

6.6 Indigenous plant dominance

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State of Knowledge of the “Indigenous plant dominance” attribute: Excellent / well established – comprehensive analysis/syntheses; multiple studies agree.

The evidence and use of indigenous plant dominance for understanding and reporting on environmental change is overwhelming, and is strongly linked to all components of ecosystem integrity and to many other attributes. More specifically for indigenous dominance, **ecological integrity** (as defined from the NBA Bill) is the ability of the natural environment to support and maintain:

- representation: the appropriate spatial distribution of indigenous species,
- composition: the abundance of indigenous species within different communities,
- structure: the effects of indigenous dominance on physical and abiotic properties of ecosystems, and
- functions: the effects of indigenous dominance on ecological and ecosystem processes.

Indigenous (native) plant dominance is considered an essential biodiversity variable (EBV; see Pereira et al. 2016). Dominance can be measured in multiple ways including cover, biomass, population density, height or size. Dominance measures vary among taxa and systems. In addition, these measures reflect different aspects of abundance or species’ effects on representation, composition, structure and function. The spatial scale over which dominance is measured ranges from the biomass of individual trees, through to national-scale reporting of distribution or abundance of specific taxa for biodiversity reporting and carbon accounting.

There are some established targets or limits for indigenous plant species dominance such as maximum height determined for land clearance. Our knowledge and ability to establish targets or limits for indigenous dominance related to ecological functioning or ecosystem processes is rare, because this relies on understanding the per unit impacts of species on ecological process and dominance of ecological process and function (Lee et al. 2005), not biomass or abundance *per se*. Nonetheless, indigenous plant species dominance is one of the most important attributes or indicators of ecological integrity for terrestrial systems.

Part A—Attribute and method

A1. How does the attribute relate to ecological integrity or human health?

In general, increasing indigenous plant species dominance is considered positively related to all aspects of ecological integrity (Lee et al. 2005; McGlone et al. 2020). Indigenous plant dominance directly relates to the ecological integrity of terrestrial ecosystems through composition (i.e., by directly influencing community composition) and structure (i.e., by contributing to vegetation structure). Dominance relates to functions in multiple ways through controlling key ecosystem processes such as primary productivity, nutrient cycling and decomposition, or specific ecological processes like habitat or resource provision for other taxa. Dominance can also be related to representation if changes in abundance or cover of species are evaluated over larger spatial scales (e.g., changes in the cover or biomass of species across a population, sites or across the overall species distribution). An important concept reflecting how dominance varies spatially are abundance-occupancy relationships, which capture the distribution of species throughout their range.

Dominance in some cases is also related to human health through landscape values (e.g., tussock grasslands considered outstanding vegetation in the McKenzie basin), cultural values (e.g., dominance of taonga species), and in a few cases, economic wellbeing (e.g., mānuka honey).

There are some important exceptions to the generalisation that indigenous plant dominance is directly and positively related to ecological integrity. Some examples include:

- The deliberate anthropogenic introduction of, and sometimes invasion by, native plant species beyond their historical distribution (e.g., karaka in coastal habitats).
- Maintenance of indigenous dominance through deliberate management (e.g., maintenance of mānuka monocultures for honey production through repeated disturbance).

In these cases, measures of indigenous dominance alone are insufficient to understand ecological integrity; additional knowledge of ecosystems, management or pressures is required.

A2. What is the evidence of impact on (a) ecological integrity or (b) human health? What is the spatial extent and magnitude of degradation?

There is strong evidence that changes in indigenous plant dominance affects ecological integrity and human health, but these effects vary widely among regions and species. Indigenous species dominance is also considered an essential biodiversity variable (EBV), and as a consequence is widely used to report on state and trend of biodiversity and ecosystem condition (Bellingham et al. 2020, McGlone et al. 2020; see also A4 below). International and national concerns about long-term declines in biodiversity have driven major data collection and reporting efforts, most of which capture data for indigenous plant species dominance (see A5 and C2 below). Dominant indigenous plant species control many ecosystem properties and processes across different vegetation types such as forests and wetlands (see Allen et al. 2003, 2013, Tierney et al. 2009).

Changes in indigenous plant dominance, both in spatial extent and abundance across scales, are relatively well documented and understood. Shifts in the abundance of many indigenous plant

species or vegetation dominated by indigenous species reflect major landscape-scale processes including:

- Altered disturbance regimes, particularly fire: indigenous forest cover and many indigenous plant species declined sharply following anthropogenic fires (Perry et al. 2014, McWethy et al. 2014);
- Land use change and intensification: historical and ongoing land management often replaces indigenous plant species with non-native taxa for a range of purposes or services (Atkinson et al. 1993, Craig et al. 2000, Moller et al. 2008). Land use and management has also fundamentally altered nutrient availability and hydrology or soil properties which often favours dominance of non-native plant species having a strategy of faster growth and resource use compared to most indigenous plant species (Brandt et al. 2021);
- Introduction and impacts of non-native mammalian herbivores: historical and ongoing impacts of different invasive herbivores can drive declines in preferred (palatable, or selected) species but also increases in avoided (unpalatable) species (Coomes et al. 2003, Peltzer et al. 2014).

Additional drivers of change in indigenous species dominance are likely, but poorly understood. These include historical selective harvesting (logging) of a subset of indigenous tree species (McGlone et al. 2022), declines or loss of other indigenous species that affect plant population processes (e.g., loss of mutualistic species such as pollinators), long-term impacts of vegetation fragmentation creating extinction debts (Ewers et al. 2006), and ongoing increases in the number and abundance of non-native environmental weeds that can suppress or displace co-occurring indigenous species (Brandt et al. 2021; PCE2021).

Overall, there is low evidence relating indigenous plant species abundance directly to human health. This is driven by a paucity of information available for most taxa; our understanding and data are restricted to a few species such as:

- Abundance of *Coriaria arborea*/tutu because of potential tutin poisoning of honey.
- Mānuka underpinning honey production and economic wellbeing.
- Declines in major taonga species such as Kauri (*Agathis australis*), linked to multiple cultural values of forest ecosystems (Waipara et al. 2013).
- Declines in broader sets of dominant species such as myrtle rust, potentially having important social, cultural and economic impacts (Lambert et al. 2018).
- Loss of podocarps and related cultural impacts (Lyver et al. 2017a,b).
- Perceptions of loss for wild food and thus human health.

See also details in other templates which consider loss in extent or dominance of indigenous plant species: lowland forest extent, wetland extent, and canopy tree dieback extent.

A3. What has been the pace and trajectory of change in this attribute, and what do we expect in the future 10 - 30 years under the status quo? Are impacts reversible or irreversible (within a generation)?

Dominance of many indigenous plant species has changed profoundly historically, and is expected to continue changing over the coming decades, although the pace and trajectory of change will differ widely among taxa and regions.

Historical baselines of plant dominance together with paleo records and biocultural observations provide overwhelming evidence that native plant dominance has declined sharply for many terrestrial ecosystems from pre-human baselines. These declines range from trivial (alpine vegetation) to >95% declines in peat bogs (see Burge's wetland extent attribute) and some lowland forest systems (see Walker's assessment of lowland forest extent attribute). Broad-scale declines in dominance were driven initially by anthropogenic fires, and subsequently by land use changes and introduced pests, weeds and diseases (Perry et al. 2014). In some areas indigenous plant dominance has recovered during the previous decades through changing land management, fire suppression, and discontinuation of harvesting (Allen et al. 2013).

Over the coming decades, dominance of many indigenous plant species is expected to maintain a status quo or increase slightly at the national scale. This is being driven by relatively widespread interest in managing marginal or previously-disturbed vegetation for carbon resulting in increased areas of successional vegetation (Forbes et al. 2023). Moreover, much of this management also improves biodiversity associated with C management, either implicitly (i.e., many woody successions include multiple indigenous plant species) or explicitly (e.g., through market premiums in ETS, or potentially through future biodiversity credits schemes). Both national and regional incentives are driving broad restoration efforts in some wetlands, riparian marginal strips, and enrichment planting; and the effects of these schemes should persist for decades.

Declines in dominance are expected for some indigenous plant species and regions, including taxa susceptible to emerging pathogens (e.g., myrtle rust impacts on myrtaceous plant species that include dominant forest tree species), replacement of some native taxa by non-native species, and by increasing weather extremes and associated disturbances such as cyclones and fire (see also tree canopy dieback extent attribute).

Evaluating current state and change in indigenous plant species dominance requires an explicit consideration of scale. At the national scale, past land use and intensification, species introductions, and landscape-scale changes have reduced the extent and condition of indigenous vegetation in most regions, and altered the distribution and abundance of many common indigenous plant species. These historical legacies have set the trajectory and scale of change possible for the next 30 years and much longer, and are not reversible in the short-term. However, at regional or local scales, changes in dominance can respond far more quickly and are often the focus of data collection efforts and reporting (discussed below). Natural ecosystems and dominance can change quickly, even with natural disturbances (e.g., the 2016 Kaikoura earthquake had immediate impacts on dominance, but recovery will occur over decades; see also Allen et al. 1999).

The pace and trajectory of change in plant species dominance has been evaluated across scales. Nationally, biodiversity monitoring and carbon accounting efforts capture repeated measures of abundance for most indigenous forests and shrublands over the past few decades (e.g., using DOC tier one monitoring). Regional monitoring efforts and plot networks have been established and provide important baselines and often include ancillary information for interpreting change in species dominance (Richardson et al. 2024). Longer-term changes are commonly evaluated at the local or catchment scale, because this is the scale at which management and community interests often emerge.

Overall, long-term (i.e., decadal) change in the dominance of species is the norm in dynamic systems, but there are well established efforts to monitor state and change in many indigenous plant species. Coverage of non-woody vegetation less well monitored (e.g., in alpine systems or marginal/successional systems). The ongoing challenge is assessing these changes against shifting baselines in disturbance regime, climate and other pressures like pests and pathogens (Bellingham et al. 2020, Lyver et al. 2021).

A4-(i) What monitoring is currently done and how is it reported? (e.g., is there a standard, and how consistently is it used, who is monitoring for what purpose)? Is there a consensus on the most appropriate measurement method?

Development of major biodiversity and carbon monitoring and reporting efforts over decades have generated **excellent monitoring standards and methods for assessing dominance of indigenous plant species. NHMS developed by DOC** (Lee et al. 2005) built on plot-based assessments of vegetation condition and change (e.g., Bellingham et al. 2000) provide detailed methods, standards and interpretation of biodiversity state and change across spatial scales. In addition, many of these methods are consistent with international efforts (EBVs) and can be used to quantify ecological integrity (McGlone et al. 2020). Ongoing refinement and updating of these methods offers opportunities to leverage data collection, monitoring and use. For example, DOC's tier one monitoring of terrestrial vegetation explicitly measures dominance and representativeness of indigenous plant species (measures for maintaining ecosystem composition, including 'species composition and diversity' and 'Species occupancy of natural range', (Bellingham et al. 2020)).

Excellent compositional data have been collected nationally as a consequence of multiple long-term monitoring efforts for biodiversity, vegetation assessment, soil characterisation, and carbon estimates (Bellingham et al. 2020). Many structural attributes are collected at the same time as compositional data. Structure is driven across scales by abundance and dominance of plant species within sites, reflected by numerous measures of biomass, vertical structure, cover (including in height tiers). Terrestrial vegetation is relatively well-characterised, and quantitatively-derived communities are available (e.g., Wiser et al. 2011). Major national and regional plot networks, monitoring efforts, and land cover spatial data provide fundamental data for quantifying and reporting changes in indigenous plant species dominance (or proxy measures such as vegetation cover class). Cover is the most consistent and commonly-reported measure of dominance (Bellingham et al. 2020). Biomass, density, height are also reported for specific purposes, vegetation-types, or regions. Dominance estimated by proportional contribution to community richness is fraught, because it does not relate strongly to the components of ecological integrity.

Many plot-based monitoring efforts capture information on population abundance and structure and can provide additional insights into population processes at site or larger spatial scales. For example, changes in tree size class distributions both determine forest physical structure, but also reflect habitat quality, effects of disturbance, and numerous ecological functions when considered with functional traits or additional attributes such as disturbance or pest abundance (Peltzer et al. 2014). Similarly, population size or structure is often used as a target or limit for species, both to understand declines or recovery of native species.

Standard methods are widely used but also vary in their application among regions. National standards for biodiversity and carbon are derived from plot-based (point) measures of vegetation composition and structure. This information is a crucial part of Te Mana o te Taiao; regional reporting on biodiversity by RCs and territorial authorities; and LUCAS (Bellingham et al. 2016, 2020). The

practical application and ability to apply these standards varies among regions. Tier one measurements used by councils and central government is standardised and widely used but requires point-based measurements. Data are remeasured currently on a 5-yearly return interval across PCL (but not private land), although discussions are in progress for pushing this out to a 10-yr return interval. These methods are applied by Greater Wellington Council across all vegetation-types, by Bay of Plenty/Auckland at the regional scale, and at the catchment or local scales for other regions. Although there is broad consistency in the methods used by different agencies, the spatial coverage and repeat measurement are not consistently applied.

Most measurements link to DOC tier one measurement across all landscapes but are not done in some land cover classes such as pastures or plantations. For example, landscape-scale structure has been relatively well characterised for forests and wetlands, but not for many other systems including naturally rare ecosystems (Williams et al. 2007, Holdaway et al. 2017). Furthermore, plot- or point-based measures are usually amalgamated into land use or cover classes for reporting purposes (Cieraad et al. 2015, Dymond et al 2017). Use of these data for reporting purposes span:

- Convention on Biological Diversity (CBD) reporting.
- MfE State of Environment reporting.
- biocultural monitoring (see B5 below).
- RC regional biodiversity reporting.
- catchment or site-based assessment of changes in indigenous dominance.

Both mapped changes in land use or cover of vegetation classes dominated by indigenous species are commonly reported, but have some inconsistencies in underlying data. For example, the resolution of mapping and ground based validation is low (Cieraad et al. 2015) partly because there are underlying errors in threshold and detection from imagery or vegetation classification (e.g., between versions of LCDB). These underlying assumptions or errors require uncertainty estimates for mapping, both false positives and negatives, that are known but rarely implemented in practice. A specific example: 'depleted grassland' includes a wide range of indigenous dominance, and this can vary crucially among different height tiers, at the extreme, dominance by tussocks at 30cm height but non-native species like *Hieracium* spp at the ground level (Weeks et al. 2013). Another example is for Auckland, where LCDB spatial resolution is considered too coarse to report changes in peri-urban fringe, or equally, smaller increases through restoration.

An ongoing challenge for reporting changes in indigenous dominance is linking plot-based data and spatial (mapped) information across scales. Emerging technologies like high density LiDAR and high-resolution hyperspectral imagery can be used to quantify structure, but cannot yet distinguish species without ground-based validation (see also Wiser et al. 2021, Ye et al. 2021). Most commonly, the default reporting units are based on LCDB, but reporting could also be done linking point-based measures with other mapped areas or cover classes within LENZ, qualitative ecosystem typologies, or more quantitative typologies (i.e., like Cieraad et al. 2015). These issues could be resolved for some systems, for example, by assessing if declines in wetland extent are mirrored by declines in indigenous plant species dominance. Including both plot-based and spatial data from remote sensing for assessing biodiversity change is the approach championed internationally through EBV development (Bellingham et al. 2020).

A4-(ii) Are there any implementation issues such as accessing privately owned land to collect repeat samples for regulatory informing purposes?

Implementation of data collection is well established through major monitoring efforts such as LUCAS and regional plot networks (Richardson et al. 2024). Practical or logistical considerations (not barriers *per se*) include:

- Botanical skills (for identifying indigenous taxa) and maintenance of qualified personnel (i.e., for field measurements, data collection, and analyses).
- Provision is in law via the RMA to collect data, but can be logistically difficult for some sites or communities.
- Repeated samples to evaluate changes in dominance require databases, long-term archiving, and access.
- Increasingly, transparent documentation of data processing, analyses and interpretation is required or expected (for both reporting and publication purposes). Similarly, explicit evaluation of assumptions and uncertainty in the data or analyses are required. Remote sensing methods and new technologies (e.g., eDNA) are not immune to these issues; all increasingly require informed consent for data collection, analyses and reporting.
- Intellectual property and data sovereignty issues are a potential barrier to data collection, use and access, and require ongoing consideration as part of monitoring efforts.
- A practical barrier is the lack of sustained/long-term commitment for collecting the primary data by most RCs through lack of funding or prioritisation of efforts elsewhere (see PCE 2019, 2020).
- Current monitoring efforts do not have complete national coverage. There is a bias in data collection against lowlands and rapidly changing (marginal) land use classes. Most reporting has focussed on PCL because one lead agency (DOC) implements monitoring, where potentially changes in indigenous plant dominance have the slowest/modest change. Spatial coverage of data in other land use classes is poor because of multiple agencies involved, has lower priority for many regions compared to other competing issues (e.g., water issues).

A4-(iii) What are the costs associated with monitoring the attribute? This includes up-front costs to set up for monitoring (e.g., purchase of equipment) and on-going operational costs (e.g., analysis of samples).

Costs per plot including data management are well discoverable from DOC and RCs, but do not distinguish costs for measurement of indigenous plant species dominance from other measurements (e.g., plant diversity, pellet counts, deadwood assessment) carried out at the same time. Aside from central and regional government, multiple agencies have the infrastructure, skills and ability to monitor this attribute including Manaaki Whenua and Wildlands. Additional costs for quality assurance of data collection and analyses are also available (e.g., from validation of DOC tier one measurements).

Detailed costs for other monitoring approaches such as remote sensing data are available, but heavily depend on the provider, quality of data collected, and highly variable costs associated with data management and processing or analyses. Costs of data acquisition are available, but depend on provider and data quality captured. Additional costs of specialist skills for data processing, analyses and interpretation are highly variable depending on the scale and reporting needs; normally the purpose is not to monitor indigenous plant dominance but to estimate land (vegetation) cover in which some cover classes are dominated by indigenous species.

A5. Are there examples of this being monitored by Iwi/Māori? If so, by who and how?

There are several examples of indigenous plant dominance being monitored. A few of these include:

- Forest diversity and condition monitoring in the Urewera's (e.g., Lyver et al. 2017) as well as shifting baselines in abundance of species and how communities respond or manage.
- Surveillance and impacts of Kauri dieback (see canopy tree dieback attribute).
- Monitoring of pingao abundances as a weaving resources.
- Mānuka as a resource for honey/economic activities (e.g., including Māori-owned businesses such as manawa honey).
- Rongoā plant species abundance (widespread).
- Abundance of dominant plant species such as raupō in wetland restoration (e.g., the Awarua-Waituna Ramsar site, Southland).
- Titi islands. Indigenous plant dominance but by different species because of burning regime associated with muttonbirding on these islands.

Some general considerations for monitoring effects by Iwi/Māori include:

- Standard (i.e., standardised tier one) measures do not apply. Communities do place-based monitoring, decide on what to measure, for what purpose. Fitness of purpose for scaling up monitoring efforts is not a major driver.
- 'Limits and targets' for abundance are set by communities.
- Biocultural monitoring (Lyver et al. 2018, Harcourt et al. 2022, Pou et al 2022) includes more integrated assessment of condition rather than focussing on a single attribute, but these can be used to better interpret change and impact.
- For Māori, everything is connected - biodiversity outcomes are important but cannot be considered separately from socio-cultural outcomes.

A6. Are there known correlations or relationships between this attribute and other attribute(s), and what are the nature of these relationships?

Indigenous plant species dominance is a unifying attribute across terrestrial ecosystems, and is closely linked to other attributes. Lowland indigenous forest and wetland extent are both directly, positively related to this attribute. The spatial distribution and representativeness of indigenous

plant species dominance and other terrestrial attributes are tightly coupled because they respond (negatively) to most of the same drivers of change, i.e., disturbance history, land use change and management, and the impacts of biological invaders.

Part B—Current state and allocation options

B1. What is the current state of the attribute?

The current state and recent change (i.e., over past few decades) in indigenous vegetation cover is well understood at the national scale. Excellent national monitoring data for indigenous plant species is available through tier one DOC monitoring, spatial change of indigenous-dominated vegetation through multiple updates to LCDB, and monitoring effects by many Regional Councils. Since human arrival in New Zealand, indigenous vegetation cover overall is reduced to approximately 44% of New Zealand. However, national and regional monitoring efforts show large variation in the scale of lost indigenous vegetation cover with the greatest declines driven initially by fire, and subsequently by land use changes in low elevation environments (Perry et al. 2014; see also lowland forest extent attribute). For example, the east coast of the South Island and most low-lying areas of the North Island have lost most of their indigenous vegetation cover. Changes in the abundance of individual indigenous plant species can also be quantified, but both spatial resolution of the data, some bias in monitoring efforts toward woody vegetation, and differences among regions in the drivers of species abundance changes (both positive and negative) make detecting changes at the scale of a decade modest for many plant species.

Representation is a major underpinning measure for NHMS and biodiversity monitoring schemes (e.g., Lee et al. 2005, Walker et al. 2016, Bellingham et al. 2020, McGlone et al 2020;). Some of the most widely adopted approaches are large-scale mapping of representative environments derived from multiple environmental, climatic and landscape features, but not diversity (e.g., LENZ, TEC, NZ data stack). Moreover, greater inclusion of biodiversity into ecological representation is likely in the future, given this is an Essential Biodiversity Variable (EBV) (see also Schmeller et al. 2018).

Some indigenous plant species can also be invasive, increasing in dominance through land use change, anthropogenically-altered disturbance regimes, or deliberate movement of species. Some examples include:

- The expansion of native tussock grass species following historical burning and removal of woody vegetation cover (Walker and Lee 2000).
- The deliberate introduction of the native tree species *Corynocarpus laevigatus* (karaka) that is now considered invasive outside of its historically naturally distribution (Costall et al. 2006).

Overall, there are excellent, long-term data available for determining current abundance of many indigenous plant species or vegetation-types. This provides a robust base of evidence for developing indigenous plant species dominance as an attribute for EI.

B2. Are there known natural reference states described for New Zealand that could inform management or allocation options?

Many areas of indigenous-dominated vegetation have baseline information that can be derived from previous measurements or monitoring, relative intact/undisturbed areas of comparable vegetation, and through historical or paleoecological evidence for abundance and distribution of species. In some areas of highest indigenous vegetation loss, little information is available for natural reference states (e.g., Mackenzie basin pre-fire, many lowland forest areas of the eastern South Island). For long-modified areas having little to no reference state based on monitoring data, paleoecological approaches can provide historical baselines for many species (e.g., Wilmshurst et al. 2007, McWethy et al. 2014, Dietl et al. 2015).

Many drivers of changes for indigenous plant species dominance are chronic, driven by ongoing damage by pests, weeds and diseases. Understanding the speed and magnitude of change in indigenous plant species, and how these respond to different management or allocation options, is a major ongoing effort at national and regional scales.

B3. Are there any existing numeric or narrative bands described for this attribute? Are there any levels used in other jurisdictions that could inform bands? (e.g., US EPA, Biodiversity Convention, ANZECC, Regional Council set limit)

There are several numeric and narrative bands established for the cover of indigenous-dominated vegetation cover, and in a few cases, for dominance of species such as:

- A threshold of LCDB categories classified as indigenous dominance (Cieraad et al. 2015).
- Different bands or levels of indigenous dominance as cover are commonly reported (e.g., by RCs, Bellingham et al. 2016).
- Thresholds of indigenous vegetation have been developed at the property and catchment scale of minimum 15% cover, and ideally 30% cover (e.g., Rout et al. 2021).
- Dominance considered as vegetation height has been applied to restrict land clearance of indigenous woody vegetation to <6m. Maximum extent and magnitude of indigenous vegetation clearance is provided in the NES-PF (e.g., <1.5% of area deemed indigenous vegetation, or <30% damage to indigenous tree crowns);
- Implied thresholds from 'intact' to 'degraded' status of indigenous vegetation are usually driven by declines in native vegetation, or sometimes increases in weed cover (e.g., due to wilding pine invasion; LCDB classes for grasslands, Weeks et al. 2013);
- There are quantifiable tipping points with fire for successional woody vegetation where risk is a non-linear function of vegetation biomass (Perry et al. 2014, Taylor et al. 2017);
- Internationally, there are many different limits and narrative bands established for indigenous plant species (or biodiversity more generally); there is an entire body of literature on this that cannot be covered here, but provides additional conceptual and practical guidance for establishing limits or thresholds for terrestrial biodiversity (e.g., Nicholson et al. 2021).

Nearly all these approaches require species- and system-specific information to set a limit or threshold. Some additional considerations:

- In some cases the potential future contribution of species is used, for example, presence of a late successional tree species at a site is used to classify the potential for the site to become indigenous forest over time (Mason et al. 2010).
- Successional or other directional changes in indigenous plant dominance are widespread, including the long-term decline of broad-leaved tree species whereas emergent podocarps persistent over the long-term following major disturbance (Richardson et al. 2020).
- Many terrestrial vegetation-types contain a mixture of indigenous and non-native species, and the threshold or limits of these reflect the social or cultural value of the vegetation (e.g., retention or restoration of indigenous woody plant species on farms; retention of some weed species as ‘nurse crops’ for native plant species).
- There are ongoing consequences of land use change and management effects, but allocation options and priorities are not well co-ordinated, particularly between biodiversity and biosecurity management objectives.

B4. Are there any known thresholds or tipping points that relate to specific effects on ecological integrity or human health?

There are established or potential thresholds, but no universal thresholds across indigenous vegetation types or species.

Minimum abundance or dominance occurs for species to maintain viable populations at the site to catchment scale. For example, minimum dominance for regeneration success (e.g., dioecious tree species; pollination efficacy). Similarly, there should be density-dependent effects of plant species that provide habitat or resources to other taxa.

Tipping points in cover have occurred in the past, and these can be used to understand future thresholds. For example:

- Past transitions from indigenous forest to non-woody vegetation have occurred widely, dominance by indigenous plant species was maintained throughout; this suggests that knowledge of representativeness and past disturbance is needed to understand the appropriateness of which indigenous species dominance is suitable for quantifying EI.
- In addition, there is now often a different tipping point toward dominance by non-native woody plant species (pines, legumes, thyme) and their long-term dominance suggesting that increases in non-native plant species dominance are a useful and information indicator of EI (see also Sapsford et al 2020; 2022).
- Novel biogeochemical processes, particularly increasing the availability of macronutrients, have occurred. National-scale terrestrial eutrophication has occurred, linked closely to land use management (Parfitt et al. 2012), and high-nutrient availability feeds-back to maintain dominance of mostly non-native plant species (e.g, Dickie et al. 2022).
- Fire traps/ altered disturbance regimes have occurred whereby indigenous dominance has declined because of fire, has been replaced by more fire-adapted non-native

species (or few native species like manuka), creating higher risk of future fires. Climate change is thought to exacerbate such feedbacks.

The links between declines of indigenous plant species and human health are less well known, and more indirect effects appear common. For example, shifts towards dominance of some non-native plant species are a major source of allergens (e.g., from pollen of privet and silver birch, and many introduced grasses). Declines in taonga tree species are thought to pose a major issue for community wellbeing, especially of mana whenua (see tree canopy dieback attribute).

For many naturally uncommon or rare ecosystems dominated by indigenous plant species, some tipping points have been observed that are thought to be driven by climate, weeds, increased nutrients, or altered hydrology; these effects are largely considered tipping points by irreversibly disrupting processes maintaining these highly specialised ecosystems (Holdaway et al. 2017). Some require ongoing management.

B5. Are there lag times and legacy effects? What are the nature of these and how do they impact state and trend assessment? Furthermore, are there any naturally occurring processes, including long-term cycles, that may influence the state and trend assessments?

Much of this question is discussed above in A3. Species' dominance is dynamic, responding to past disturbances, environmental conditions, interactions with other species (i.e., competing plant species, herbivory, pathogens). Lag times of decades are typical, for example, post-fire recovery of vegetation or directional successional changes in vegetation (Wyse et al. 2018). Shorter-term lags of few years are possible for some measures of dominance, for example, cover declines driven by defoliation of canopy trees. Historical fires, ongoing land use, and biological invasions each have major legacies that determine the distribution and abundance of indigenous plant species nationally, although these legacies vary regionally from near complete elimination of dominant species (e.g., lowland forests in Canterbury) to relatively minor legacies (e.g., indigenous forests on some offshore islands).

Both lags and legacies strongly affect current state and trends in indigenous plant dominance. Some of these effects are part of natural disturbance dynamics and successional processes (e.g., Allen et al. 1999, Wyse et al. 2018). However, there are several long-term progressive changes including climate and biological invasions. Increasing fire risk is region-specific, and creates feedbacks to more fire-adapted species; most of which are non-native plant species (Perry et al. 2014). Environmental weeds are increasing in number and distribution over time (Brandt et al. 2021), and these invasions can drive declines in indigenous plant species (Sapsford et al. 2020).

B6. What tikanga Māori and mātauranga Māori could inform bands or allocation options? How? For example, by contributing to defining minimally disturbed conditions, or unacceptable degradation.

Although we cannot comment directly on mātauranga Māori, we do provide suggestions from our experience that for indigenous plant dominance, there are condition or states described from a Māori perspective. A few points:

- Place-based goals or aspirations for indigenous plant species exist, but these are specific to that site or hapū. For example, recovery of native tree species canopy in the Raukūmara Range following management of hyperabundant deer and goats.

- Species-specific goals are possible; for example, protecting taonga forest species from tree diseases such as Kauri dieback and more recently, myrtle rust. These examples show that protection of individual species (kauri) is crucial for safeguarding wider values across Iwi for enacting kaitiakitangia, and mauri of the ngahere (Lambert et al. 2018). For myrtle rust, an entire family of plants (the myrtaceae) are at risk and this includes structurally dominant tree species; in this case an observational decline in dominance attributed to the disease is an implied threshold.
- There are a few initiatives where biocultural monitoring of diversity, including dominance of indigenous plant species and shifting baselines of species abundance or condition, has been implemented by mana whenua (e.g., Lyver et al. 2019, 2021).

To summarise, place-based goals or acceptable changes in species dominance require community-specific approaches and cannot be directly applied to other sites. The goals or prioritisation of these activities are not driven by national standards or reporting, and as a consequence, scaling up site-based or biocultural approaches to regional or national scales is difficult.

Part C—Management levers and context

C1. What is the relationship between the state of the environment and stresses on that state? Can this relationship be quantified?

Dominance of individual species or indigenous plant species within a community or region is strongly related to the state of environment, stresses or management.

Direct responses are easiest to demonstrate, and this can be through direct harvesting or management of species themselves, or land use change altering local population abundance or the species spatial distribution (occupancy; e.g., Weeks et al. 2013, Cieraad et al. 2015). Anthropogenic disturbances such as fire or grazing also directly affect indigenous plant dominance (negatively for most species; Perry et al. 2014). Restoration or enrichment planting (e.g., of riparian margins) can directly increase the dominance of some indigenous plant species albeit at relatively small spatial scales (Forbes et al. 2023).

Indirect responses are commonplace, but far more difficult to demonstrate. These include altered hydrology, nutrient addition/eutrophication, and the impacts of invasive pests, weeds or diseases. Management interventions are commonly deployed for these stresses or drivers, but usually address one problem in isolation from other drivers. Regardless, a common goal of most management is maintenance or improvement in the abundance of indigenous (plant) species, although the relationship between driver, management and response is usually assumed rather than based on robust evidence (Allen et al. 2023).

C2. Are there interventions/mechanisms being used to affect this attribute? What evidence is there to show that they are/are not being implemented and being effective

Multiple interventions or interventions are deployed at all scales from site/property through to national that are too numerous to cover here. Most interventions seek to maintain or increase indigenous plant dominance, either explicitly, or at times, implicitly. Illustrative examples include:

C2-(i). Local government driven

- Regional biodiversity and biosecurity plans prioritise pest animal, weed or disease management for indigenous biodiversity benefit (i.e., to maintain dominance at sites or spatial extent; see also Bellingham et al. 2016, 2021, Hansen et al. 2021).
- Consenting processes (e.g., urban development) include explicit guidance on managing impacts on indigenous (plant) species, or recommendations for their inclusion in revegetation.
- Land use planning: maintenance of land cover classes containing indigenous plant species.
- The maintenance and protection of heritage trees includes indigenous species.

C2-(ii). Central government driven

- Central government: indigenous (plant) dominance is a key indicator for biodiversity and environmental monitoring (e.g., for Te Mana o te Taiao).
- Incentives for afforestation (billion trees initiative) seek to increase the abundance of few indigenous tree species.
- Carbon management. Regulation (ETS) and management of permanent or carbon forests includes retention, or an expectation, that indigenous woody plants will increase in dominance over the long-term (Forbes et al. 2023).
- Management of PCL using NHMS: indigenous plant dominance underpins measures of ecosystem condition and representativeness.
- Species recovery plans have been developed for some vulnerable indigenous plant species (DOC).
- Site-based management (DOC, MPI, LINZ): priority is given for high biodiversity value sites.
- Land cover that is ‘predominantly’ indigenous vegetation underpins the Threatened Environments Classification (TEC) used by both regional and central government to support identification of high-value habitats or vegetation, and to prioritise protection or management (e.g., see Cieraad 2015).
- Predominantly indigenous vegetation is also considered in the NPS for Plantation Forestry (2017) providing guidance for protection as part of commercial forestry operations.
- National Policy Statement for Indigenous Biodiversity (NPS-IB) sets out 10% indigenous vegetation cover for any urban or non-urban environment (with below 10% cover), providing a guideline for minimum level target of indigenous dominance.

C2-(iii). Iwi/hapū driven

Maintenance or renewal of taonga or key plant species is common and is a crucial component of biocultural restoration (Lyver et al. 2019). A few examples:

- For example, Wakatū has established an extensive regional database of indigenous and endemic vascular plants (Harcourt et al. 2022; Foster 2021 https://issuu.com/wakatu/docs/koekoea_issue3_ngahuru_2021).
- Improving the condition of indigenous forests and species in the Ureweras management included developing an intergenerational vision for restoration activities, and established a global precedent of establishing the forest as a legal personhood requiring protection under law (Lyver et al. 2017a, b; McAllister et al. 2019; Reihana et al. 2024).
- Kauri dieback surveillance, interventions, and rāhui on access to prevent future damage to the species by multiple hapū in Northland and the Coromandel (Waipara et al. 2013); see also Shortland T 2011 (<https://www.cbd.int/financial/micro/newzealand-monitoring-kauri.pdf>).
- Post-pine restoration using indigenous tree species to speed recovery (Forbes et al. 2021).
- Deliberate management to maintain dominance of mānuka for honey production (widespread management activity).

C2-(iv). NGO, community driven

There are too many community or NGO-driven initiatives that seek to increase indigenous plant dominance usually as part of land management or site restoration to describe here. Maintenance or increases in one or more indigenous plant species are common, and have been increasing over the past decade (Forbes et al. 2023). These activities include:

- Removal of grazing animals (retirement of previously pastoral land) for covenants (e.g., through QEII Trust).
- Hundreds of local/community-driven initiatives for restoring indigenous vegetation (e.g., Peters et al. 2015).
- Some NGO's focussed on increasing the number and availability of indigenous plant species (e.g., Tane's Tree Trust).
- Targeted campaigns by Forest and Bird, including advocating for additional pest animal management for carbon gains and improved ecological condition of indigenous forests (Hackwell and Robinson 2021).

C2-(v). Internationally driven

There are numerous international agreements and obligation for biodiversity, and most include goals for protection of condition and extent of indigenous-dominated vegetation or habitats. We provide only a few examples here:

- Indigenous species dominance is considered an EBV, and thus essential for reporting and meeting international agreements such as the Convention on Biological Diversity and IPBES (see Bellingham et al. 2020).

- Emerging biodiversity credit markets rely heavily on validation of investment using indigenous vegetation cover or management as evidence for biodiversity protection and improvement.
- ‘Red-listing’ of naturally rare ecosystems have been done qualitatively to meet IUCN criteria of threats to these high biodiversity value ecosystems (Holdaway et al. 2017), and more quantitative approaches across additional indigenous-dominated ecosystems are currently being developed (in collaboration with DOC and MfE).
- The Kunming-Montreal Global Framework for Biodiversity Target 3: “Ensure that at least 30 per cent globally of land areas and of sea areas, especially areas of particular importance for biodiversity and its contributions to people, are conserved...”
- The EU biodiversity strategy has a 2030 goal of legally protecting at least 30% of EU’s land, and strictly protecting 10% of land. They have also proposed a new law “Nature restoration law” which calls for binding targets to restore degraded ecosystems. This sets an international standard for protection.

Part D—Impact analysis

D1. What would be the environmental/human health impacts of not managing this attribute?

Not managing indigenous vegetation cover or maintaining dominance of indigenous plant species would lead to continued biodiversity loss and reduction in the extent and representativeness of indigenous species and ecosystems (Ewers et al. 2006, Cieraad et al. 2015, McGlone et al. 2020). This impact would be particularly problematic in already highly reduced environments such as lowland indigenous forests and wetlands.

We would also expect declines in many ecological processes, additional future problems (e.g., accelerated loss of species through extinction debts), and greater risks of crossing thresholds or tipping points. For example, knock-on declines in other species dependent on indigenous plant species (e.g., specialist insects and birds), or greater risks from pests, weeds and diseases for more fragmented or isolated populations or patches of indigenous vegetation (see Landscape Connectivity and Canopy Tree Dieback extent templates).

Human health impacts are largely unknown, but likely to be trivial in most cases (with the exception of declines in foundational or taonga species).

D2. Where and on who would the economic impacts likely be felt? (e.g., Horticulture in Hawke’s Bay, Electricity generation, Housing availability and supply in Auckland)

Few direct economic impacts are likely. Forestry land uses rely almost exclusively on non-native tree species. However, permanent or carbon forestry owners may enjoy an as yet unknown financial benefit or premium for increasing indigenous woody plant species in marginal sites through carbon or emerging biodiversity credit markets. Pastoral system also rely primarily on non-native herbaceous species, even though some systems also contain indigenous plant species; there are only

a few cases where benefits of increasing native plant species could be beneficial (e.g., better animal welfare in pastures containing native shrubs like matagouri, *Discaria tomentosa*).

D3. How will this attribute be affected by climate change? What will that require in terms of management response to mitigate this?

Climate change will affect indigenous plant dominance both directly and indirectly, but most of these effects cannot be managed. Direct effects relate to species' environmental tolerance and distribution, such that dominance could shift throughout the current range, and some species could adjust their range (e.g., upslope adjustment of subalpine species).

Multiple indirect effects of climate are likely, highlighting the problem of the 'twin crises' of climate change and biodiversity loss. Fire regimes are poised to worsen with increases in dry cycles and both natural and anthropogenic ignition sources; most indigenous species are not fire adapted, and thus their dominance is poised to decline with increased fire frequency or intensity. Rapid fire management. Storm damage can exacerbate the impacts of pests, weeds and diseases, including from other native species (e.g., as shown by outbreaks of native beetles causing canopy collapse of tree species following canopy disturbance, Wardle and Allen 1995). The number and distribution of environmental weed species is thought to be exacerbated by warmer conditions and disturbance, creating opportunities for non-native plants to suppress or replace indigenous plant species during successions (e.g., in coastal zones, following glacial retreat, post-fire).

Although the effects of climate change itself cannot be managed, landscape-scale approaches to planning and development can be used. For example, early work shows that land use and species selection can be used to promote more 'climate smart' landscapes in the future that could support dominance of indigenous plant species. Climatic refugia can be considered even for common taxa and included in biodiversity planning. Post-disturbance biosecurity efforts could be timed to reduce the impacts of invasive mammals or weeds. Longer-term, and for some taxa, new technologies could also be deployed to maintain dominance (e.g., RNAi inoculation of rata for myrtle rust). Finally, broader scale interest in afforestation using indigenous tree species could be developed (e.g., by scaling up direct seeding of native species to overcome dispersal or reproductive barriers to range adjustment under climate change; Douglas et al. 2007).

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