy a comparatively small total area, and some face pressing risks of abiotic degradation or altered biotic processes (e.g. the risk assessment conducted of New Zealand's naturally uncommon terrestrial ecosystems; Holdaway et al. 2012).

Assessing the spatial extent of ecosystems is more achievable when they can be readily mapped (e.g. terrestrial ecosystems, and if they can be structurally and compositionally differentiated using satellite or aerial imagery; Murray et al. 2018), but will be more challenging for other systems (e.g. marine ecosystems, unless they have strong structural features, Lohrer et al. 2024). For New Zealand RLE assessment to proceed at GET levels 4-6 (as recommended by the IUCN and required for international reporting), investment into accurately mapping the historical and current extent of ecosystems is required (including assessments of map accuracy and ground-truthing). This is also contingent on parallel work to improve or develop existing ecosystem typologies. These are specialist tasks whose successful delivery depends on multidisciplinary teams and sustained, multi-year investment.

8.3 Thresholds

A structured programme of research and monitoring is required to fill data gaps that pertain to ecosystem condition, threats, and functioning, across domains, so that defensible, evidence-based thresholds can be developed. The international reviews we evaluated concluded that, while critical thresholds and tipping points are useful concepts, they are challenging to predict and are typically identified retrospectively after collapse has occurred (Newton 2021; Sato & Lindenmayer 2018).

However, assessments under the RLE framework have the scope to incorporate quantitative thresholds of ecosystem collapse, particularly for assessing environmental degradation and disruptions to biotic processes. Setting defined quantitative thresholds may be possible where there are enough examples of collapse of a widespread ecosystem, and where thresholds are abrupt or stepped (Bergstrom et al. 2021). Defining a quantitative threshold where the trajectory of an ecosystem towards collapse follows a smooth profile (section 4) is more difficult. Setting qualitative thresholds for smooth or fluctuating profiles of ecosystem collapse could be a pragmatic approach, but it needs to be defensible and widely repeatable and will require ecosystem-specific approaches.

Although most of the case studies we reviewed considered ecosystem collapse with respect to the RLE approach (i.e. collapse risk was considered with respect to all ecosystem occurrences), ecosystem collapse can also be considered at the local scale (i.e. the scale of an ecosystem occurring at a one location). Research at the local scale will be needed to formulate quantitative or qualitative thresholds for collapse risk assessment; this could include, for example, formulation of predictive models of collapse risk, space-for-time substitutions, or comparison of degraded vs functional examples of ecosystems. For RLE, this must be complemented by standardised ecosystem monitoring, so that severity of degradation can be assessed across the entire ecosystem extent to determine whether thresholds have been exceeded.

Our case studies of widespread ecosystems in the Mackenzie District at three baseline periods provides few instances of thresholds that are defensible. The best available data are about former geographical distribution. Even the >90% reduction of natural forests in the district (from a 1250 baseline) does not constitute their complete collapse, and it would be difficult to assign a threshold (other than their complete destruction) that represented collapse. The trajectory for the next 50 years in forest extent could be static, as it has been for the last 50 years at least, in which case setting thresholds based primarily on their geographical extent is of little use. Clearly, current forest extent requires complete protection.

The greater threats to the ecosystems of the Mackenzie District are disruption of biotic processes and environmental degradation. The criteria required to define the likelihood of collapse in terms of abiotic degradation or disruption of biotic processes (e.g. in the IUCN Red List, or to set environmental limits) requires estimating the magnitude of past or future environmental degradation or disruption to biotic processes, expressed as a percentage relative to a change large enough to cause ecosystem collapse. Even for such comparatively well-studied ecosystems as those in the Mackenzie District, with long histories of measurement, we judge that the data needed to assess whether changes caused by abiotic degradation or disruption of biotic processes are large enough to cause ecosystem collapse are inadequate and fall far short of those needed to set defensible thresholds. Multidisciplinary expert panels could assess the likelihood of collapse of these ecosystems under various future scenarios (e.g. increased fire, biological invasions, drought) but any qualitative assessment runs the risk of not being repeatable, say, 50 years hence.

8.4 Phasing of RLE assessments in New Zealand

International reporting commitments require RLE assessments to be conducted for all New Zealand ecosystems across all domains. Evaluation of RLE status is probably possible nationally at a high level of global ecosystem classification, such as GET Level 4, once the ecological zones for this level are defined for New Zealand (similar to the approach currently being trialled for New Zealand wetlands). Evaluation of RLE status should also be possible at the next level of ecosystem classification (global ecosystem types, at Level 5 of the IUCN GET; Keith et al. 2022). RLE assessments even at high levels such as these might serve as the basis for some prioritisation of national-scale policy and management. However, they will not suffice for setting policy or management at smaller scales (e.g. for New Zealand's regional and local government) and they may not suffice as a basis to inform resource management decisions.

To ensure RLE assessments are useful for identifying and distinguishing the ecosystems most at risk of collapse, finer levels of ecosystem classification are required. Assessments are required for sub-global types, i.e. Level 6 of the IUCN GET (Keith et al. 2022), or in the case of terrestrial ecosystems – for alliances and associations (Levels 7 and 8, respectively), of the International Vegetation Classification (Faber-Langendoen et al. 2025). At these finer levels of ecosystem classification, and when applied at finer geographical scales (e.g. at the scale of regional or district councils for resource management decisions), gaps in data to assess RLE criteria A–E are likely (Parliamentary Commissioner for the Environment 2019), resulting in many ecosystems being determined as data deficient or reliant on expert opinion and qualitative data to assess their risk of collapse.

The risk of relying on the latter instead of filling gaps in environmental data is non-repeatability of assessments or a higher risk of litigation in challenging decisions about particular ecosystems. At a minimum, standardised approaches to determining collapse thresholds need to be established, at least within the same or similar ecosystems, while adhering to international best practice. This should ultimately facilitate comparisons among regions and compiling evaluations at the national scale. It is advisable to continue with those ecosystems known to have reduced or restricted extent (e.g. Collins Consulting 2025) while New Zealand expertise in RLE evaluations is built and standardised approaches to determining collapse thresholds are developed.

Using the RLE framework to assess the risk of ecosystem collapse has value in New Zealand and could be applied across all ecosystems and domains. This is because it is a well-documented process to identify the ecosystems most at risk of collapse. Preventing collapse in ecosystems at risk may in many cases be far more feasible and cost-effective than attempting to restore collapsed ecosystems, which highlights the importance of developing guardrails to prevent ecosystem collapse. Information from RLE evaluations can inform prioritisation for ecosystem protection and, where possible, restoration. If threats to the ecosystems most at risk are a result of ongoing human-induced reduction in their extent, or abiotic degradation or alteration of biotic processes caused by human activity, these issues could be addressed through policy or active management.

If assessments of the risk of ecosystem collapse are stymied because evaluations of any or all the RLE criteria are data deficient, this can serve to direct effort to fill gaps in our knowledge of ecosystem extent, threats to these ecosystems, and biotic disruptions. Pragmatism seems to determine most of the RLE assessments in the case studies we evaluated, because they used only a subset of the RLE criteria or sub-criteria. Limits and thresholds for assessing collapse risk with respect to environmental degradation and biotic disruption varied depending on the ecosystem and the strength of the existing research. Biotic disruption was far less often evaluated than distributional criteria. A major challenge for assessing biotic disruption is that its severity needs to be assessed across the entire extent of the ecosystem, but in many cases data are missing for some or most locations. RLE assessments can be repeated (e.g. at decadal intervals) to incorporate new data and determine the effectiveness of policy or management intervention.

New Zealand can make a significant global contribution to the assessment of risk to ecosystems, including evaluation of the risk of ecosystem collapse, by drawing on mātauranga Māori. Indigenous knowledge is increasingly recognised as filling gaps and enhancing understanding of ecological processes and can also contribute to defining historical baselines against which to evaluate, for example, change in an ecosystems distribution (RLE criterion A) or disruption of biotic processes (RLE criterion D) (e.g. Zent 2025). Moreover, the acknowledgement that people are part of ecosystems (as in te ao Māori; McAllister et al. 2019) is relevant for ecosystem collapse risk evaluation. For example, assessment of disruption of biotic processes in New Zealand forests that result from declines in kererū (*Hemiphaga novaeseelandiae*) could set thresholds for criteria about disruption of processes based on the effectiveness of this pigeon to disperse the seeds of forest trees over long distances (e.g. Wotton & Kelly 2012). In contrast, a perspective based on mātauranga Māori might set thresholds based on the body condition of kererū, the timing of its harvest as climate change affects fruiting patterns in a key tree species in its diet (toromiro, *Pectinopitys ferruginea*), and the capacity to sustain people's welfare through its harvest (Lyver et al. 2009).

This review shows that, internationally, the process of quantifying ecosystem collapse is still in its infancy. The Red List of Ecosystems provides a formalised process to conduct evaluations of the risk of ecosystem collapse across domains. As a headline indicator for reporting on the UN Convention on Biodiversity, New Zealand is obliged to use the RLE, so it makes sense to also explore its use at subnational (e.g. regional) scales for local decision-making.

Our review of international literature has also shown that determining thresholds for ecosystems that might predict their collapse is very seldom achieved, and that thresholds are usually apparent only after collapse. More evaluations of New Zealand ecosystems are needed to determine whether this applies here. It is very likely that case studies and the RLE process will reveal data gaps that are an impediment to evaluating the risk of ecosystem collapse. The RLE process gives a structured approach that will help determine, for any given ecosystem, the data gaps most critical to fill.

9 Recommendations

- Commission case studies to evaluate the risk of ecosystem collapse in freshwater and marine ecosystems, including the utility of indicators and collapse thresholds in these systems, and compare and contrast results with those from terrestrial ecosystems.
- Implementing the RLE process requires:
  - continued investment in a unified ecosystem typology and mapping of ecosystems, because these underpin assessments of ecosystem collapse risk
  - achieving a consensus on baseline dates to underpin RLE assessments, with peer review, by evaluating the advantages and disadvantages of using historical baselines of 1250 vs 1750 in determining collapse risk
  - formalising international peer review processes (e.g. drawing on international expertise).

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Declaration of the use of generative AI / AI-assisted technologies

During the preparation of this report the authors used SciSpace to sort and/or summarise scientific publications to identify relevant case studies and review articles on ecosystem collapse. After using this tool, summaries of case studies were reviewed by the authors and found to be inaccurate, so these summaries were not used and all original papers and reports were consulted. The content of this report was not generated using AI, aside from the reference list, and Figure 1, which were reformatted using ChatGPT.

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Appendix 1 – Red Listing of Ecosystems (RLE) case studies

Table A1. Reviewed case studies using the RLE framework to assess ecosystems collapse risk. Criteria for the ecosystem collapse risk assessment are A: reduction in geographic extent; B: restricted geographic distribution; C: environmental (abiotic) degradation; D: disruption of biotic processes or interactions; E: quantitative analysis

Global Typology Realm	Ecosystem	RLE Criterion					Reference
		A	B	C	D	E	
Freshwater	Connected wetlands of the Lake Eyre Basin	+	+	+	+	ne	Pisanu et al. 2015
Freshwater	Karst Rising-Spring Wetland Community, Australia	+	+	+	dd	dd	Keith et al. 2013
Freshwater	Glacial lakes, Myanmar	dd	+	dd	dd	ne	Murray et al. 2020
Freshwater	Raised bogs in Germany	+	+	+	ne	dd	Keith et al. 2013
Freshwater	Coastal sandstone upland swamps, Australia	+	+	+	+	dd	Keith et al. 2013
Freshwater/Marine	Chesapeake Bay, US	+	+	+	+	ne	Mahoney and Bishop 2017
Freshwater/Marine	Coorong lagoons and Murray Mouth inverse estuary, Australia	+	+	+	+	+	Keith et al. 2013
Marine	Antarctic shallow invertebrate dominated ecosystems	dd	+	+	dd	dd	Clark et al. 2015
Marine	Oyster Reef Ecosystem, Australia.	+	+	+	+	dd	Gillies et al. 2020
Marine	Seagrasses of Moreton Bay, Australia	+	+	+	+	ne	Sievers et al. 2020
Marine	European Native Oyster Reef Ecosystems	+	+	dd	+	na	zu Ermgassen et al. 2024
Marine	Seagrass Community of South Australia	+	+	+	dd	dd	Keith et al. 2013
Marine	Mangroves of New Zealand	+	+	dd	dd	ne	McNeill et al. 2024
Marine	Meso-American reef	+	+	+	+	+	Bland et al. 2017
Marine/Freshwater/Terrestrial	Saltmarshes of Moreton Bay, Australia	+	+	+	+	ne	Sievers et al. 2020
Marine/Freshwater/Terrestrial	Philippine fringe mangrove forests	+	+	ne	ne	ne	Marshall et al. 2018
Marine/terrestrial	Tidal flats of theYellow Sea	+	+	+	+	dd	Murray et al. 2015
Terrestrial	Eastern Stirling Range Montane Heath and Thicket, Australia	+	+	dd	+	ne	Barrett and Yates 2015
Terrestrial	Alpine snow patch herbfields, SE Australia	+	+	+	+	ne	Williams et al. 2015
Terrestrial	Gnarled Mossy Cloud Forest, Australia	+	+	+	dd	ne	Auld et al. 2015
Terrestrial	Coastal lowland rainforests of the Wet Tropics Bioregion, Australia	+	+	ne	ne	ne	Metcalfe and Lawson 2015
Terrestrial	Mountain Ash Forest in the Central Highlands of Victoria, Australia	+	+	+	+	+	Burns et al. 2015
Terrestrial	Cumberland Plain Woodland, Australia	+	+	dd	+	dd	Tozer et al. 2015
Terrestrial	Drylands of Tarim River Basin, China	+	+	+	ne	ne	Chen and Ma 2025
Terrestrial	Interior Wetbelt and Inland temperate rainforest, Canada	+	ne	+	+	+	DellaSala et al. 2021
Terrestrial	Mursala Island, Indonesia	ne	+	ne	ne	ne	Robiansyah et al. 2024

Notes: **+** indicates the criterion was assessed, with those cells highlighted in red informative for the overall Red Listing outcome; dd= data deficient; ne= not evaluated.

Table A2. Reviewed case studies using the RLE framework to assess ecosystems collapse risk, with examples of indicators used. Criteria for the ecosystem collapse risk assessment are as for Table A1. 'Criteria for EC risk' provides the criteria that were informative for the Red List outcome; in some cases other criteria were also assessed but were assigned a lower risk category

Global typology realm	Ecosystem example	Main threats identified	Criteria for EC risk	Description	Specific indicators or approach	Reference
Freshwater	Connected wetlands of the Lake Eyre Basin	Water extraction, reduced water flow, climate change, mining, water quality, livestock, invasive plants and animals	A, D	Increased water extraction and changes in flow regime are considered the major threats for Lake Eyre Basin. Waterholes may be affected by stock access, introduced animal and plant species and recreational activities, as well as localised pumping for pastoral and domestic water use.	Annual flows (creeks), waterbird diversity	Pisanu et al. 2015
Freshwater	Karst rising-spring wetland community	Sea-level rise	B, C	Karst rising-spring wetland has a restricted extent with evidence that the ecosystem is under continuing threat due to decrease in rainfall, groundwater discharge, and invasive species. Sea-level rise and saltwater intrusion arising from climate change are predicted to affect >85% of the ecosystem by 2070.	Decrease in groundwater discharge, future predictions of sea-level rise	Keith et al. 2013
Freshwater	Glacial lakes	Climate change	B	A basic assessment based on limited existing knowledge. Glacial lakes across the Himalayas are reported to be expanding in area in response to increased rates of glacier melt. No data were available to quantify increases. Similarly, no information was found that suggests a Myanmar-wide decline in extent. Despite being restricted to a small area in northern Myanmar, there was no evidence of ongoing decline in extent, environmental quality or biotic interactions (criteria A–D).	Decline in lake extent	Murray et al. 2020
Freshwater	Raised bogs in Germany	Habitat loss, increased nitrogen	A, C	Raised bogs in Germany have declined in extent by 98%, mainly caused by land drainage. They are subject to nutrient leakage from intensive agricultural use in their vicinity as well as aerial deposition of nutrients.	Reduced distribution, increased nitrogen concentration	Keith et al. 2013

Global typology realm	Ecosystem example	Main threats identified	Criteria for EC risk	Description	Specific indicators or approach	Reference
Freshwater	Coastal sandstone upland swamps, Australia	Extraction of coal and/or gas, changing fire regime, climate change	A, B, C	Assessed based on projected trends in future geographical distribution reduction and future environmental degradation (hydrological suitability), with a combination of relative severity and extent that is above the 'Endangered' threshold, under sub-criterion A2a and C2a.	Reduced and restricted distribution, with evidence of threatening processes, climate modes / climatic suitability, trend in woody resprouters	Keith et al. 2013
Freshwater /Marine	Chesapeake Bay	Habitat loss, sedimentation	D	Oyster habitat has declined by >90% compared with historical distributions, and abundance has declined by >85%	Decline in seagrass habitat, loss of oyster banks	Mahoney & Bishop 2017
Freshwater /Marine	Coorong lagoons and Murray Mouth inverse estuary	Climate change and water extraction	B, C, D, E	There is evidence of increasing salinity, and further environmental declines are forecast with a changing climate. A simulation (quantitative assessment of collapse risk) showed that a dry future climate projection with current extraction levels is always likely to result in ecological collapse.	Restricted location, evidence of continuing decline, annual average salinity, barrage volume, maximum salinity, spatial coverage of keystone species, quantitative simulation model	Keith et al. 2013
Marine	Antarctic shallow invertebrate-dominated ecosystems	Climate change, human activity (tourism)	B	Best available data suggest that shallow, ice-covered ecosystems are likely to be 'Near Threatened' to 'Vulnerable' in places, although the magnitude of risk is spatially variable and requires additional data to strengthen the assessment.	Change in sea-ice coverage	Clark et al. 2015
Marine	Oyster reef ecosystem of southern and eastern Australia.	Over-harvesting, dredging, habitat modification, land-use change, water quality, episodic flooding, diseases/ parasites, climate change	A, D	Ecosystem decline occurred over a 150-year period from 1800 to 1950, which coincided with the peak wild oyster harvest fishery, landscape modification, and industrialisation of coastal areas. Historical accounts describe extensive reef systems that were intensively harvested, causing total collapse. Though there were limited data to assess criterion D, there is considered sufficient evidence to deduce that severity related to loss of oyster biomass causing ecosystem decline is >90% of past biomass (all studies indicate ecosystem collapse through the loss of oysters).	Reduced distribution (historical), sedimentation, substrate simplification, qualitative assessment of disruption to biotic processes based on historical accounts of oyster abundance	Gillies et al. 2020

Global typology realm	Ecosystem example	Main threats identified	Criteria for EC risk	Description	Specific indicators or approach	Reference
Marine	Seagrasses of Moreton Bay	Sediment inputs, land-use change, climate change, eutrophication, fishing, physical damage	A, B, C, D	The status assigned is ‘Least concern’ because there has been no change in ecosystem extent, and recent stock reports suggest that population trends and catch rates of seagrass-associated species are largely not declining. Dugongs and sea turtles are ecologically important grazers of seagrass ecosystems, and their populations are unlikely to have declined much over the past 50 years.	Reduced extent, restricted extent, dissolved oxygen and Secchi depth, sea level rise, catch per effort for key species, trends in key wildlife species Extent, populations of key biota	Sievers et al. 2020
Marine	European native oyster reef ecosystems	Overexploitation /harvest, climate change, sedimentation, water salinity	A, B, D	This study concludes that European native oyster reef ecosystems have collapsed. While historically they persisted at large scales, they now occur mostly as scattered individuals, or very rarely in small clumps of a few square metres.	Reduced distribution, restricted distribution, abundance of key species, structural complexity, trophic diversity	zu Ermgassen et al. 2024
Marine	Seagrass community of South Australia	Agricultural run-off and pollution	A, C	Based on analyses of more than 45 years in sites assumed to be representative of the seagrass community in South Australia, there was an estimated decline in extent of 33–98%. The principal mechanism of environmental degradation is through decline in water quality caused by agricultural run-off, land-based pollution, and changes in the optimal conditions for seagrasses due to natural events.	Reduced distribution with evidence of continuing decline, evidence of threatening processes, increased nitrogen concentration	Keith et al. 2013
Marine	Mangroves of NZ	Reduced sedimentation rates, catchment management, climate change	A, B	Unique pressures and drivers of change are leading to mangrove range expansion rather than contraction. Under a high sea-level rise scenario >16% of the area currently covered by mangroves would be submerged by 2060. Possible changes to land use and sediment fluxes in catchments draining into estuaries represent further uncertainties in predictions.	Distribution (increased, and not restricted)	McNeill et al. 2024
Marine	Meso-American reef	Hurricanes, invasive species, harvest, pollution, climate change, disease, sedimentation, land-use change	C, D	The Meso-American Reef is considered to be at high risk from mass bleaching, ocean acidification, hurricanes, pollution, and fishing. Thresholds for biotic disruption and environmental degradation were set, and a simulation tested multiple future scenarios to assess quantitative risk of collapse.	Coral cover, mass bleaching rates, stochastic model	Bland et al. 2017

Global typology realm	Ecosystem example	Main threats identified	Criteria for EC risk	Description	Specific indicators or approach	Reference
Marine/ Freshwater /Terrestrial	Saltmarshes of Moreton Bay, SE Queensland	Land-use change, climate change, coastal development, sedimentation, fishing, pollution, biological invasions, physical disturbance	A, B	The area of saltmarsh was reduced because of anthropogenic drivers, including livestock grazing and coastal development.	Reduced distribution, with evidence of threatening processes, restricted distribution (1)	Sievers et al. 2020
Marine/ Freshwater /Terrestrial	Philippine fringe mangrove forests	Sea-level rise, deforestation for aquaculture	A, B	Assessed as least concern based on limited evidence of decline in distribution.	Reduced and restricted distributions	Marshall et al. 2018
Marine/ Terrestrial	Tidal flats of the Yellow Sea	Declines in sedimentation, coastal development, invasive species, pollution, harvest, erosion, climate change	A, C, D	Severe observed declines in the extent of the Yellow Sea tidal flat ecosystem over the past 50 years, as well as long-term declines in sediment delivery (considered a key abiotic process), determine its assessment as endangered (A1, C1, D1). Owing to the difficulty finding useful, long-term data sets for biotic and abiotic factors across the region, further work to identify data sources would improve confidence in the assessment.	Reduced distribution, with evidence of continuing decline, sediment discharge rates, declines in endemic fauna, declines in migratory species	Murray et al. 2015
Terrestrial	Eastern Stirling Range montane heath and thicket	Plant pathogen (*Phytophthora*), fire, climate change,	B, D	Long-term monitoring and historical sources demonstrate decline in characteristic and functionally important plant species due to *Phytopthera* infestation. The pathogen is considered impossible to eliminate.	Reduced or restricted distribution, decline in species susceptible to pathogen, change in fire return intervals,	Barrett and Yates 2015
Terrestrial	Alpine snow patch herbfields, SE Australia	Weed invasion, climate change	B	Although cover of exotic species has increased, it is not yet causing concern. It was assumed that transition from snow-patch herbfield to heathland will occur when shrub cover reaches 50% ± 25%.	Change in species composition (increased shrub cover), restricted distribution with evidence of threatening processes, snow cover	Williams et al. 2015

Global typology realm	Ecosystem example	Main threats identified	Criteria for EC risk	Description	Specific indicators or approach	Reference
Terrestrial	Gnarled Mossy Cloud Forest, Lord Howe Island, Australia	Climate change, land-use change, invasive species	B	Long-term monitoring data on both rainfall and cloud cover provide evidence for a continuing decline in abiotic components of the ecosystem. The ecosystem is susceptible to stochastic processes such as tropical cyclones and wildfire within a short time period in an uncertain future.	restricted distribution with evidence of threatening processes, future restricted distribution based upon changing environment, trends in rainfall, cloud cover	Auld et al. 2015
Terrestrial	Coastal lowland rainforests of the Wet Tropics Bioregion, Queensland, Australia	Forest clearance, cyclones, invasive weeds, fragmentation, herbivory, fire regimes, hydrological changes, climate change	A, B	Remaining coastal lowland rainforest exists in protected areas, but fragmentation continues through road, rail and power corridor expansion having significant impacts on biodiversity and ecosystem function. There has been a decline in extent of at least 60% since the late 19th century. There is evidence of disruption of biotic interactions due to development, invasive species, cyclones, and habitat modification by anthropogenic disturbance.	Reduced distribution, and restricted distribution, with evidence of threatening processes	Metcalfe & Lawson 2015
Terrestrial	Mountain ash forest in the Central Highlands of Victoria, SE Australia	Forest fire/ logging	E	Intensive logging and repeated fires have removed or degraded key habitat structures and keystone elements in mountain ash forests, producing a hidden, multidecadal decline in ecosystem condition and biodiversity and disrupting processes in ways that make full recovery unlikely without early management intervention (Burns et al. 2015).	Loss of structural complexity (large hollow-bearing trees)	Burns et al. 2015
Terrestrial	Cumberland Plain woodland, New South Wales, Australia	Historical clearing, forest fragmentation, weed invasion, fire regime, herbivores, logging, altered nutrient cycles	A	Based on the impact of land clearing, the ecosystem is considered ‘Critically Endangered’ due to the extent of historical decline (92–94%)	Reduced distribution, restricted distribution, with evidence of threatening processes, weed cover	Tozer et al. 2015

Global typology realm	Ecosystem example	Main threats identified	Criteria for EC risk	Description	Specific indicators or approach	Reference
Terrestrial	Drylands of Tarim River Basin, China – includes multiple vegetation types	Land-use change, increasing aridity	A, B, C	Vegetation was threatened due to land-cover change, restricted geographical ranges, and projected climate change. Together, threatened vegetation accounted for 72.8 % of the total area assessed, mainly located in low-elevation regions with intensive land use.	Mapped extent and change, climate models and habitat suitability, land suitable for development for assessing future impacts	Chen & Ma 2025
Terrestrial	Interior wetbelt and inland temperate rainforest, British Columbia, Canada	Land-use change, logging, land conversion	A, C, D, E	Forest types were assessed using spatial and biotic criteria. Collapse was predicted based on a model using historical logging rates to estimate the proportion of unfragmented forest that might remain after 50 years. The assessment relies on an assumption that if only small patches of forest remain, their ecosystem functions are so fragmented that functional collapse of primary areas is imminent.	Logging rates, habitat suitability for wildlife	DellaSala et al. 2021
Terrestrial	Mursala Island, Indonesia	Logging, land conversion	B	Assesses tropical lowland rainforest on Mursala Island against criterion B (restricted distribution). Logging is cited as the main threat, but was not quantified.	Restricted distribution, with evidence of logging	Robiansyah et al. 2024

Appendix 2 – Mackenzie District vegetation history

This table shows Mackenzie District vegetation history and vegetation states over time. Descriptions of vegetation at approximate timepoints since 800 summarise studies cited in each row (direct quotations denoted*).

Main state	Year (AD)	Descriptor	References
Forest/tall tussock grassland.	800	'*Halocarpus* … formed a considerable proportion of the vegetation cover. Snow tussock grassland … common'*. Legacy of earthquake disturbance and some natural fire disturbance.	McGlone & Moar 1998*
Forest / tall tussock grassland	1200	Beech profits from increasing natural fire disturbance at the expense of *Halocarpus*. *Phyllocladus* and *Halocarpus* formed extensive stands on droughty soils in the glacial outwash surfaces.	McGlone & Moar 1998; McGlone 2004
Short tussock grassland / forest	1350	After Māori settlement and fire, grassland dominated by short tussocks and grasses (*Poa colensoi* and *Festuca novae-zealandiae*). Local patches of *Chionochloa*-dominated grasslands with *Aciphylla*. Beech and *Halocarpus* mostly destroyed by fire. Local patches of *Halocarpus* (with some *Phyllocladus*) survive.	Cockayne 1928; Calder 1961; Connor 1964; Connor & Macrae 1969; McGlone & Moar 1998; McGlone 2004; Vandergoes et al. 2018
Tall tussock grassland / forest	1750	*Chionochloa*-dominated grasslands with *Aciphylla* and low shrubs dominant. 'Tussock grassland and scrub… ran in a continuous sweep from valley floor to alpine herbfields'*. It may have been '200 years … before a close-canopied *C. rigida* grassland was formed in the Mackenzie Country' after initial dominance by *Poa colensoi* and *Festuca novae-zealandiae*. Local patches of *Halocarpus* (with some *Phyllocladus*) on droughty soils.	Connor & Macrae 1969*; Wardle 1991; McGlone & Moar 1998; McGlone 2004
Tall tussock grassland / forest	1850	'Large *Chionochloa* tussocks were widely dominant on the older, more acid soils', an 'abundance of large *Aciphylla*', 'shrubs, forbs, and other grasses, and … deep litter'*. Non-native herbs (e.g. *Rumex acetosella*) colonise. Local patches of *Halocarpus* (with some *Phyllocladus*) on droughty soils.	Wardle 1991*; Vandergoes et al. 2018
Tall tussock grassland / (very little) forest	1880	Repeated burning and grazing by extremely high densities of sheep resulted in major reduction in large *Aciphylla*, shrubs, and the most palatable herbs (e.g. *Gingidia montana*), and reduction in biomass of widely dominant *Chionochloa rigida*, almost complete extirpation of previously rare *C. rubra*, increasing dominance of *Festuca novae-zealandiae*, and increasing presence of non-native species (e.g. *Rumex acetosella*). Reduction in extent of any remaining forest fragments.	Connor & Macrae 1969; Wardle 1991; McGlone & Moar 1998
Short tussock grassland / (very little) forest	c. 1920	Repeated burning and further grazing by sheep and rabbits (present since c. 1890s) resulted in widespread extirpation or rarity of *Chionochloa* tussocks (absent or 'very sparsely dotted throughout…in an erratic manner'*) and its replacement by short tussock, with *Festuca novae-zealandiae* as the main dominant species, along with native and non-native herbs. ('No precise timing for the change from tall-tussock to fescue-tussock grassland can be given, but much of it has taken place within the century since settlement, as a direct result of heavy grazing following large-scale burning'*). Reduced tall tussock communities. Hugely reduced native forest fragments.	Cockayne 1928; Connor & Macrae 1969*

Main state	Year (AD)	Descriptor	References
Herbfield/short tussock grassland / (very little) forest	1960	Biomass of short tussocks depleted by grazing; fire regulated since 1945. Increasing biomass of naturalised and planted non-native grasses and herbs. On 'alluvial terraces and sunny slopes tussock grassland had been replaced by open communities of low-growing perennial dicots (mainly native) and vernal annuals and tall biennials (mostly non-native)'*. Reduced tall tussock communities. Hugely reduced native forest fragments.	Connor 1964; Connor & Macrae 1969; Hunter & Douglas 1984; Wardle 1991*
Herbfield / short tussock grassland / (very little) forest	1970	Low biomass of short tussocks with continued but much reduced levels of grazing by sheep, rabbits, and hares. Increasing biomass of naturalised and planted non-native grasses and herbs. Non-native conifers (multiple species) increasing from 1960 onwards. Constraining of braided rivers and reduced flow, and altered lake levels with hydro schemes. Reduced tall tussock communities. Alluvial non-native willow forests develop. Hugely reduced native forest fragments.	Beauchamp 1962; Hunter & Douglas 1984; Vandergoes et al. 2018
Herbfield / short tussock grassland / (very little) forest	1990	Short tussock, with equal frequencies of *Pilosella offincinarum* and non-native grasses grazed by sheep, hares, and rabbits. Reduced tall tussock communities. Increasing naturalisation of non-native conifers. Alluvial non-native willow forests. Hugely reduced native forest fragments.	Moller et al. 1997; Meurk et al. 2002; Scott et al. 2001; Walker & Lee 2002
Herbfield / intensively farmed non-native grasses / short tussock grassland / (very little) native forest / non-native forest	2010	Short tussock, with equal frequencies of *Pilosella offincinarum* and non-native grasses grazed by sheep, hares, and rabbits. Local fertilised, irrigated pasture for dairy cattle. Reduced tall tussock communities. Local non-native conifer forest patches. Alluvial non-native willow forests. Hugely reduced native forest fragments.	Weeks et al. 2013
Herbfield / intensively farmed non-native grasses / short tussock grassland / (very little) native forest / non-native forest / (very little) tall tussock grassland	2025	Short tussock, with equal frequencies of *Pilosella offincinarum* and non-native grasses. Local fertilised, irrigated pasture for dairy. Reduced tall tussock communities. Reduction in extent of non-native conifer forest patches, replaced by non-native grasses. Alluvial non-native willow forests. Hugely reduced native forest fragments.	

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