

6.7 Canopy die back extent

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Preamble: Forest canopy dieback extent is a well-established attribute related to the ecological integrity of forest ecosystems. Canopy dieback directly affects the physical structure of forest vegetation, and alters major ecosystem functions like primary productivity and nutrient cycling. Historical high-profile areas of tree dieback in New Zealand have driven efforts to quantify changes in tree canopy condition resulting from both invasive animals (e.g., possum browse and responses to management) and other observed canopy collapse of forest stands (e.g., native beetle outbreaks in *Nothofagus* forests). More recently, both novel pathogens (e.g., myrtle rust) and climate-related increases in disturbance (e.g., cyclone damage to canopies; drought-induced stress or tree mortality) have increased interest in understanding changes in forest canopy condition, the maintenance or recovery of key tree species, and the long-term impacts on forest ecosystems. Quantification of canopy condition have well-established protocols and data collection methods, and emerging technologies in remote sensing (high density LiDAR and hyperspectral imagery) have the potential to better characterise changes through time and at greater spatial scales. Given the multiple potential drivers of forest canopy condition or collapse, this attribute requires additional information to interpret the causes of canopy collapse, and distinguishing between natural canopy declines (i.e., caused by succession) from new and sometimes manageable threats (i.e., emerging diseases, invasive herbivore damage). Most current data collected follow from discrete dieback observations of one or more species at a site; but whether these issues occur at broader spatial scales, or cause longer-term declines in forest condition and ecosystem integrity require more systematic approaches to measurement, and understanding of whether regeneration is offsetting canopy dieback and tree mortality.

State of knowledge of the “Forest Canopy Dieback Extent” attribute: **Good / established but incomplete** – general agreement, but limited data/studies

Part A—Attribute and method

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

Forest canopy dieback extent is a direct measure of forest structure or state occurring across one or more canopy tree species. As a consequence, this attribute should relate to all components of ecological integrity (EI) including representation and composition (where dieback is related to increased species-specific mortality; discussed below), indigenous dominance (structure, by altering canopy structure) and functions (through alteration of ecological processes such as primary productivity or reproduction, often resulting from altered community composition). Moreover, given the importance of canopy tree species for driving forest ecosystem processes, this attribute will also indirectly be related to human health through alteration of some ecosystem services, and potentially well-being through values of forest health or condition, or direct risk from canopy collapse (although these possibilities have rarely been investigated).

In terms of human health, a reason this attribute commands attention is because people dislike seeing large areas of dead trees. Most people with only a passing knowledge of natural forests often don't appreciate that large-scale dieback of trees often reflects their large-scale recruitment at some stage in the past and them all reaching, simultaneously, a physiological stage where their canopies cannot be maintained. Moreover, most people do not grasp that decay rates differ among dead tree species such that some remain visibly dead in landscapes for at least 50 years (Peltzer et al. 2003; Mason et al. 2013; Fig. 1). Therefore, a key need is to draw distinctions (where it is possible) between natural processes of canopy dieback and those that are induced by either modern pressures (e.g., climate change) or novel pests and pathogens, some of which can be mitigated (see also Wyse et al. 2021).

The spatial scale and magnitude of canopy dieback extent, and over what time period this occurs, matters. For this exercise, canopy dieback is described as population-level (i.e., canopy dieback of individual trees is out of scope; Mueller-Dombois 1985). We are uncertain when a treatment of species-specific population-level dieback in a mixed species forest where most species remain alive constitutes canopy dieback (i.e., declines of canopy cover in one species are often compensated for by increases in canopies of other species). For example, does death of most kohekohe attributable to browsing possums, or death of taraire or mamaku attributable to drought but where most other trees remain alive constitute canopy dieback? We think not, but this raises the issue that most canopy dieback measurements consider individual species, whereas EI should be more closely linked to overall forest canopy condition (see Tierney et al. 2009; see more detail comments below on methods). We suggest that canopy dieback is best measured at the forest stand-level or catchment scale, and represents death of structurally dominant tree species, and could be reported at the regional scale for widespread forest types.



Figure 1. Long-persistent, slow-decaying southern rātā c. 40 years after its canopy dieback in the Kokatahi River valley, central Westland (Photo, Peter Bellingham, 1997; Mason et al. 2013).

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

The spatial extent and magnitude of canopy dieback determines impact on EI (e.g., Tierney et al. 2009).

Maintenance of dominance or shifts in dominance. We know of no examples where indigenous tree species dominance has been lost as a result of canopy dieback. Compositional shifts can occur whereby declines in the canopy cover or extent of one species are replaced by increases in one or more other tree species (Fig. 2), but not always (i.e., canopy dieback in beech canopies usually results in replacement by the same species; Wardle and Allen 1985, Ogden 1985, 1988). We are not aware of any stand-level or catchment-scale tree canopy dieback that results in non-native plant invasions, but this requires further investigation. Furthermore, major canopy dieback or disturbance events

such as cyclones coupled with increased numbers of environmental weed species could increase invasions into indigenous forests in the future. In terms of bird communities, canopy dieback can provide new habitat for birds that rely on wood decay (i.e., insect prey) and can provide nesting sites for hole nesters.

Representativeness. There is no evidence of loss of representativeness of forest composition caused by stand or catchment-scale canopy dieback, but this has not been well investigated. Novel pathogens could change this through selective damage to few tree species (e.g., as has occurred for Dutch elm disease or chestnut blight elsewhere).

Function. Changes in function are likely, but have not been investigated. Because canopy condition regulates major energy fluxes and nutrient cycling in forest ecosystems, major canopy dieback or population declines should affect multiple ecosystem functions and ecological processes. Declines or loss of ‘foundation species’ is of particular concern (Ellison et al. 2005, Genung et al. 2020).

Evidence of impact on human health. We are not aware of direct linkages between tree canopy dieback extent and human health. Visual assessment of forest or bush ‘health’ is predicated on a healthy forest being one without dieback, and this represents visual landscape value to people (e.g., Handford et al. 2021). Perceptions of the public to environmental condition have also been evaluated used repeated national surveys, but canopy dieback is not considered separately as a part of forest health or condition (see <https://www.landcareresearch.co.nz/discover-our-research/environment/sustainable-society-and-policy/environmental-perceptions-survey/>).



Figure 2. Reduced forest structure after canopy dieback, Pohangina Valley, Ruahine Range. Podocarps (mostly miro, *Pectinopitys ferruginea*, and rimu, *Dacrydium cupressinum*) that were formerly in or emergent above a canopy dominated by kāmahi (*Pterophylla racemosa*) show the former height of the canopy that showed widespread canopy dieback in the 1950s (Rogers and Leathwick, 1997), while the wholly native community present now (mostly māhoe, *Meliclytus ramiflorus*, and kātote, *Cyathea smithii*) is much shorter (Photo, Peter Bellingham, March 1996; Bellingham et al. 1999). Rogers and Leathwick (1997) attributed this canopy dieback to browsing by possums and their non-replacement in the new canopy to browsing by goats and red deer.

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)?

There is little evidence of change over the past few decades generally. Some exceptions include:

- Kauri (*Agathis australis*), where increased canopy death is frequently attributed to a phytophthora pathogen (Kauri Dieback - *Phytophthora agathidicida*). However, the trajectory of change is complicated by previous concerns that canopy death was often attributed to illegal bleeding of gum in the 1920s or earlier (Orwin 2019), and this may interact with the current pathogen. Regardless of the original cause, all size and age classes of kauri are susceptible to infection and death (Bellgard et al. 2016, Bradshaw et al. 2020).



Figure 3. Kauri dieback is a high profile disease causing tree canopy tree dieback (<https://www.kauriprotection.co.nz/>).

- A plant bacterium, *Candidatus Phytoplasma australiense*, spread by insects has selectively damaged and caused dieback ('sudden decline') of mature cabbage trees, but most young trees are left untouched (Beever et al. 2006). The magnitude of dieback has declined over the past 25 years, and varies regionally (Brockie 2020).
- The recent incursion by Myrtle rust on an entire family of plants (Myrtaceae) over several regions has the potential to create major dieback of indigenous tree species including structural dominant species (Teulon et al. 2015, McCarthy et al. 2021).

- Climate change is likely to interact with other drivers such as herbivores or resident pathogens to increase the frequency, extent or magnitude of canopy dieback extent (e.g., McCarthy et al. 2021). Novel pathogens (e.g., rapid 'ōhi'a dieback) are highly likely to further compound causes of canopy dieback at the decadal timescale.

These effects are reversible only if canopy dieback does not result in population-level declines beyond background variation. If canopy dieback increases tree mortality but also increases growth or reproduction, the net demographic effects can be neutral. This highlights that canopy dieback is one (visible) aspect of more complex forest dynamics, and additional information on tree regeneration and threats or management are needed to understand if the impacts of canopy dieback are reversible. As an example, seasonal or temporary loss of tree foliage and canopy condition is common, but this is reversible over one or few years if individual trees survive. In contrast, canopy dieback resulting in increased tree mortality, coupled with regeneration failure, would be the worst-case scenario and unlikely to be reversed over several decades given the longevity of most indigenous tree species (see also McGlone et al. 2016, 2017).

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?

Nearly all canopy dieback assessment, whether stand-based or catchment-based, focusses only on one demographic component of populations (i.e., tree mortality). Without commensurate assessment of population recruitment, the effects on population maintenance, forest ecosystem condition, and thus EI are unknown. Most scales of measurement or monitoring are relevant: site, catchment, regional, and national. Stand-level monitoring is done using different approaches including Foliar Browse Index (FBI) (canopy cover), standard plot-based vegetation measurements, and remote sensing (including LiDAR). Systematic assessment is rarely done among regions, but has been established with sites or catchments for some long-term plot-based studies (e.g., tier two networks, DOC) that occur in areas of canopy dieback and provide population-level assessments of consequences (e.g., Bellingham et al. 1999a; Richardson et al. 2024). Some of these assessments have shown that the mortality rates of some canopy trees are exceeded by recruitment rates of new stems, e.g., of kāmahi in four Westland valleys in which it has shown widespread canopy dieback (Bellingham and Lee 2006; see also Allen and Rose 1985, Rose et al. 1992). Nearly all monitoring is done at the site or catchment-scale but could be scaled up to report at regional or national scales. The main barrier to scaling up monitoring is variation in monitoring efforts among regions, rather than available methods or standards. Canopy dieback is often not the focus of monitoring aside from site- and species-specific areas of concern but can use information collected for biodiversity assessment or C accounting purposes.

Given the complementary ground- and remote-sensing based approaches to monitoring, there are opportunities for monitoring and reporting that link these information sources (e.g., Meiforth et al. 2020). Major efforts internationally to better link remote sensing or earth observation data with plot or ground-based monitoring have generated several emerging indicators of EI related to canopy condition or dieback such as the Forest Structure Condition Index and Lost Forest Configuration (see summary of Hansen et al. 2021). In many cases the underpinning data needed to develop these attributes is available in NZ, but integrating these data require additional research effort and implementation. Similarly, linking plot-based and remote sensing methods is an ideal for EBV's,

suggesting there are efficiencies to be made with biodiversity monitoring and reporting (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 (i.e., as highlighted in multiple investigations by the Parliamentary Commissioner for the Environment).
- 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., diversity, pellet counts, deadwood assessment) carried out at the same time. Individual plots

range in cost from \$2000-5000 depending on access and complexity. 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.

Efficiencies can be made with data collection and analyses for other needs. For example, remeasurement of several networks of vegetation plots on Rakiura have been prioritised by DOC (maximizing forest C programme) because interest in potential C loss from forests dominated by palatable tree species such as kāmahī, and for potential responses to planned predator control operations and eradication across the island. Additional networks are now being considered for remeasurement, and much of the information gathered for this activity (Richardson et al. 2024). There are also opportunities for efficient re-use of data collected for other monitoring purposes to quantify changes in canopy condition; here, the cost of data collection and hosting is already covered.

Detailed costs for other monitoring approaches such as remote sensing data are available. 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 (Cieraad et al. 2015).

Direct costs of monitoring vegetation plots or canopy condition are well documented, however, additional interpretive data to understand attribution or manage the causes of canopy dieback is not included in these estimates. Single drivers of dieback are rare, requiring this additional effort.

Human perception of dieback is a key driver of monitoring or interventions (e.g., cabbage tree dieback), so there is a largely unknown opportunity and cost of including communities or citizen science to monitor canopy dieback (see also <https://www.landcareresearch.co.nz/discover-our-research/environment/sustainable-society-and-policy/environmental-perceptions-survey/>).

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

There are few examples of canopy dieback being monitored by Iwi/Māori. See Raukūmara Pae Maunga, one stated objective is to monitor canopy recovery across habitats (see <https://www.raukumara.org.nz/objectives>).

Relatively high-profile research programmes such as Ngā Rākau Taketake (<https://bioheritage.nz/about-us/nga-rakau-taketake/>) explicitly supported community-based responses to these pathogens (Lambert et al. 2018). Kauri dieback has been monitored by several communities in Northland and the Coromandel, driven in part by Kauri being considered a cultural keystone/taonga species. The recent invasion by myrtle rust (*Austropuccinia psidii*) threatens a range of taonga tree species (Teulon et al. 2015). These diseases have prompted recent work on Mātauranga Māori approaches to surveillance including researchers directly working with kaitiaki and rangatira from ten tribal regions, whose taonga are affected by myrtle rust and kauri dieback to monitor the presence and impacts of these diseases (see <https://bioheritage.nz/research/integrated-surveillance/>). See also <https://bioheritage.nz/new-forest-health-tool-helps-mana-whenua-capture-culturally-important-data/>.

More generally, cultural frameworks to monitoring, including biocultural approaches (Lyver et al. 2019), do not focus on a specific attribute or metric, but could include specific indicators as part of more integrated assessment of forest condition, or ngahere ora/mauri state (Waipara et al. 2013; Reihana et al. 2024).

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

Tree canopy dieback extent is directly linked to indigenous plant dominance for indigenous forests. Canopies are formed by dominant tree species, so declines in canopy extent directly reflect change in biomass or dominance of canopy tree species. This relationship works at a stand or catchment scale, but at regional or national scales, it is the spatial extent of canopy dieback affecting species distribution and abundance that is related to dominance (i.e., species occupancy and range). Lowland forest extent should related to canopy dieback through several mechanisms but this requires investigation. These include forest fragmentation effects, increased edge effects, and closer proximity to novel weeds, pests and diseases, all of which can increase canopy disturbance (and suppress tree species recruitment in many cases). As a specific example, lowland forest fragments are often disturbed, increasing the invasion of some understory weeds (e.g., *Tradescantia fluminensis*) that, in turn, suppress tree recruitment, leading to canopy and population declines over the long-term (Standish et al. 2001).

Contemporary drivers of declines in tree canopy condition include invasion herbivores (primarily possums) and increased disturbance (including fire, pathogens, and storms). Disturbance itself is not an attribute, but better information on disturbance is needed to interpret changes in canopy condition or dieback. In contrast, attributes that consider the distribution, abundance and impacts of pests (foliar herbivores) and diseases should be directly, positively related to canopy dieback. These relationships will be species-specific because both herbivores and diseases damage species or functional groups of species differently (Nugent et al. 2000).

Part B—Current state and allocation options

B1. What is the current state of the attribute?

There is no consistent regional or national-scale reporting of tree canopy dieback extent. Most reporting is for a focal species of concern at one or few sites. No long-term commitment to remeasurement is currently in place. However, other measurement or monitoring schemes can be used to derive canopy condition or potential dieback from remeasurements, including the national vegetation plot network (DOC tier one; LUCAS) and several regional networks (Richardson et al. 2024).

Current measurement activities together with management could be used to develop this attribute as an indicator of ecological integrity. For example, comparing plot-based or remote-sensed measures of canopy condition between management units for herbivore control, predator elimination, or presence of pathogens could be used to establish baselines of dieback, and responsiveness of this to management interventions (including failure to manage; see also Peltzer et al. 2024).

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

Baseline or reference states for canopy condition, dieback and long-term changes in tree populations are available from several sources. These include:

- **Paleo-ecological information.** Long-term pollen records of stand to catchment-scale dieback. Multiple species and dieback events have been observed. This demonstrates that some tree species have gone through canopy dieback in the past (McGlone et al. 2016, 2017).
- **Plot-based measurements.** National to regional-scale networks of temporary and permanent vegetation plots focus largely on indigenous forests, and usually provide cover estimates of species. These provide excellent quantitative estimates of variation in tree canopy cover along major environmental gradients. This information can be used in combination with other information for management or disturbance history effects on different tree species. For example, evaluating the spatial impacts of an earthquake on canopy condition and mortality of mountain beech forest (e.g., Allen et al. 1999).
- **Observations.** Most canopy dieback is a visible feature of forest condition. Semi-quantitative or qualitative measures can be used to compare changes in ‘healthy’ vs. ‘dieback’ forest stands using repeated images, perceptions of forest health, or observations (e.g., Jamieson et al. 2014).
- **Remote sensing** methods including aerial photography, high-density LiDAR and hyperspectral imagery provide data for evaluating canopy condition and changes through time at larger scales. These approaches can be used to detect overall changes in canopy cover, leaf area or ‘greenness’ (NDVI), but usually do not distinguish responses of different tree species. However, these approaches have been used to monitor changes in some emergent canopy species such as kauri, to quantify extent of canopy dieback.

These approaches can be used to understand canopy condition or dieback across different scales, time periods and resolutions. Some specific measurements include quantification of structure (vegetation cover by species, live cover fraction, foliar browse index (FBI), Foliar cover index (FCI), leaf area index (LAI)) and function (NDVI, Aboveground Net Primary Production (ANPP)). Only plot-based measurements can be used to understand whether tree canopy dieback is a symptom of population declines and longer-term changes in EI.

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 no current bands or limits for canopy dieback extent. However, thresholds or limits could be developed from existing data, and should be guided by population processes (i.e., to related canopy condition to increased tree mortality rate for different species, and to evaluate whether mortality greatly exceeds recruitment rates; see also Table 1). Given the high variability among key tree species in life history strategies such as response to disturbance, longevity, and susceptibility to

herbivores, pathogens or other drivers of canopy condition (Allen et al. 2003, 2013, Richardson et al 2009), bands or limits will vary among species and potentially regions. The LUCAS plot network data could be used to understand baselines and changes at national to regional scales.

Table 1. Summary of canopy dieback. Although no quantitative bands are currently available, current data could be used to establish an ordinal scale of dieback extent for different species, forest classes, or regions. Both the canopy lost and duration of declines should be used in combination to establish bands or thresholds for describing canopy dieback extent. Spatial scale is not explicitly included, but these criteria could be applied at any spatial scale of interest.

Canopy dieback extent	Proportion canopy lost	Duration of canopy loss	Impacts on forest ecosystem integrity	Notes on potential causes of dieback
Low	<10%	<1yr	Negligible	Seasonal variability or minor disturbances. Could be an early warning of pathogens requiring surveillance.
Moderate	10-30%	1-2 yrs	Variable	Requires ancillary information to understand loss. Can result from weather events, environmental stresses. Warrants evaluation.
High	>30%	3+ yrs	Variable-high	Increased tree mortality and population declines likely. Prioritise for monitoring or intervention.

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

No. However, data are available to assess the links between canopy dieback extent and tree population declines, and this would reveal thresholds and tipping points for forest structure, function and EI. Similarly, the links to human health have, to our knowledge, not yet been investigated explicitly, but could be. For example, the relationship between visible tree death and peoples’ wellbeing could be evaluated for canopy dieback for different species or locations. The narratives for kauri dieback recently imply community concerns about forest health and the wellbeing of communities (Lambert et al. 2018).

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?

There are multiple causes of tree canopy dieback extent, and these include several lags and legacy effects that cannot be covered in detail here (but see Mueller-Dombois 1985). The major considerations include:

- Recognising and distinguishing seasonal or short-term tree canopy declines that do not cause increase tree mortality or overall population declines. This can include seasonal changes in canopy condition, short-term damage from wind, snow or browsing, and resident pathogens.
- The longevity of visible death varies widely among species, from a few years to many decades (Peltzer et al. 2003, Mason et al. 2010).
- Past, infrequent disturbance is an important legacy; many of our tree species recruit as a cohort following disturbance, but then also die as a cohort. For example, podocarps following volcanic eruptions (Richardson et al. 2009; Smale et al. 2015). Thus, infrequent historical and contemporary disturbances are a major driver of tree population structure and canopy condition.
- Tree population processes generate multiple lags including in recruitment, growth and mortality, often spanning several decades.

Despite these and other lags and legacies involved in forest dynamics (and thus canopy condition), relatively rapid changes in canopy condition or dieback occur that can be attributable to specific causes (like novel pathogens, increased or chronic herbivory by invasive animals). As a consequence, state and trend analyses will be relatively easy to apply at stand or catchment scales, but more difficult to disentangle from background processes at regional or national scales (with some exceptions, like the broad impacts of new diseases, Jo et al. 2023).



Figure 4. Major lags and legacies drive changes in canopy dieback extent including major natural disturbances like earthquakes. Ongoing mortality of trees and canopies can occur for several years following disturbance (from Allen *et al.* 1999 *Ecology* 80, 708–714).

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.

Both tikanga Māori (e.g., for biocultural monitoring) and mātauranga Māori (e.g., for understanding changes in forest condition) are emerging as crucial approaches for evaluating the condition of the environment and people, interdependencies, and acceptable conditions or goals of management (Lyver *et al.* 2019). Forest canopy dieback extent is a specific concern, and has driven several activities in recent years including:

- Mana whenua led or inclusion in surveillance and management of kauri dieback and myrtle rust.
- Co-ordination or inclusion of Iwi in monitoring or strategic long-term management of long-term forest declines (e.g., in the Raukūmara and Kaweka Ranges).
- Greater inclusion of Māori in major conservation efforts such as predator free (collaborative groups) and Te Mana o te Taiao can be linked, at least in part, to changes in forest health and tree canopy condition. Although specific bands or allocation options have not yet been developed, these efforts provide the system-level approach needed for mana whenua to identify targets, limits and thresholds.
- More generally, the scale of canopy dieback processes is usually at the stand- or catchment-scale, and this matches the scale of concern by hapū and Iwi for aspirations (e.g., Tūhoe goals, management and monitoring of Te Urewera; Lyver *et al.* 2017, 2018).

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?

Distinguishing among the multiple, and sometimes interacting drivers of tree canopy declines is complex (see B5). Tree canopy condition or dieback can be driven by cultural disturbance (harvesting), biological invaders (possum browsing, pathogens, disease), climate change (increased wind or storm damage, increased fire risk), and land use change (including forest fragmentation). Although there is a wealth of information and knowledge for forest dynamics and responses to many of these pressures, this information has not been used to relate state of the environment to canopy dieback, but rather consider individual stresses individually, and in some cases, the effectiveness of management interventions. For example, canopy defoliation by invasive possums has been widely quantified, and this information used to relate defoliation intensity or duration to tree mortality for some tree species (Urlich and Brady 2005, Gormley *et al.* 2012, Sweetapple *et al.* 2016); but whether stresses interact or compound changes in canopy dieback (e.g., defoliated trees are more susceptible to pathogens) is largely unknown.

All of the stresses mentioned could increase canopy dieback extent, and some are manageable. Relating invasive mammal impacts to forest condition, or to forest recovery following their management is a long-term issue (Husheer and Tanentzap 2023). Recent interest in linking pest animal management to C sequestration provides ample information that could link management to tree canopy dieback because the data collected to assess ecosystem C usually includes measures of canopy cover for different species at regional to national scales (see Holdaway et al. 2012, Peltzer et al. 2024). Cultural disturbance (harvesting) is trivial because forestry involving indigenous tree species was largely stopped in the mid 1980's (McGlone et al. 2022). Land use and management could contribute both negatively (e.g., through fragmentation of vegetation) or positively (e.g., restoration, enrichment planting), but the effects of this on tree canopy condition or dieback have not been assessed.

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?

Visible death has driven management/interventions, but often to no quantifiable effect. This is, in part, due to a lack of well-designed monitoring of canopy condition, and additional information needed to evaluate the efficacy of management interventions.

The most common intervention to date has been possum control following observation of dieback for few canopy tree species. Often herbivore control is applied at local (stand or site) scales, and can be effective for increasing canopy health and reducing tree mortality (e.g., Nugent et al. 2000 documented recovery of kohekohe). Often both herbivory and control are repeated, but few studies have considered how long herbivory or management is needed to improve canopy condition (but see Payton et al. 1997, Holland et al. 2013, Sweetapple et al. 2016).

Possum management can improve canopy condition and foliage cover, ultimately improving the survival of affected tree species. The best data available were part of a 'how long' study at Waihaha forest (Sweetapple et al. 2016), in which the canopy condition of 4 tree species were assessed following possum management (or unmanaged experimental controls) over 20 years. that showed:

- Three possum-palatable tree species had increased foliage cover and reduced crown dieback over the 20 years of monitoring.
- Increases in foliage cover were modest (8–19%), but consistent with other studies (Nugent et al. 2010; Duncan et al. 2011; Gormley et al. 2012).
- Canopy recovery of heavily browsed tree species took about a decade.
- These results reflect that less than half (20–49%) of trees were possum browsed at the start of the study.
- Trees that were initially heavily browsed by possums had large increases in foliage cover (36–89%) during the first 6 years of the study.

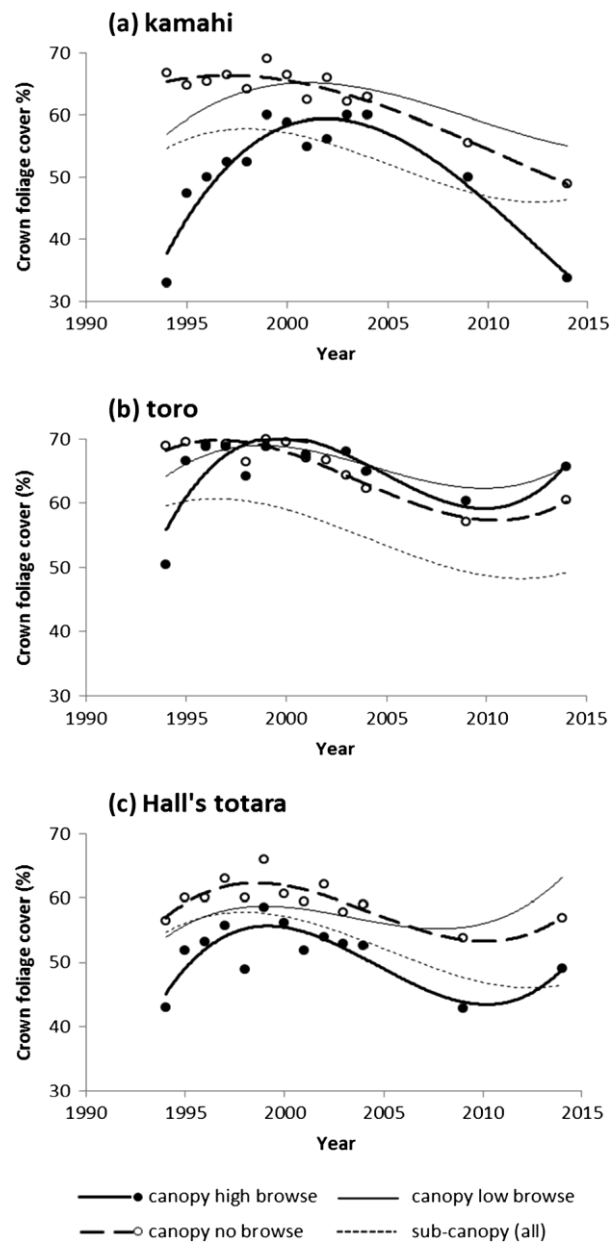


Figure 5. Interannual trends in mean foliage cover for three tree species at Waihaha. Data for each species are divided in 4 groups based on 1994 possum browse class (none, low, high). Data points are shown for canopy trees in the high browse (solid circles) and no browse (open circles) classes. From Sweetapple (et al 2016).

A few recent management options have been developed to manage kauri dieback, ranging from rāhui on access to prevent movement of the disease, phosphite injection to increase the health of infected trees (Horner et al. 2016), and Mātauranga Māori approaches using rongoā derived from indigenous plant species. All of these approaches could maintain or reduce canopy dieback of kauri individuals or sites, and appear effective over the short-term (<5 years).

How such management effects scale up to the catchment- or regional-levels has not been evaluated, but could be done by integrating management information (i.e., mapped areas of aerial possum control alongside measures of possum abundance) with repeated measures of canopy condition (see also Peltzer et al. 2024).

Early or major impacts of invasive herbivores have occurred, but this is largely restricted to observations rather than based on monitoring efforts. For example, some successional species are thought to have been locally extirpated by herbivores (i.e., fuschia, tutu).

C2-(i). Local government driven

- Auckland Council are actively monitoring integrity, and have been actively managing several current or potential tree diebacks in the region; these are included in state of the environment reporting (Griffiths et al. 2021).
- Kauri dieback. Auckland and Northland councils have active programmes for awareness, policies to limit movement of pathogens including rāhui, and have carried out direct operations (e.g., phosphite injection of trees) to protect this species. All of these interventions are relatively short-term solutions for managing the disease, and more strategic/long-term solutions are sought.
- Cyclone damage (e.g., Ita in Kahurangi, Cyclone Gabrielle in Hawke’s Bay) has required several councils to prioritise protection of indigenous forests, improve monitoring, or consider the impacts of these storms on land use.
- Cabbage tree decline (Beever et al. 1986, Brockie 2020) was a major national (primarily North Island) and regional concern that prompted many councils to carry out additional monitoring, awareness campaigns, and some management to contain spread of the disease.
- Often local dieback events or perceptions have been raised by communities or councils, and used to prioritise management of presumed drivers of decline. This has most often been possum control, and the effectiveness of such management for improvements in canopy condition or tree population improvement are rarely considered (e.g., kaikawaka/mountain cedar declines on Mt. Taranaki; beech dieback at Moa Stream canterbury, rātā-kamahi dieback in parts of Westland).

C2-(ii). Central government driven

Although most management is targeted at sites/stand-level dieback, at times concern for broad-scale dieback requires more co-ordinated approaches nationally such as:

- myrtle rust response, mostly aimed at understanding the vulnerability of key indigenous tree species to this pathogen.
- kauri dieback has required both DOC and Biosecurity NZ to manage this taonga species through awareness campaigns, restricting access to vulnerable sites, and monitoring tree canopy condition (see also <https://www.mpi.govt.nz/biosecurity/exotic-pests-and-diseases-in-new-zealand/long-term-biosecurity-management-programmes/protecting-kauri-from-disease/>).
- concern over potential broad-scale declines in key tree species such as kāmāhi using the national network of vegetation plots. Current plans and co-ordinated budgets to address this decline are in progress (e.g., via the ‘Maximising Forest C’ programme involving DOC, MPI and MfE).

C2-(iii). Iwi/hapū driven

Several examples of Iwi or hapū management include:

- Kauri dieback and access to sites or areas using rāhui.
- Aspirations and long-term plans for improving forest health, and linked cultural values (Tūhoe Tuawhenua Trust management of Te Urewera).
- Raukūmara Pae Maunga Trust seek to monitor indigenous forest canopy recovery across habitats within the Raukūmara Range, largely resulting from goat and deer control efforts (<https://www.raukumara.org.nz/>).
- Māori community responses to myrtle rust (Black et al. 2019).

This is not an exhaustive list, but indicates that community/hapū approaches to management of forests are widespread. In many cases, Iwi/hapū driven approaches utilise mātauranga Māori and conventional scientific understandings for planning and monitoring efforts.

C2-(iv). NGO, community driven

Several community-driven efforts to raise awareness and intervene or manage canopy dieback have occurred such as:

- Campaigns by Forest and Bird raising the issue of national-scale forest condition, and potential declines or dieback of tree species such as kāmāhi, invoking pest animals as the primary driver of these declines that require management.
- Public concern over rātā-kāmahi dieback: several campaigns or efforts over perceived dieback of these species several regions including Westland, Rakiura, Bay of Plenty (Pūtauaki).
- Several community-based activities around kauri dieback and myrtle rust ranging from awareness of these tree diseases, citizen science initiatives, and mana whenua-led management activities (e.g., Black et al 2019, Sutherland et al. 2020, Hill et al. 2021).
- Landowners and community trusts also commonly use photo points for documenting changes in forest canopy condition and change following changes in management like pest animal control, restoration or retirement from grazing (e.g., in some QEII Trust covenants).
- Many forest health monitoring schemes are in place, primarily as part of industry biosecurity activities for plantation forests (e.g., FOAs, NZFFA).
- See also Peters et al. (2016) discussion of community-based monitoring, which includes aspects of forest health.



Figure 6. View from the Kauri Museum at Matakoho, Northland, in 1995. By 2000, the single remaining tree at the site stood dead. From Brockie (2020).

C2-(v). Internationally driven

Tree canopy dieback itself is not required for international agreements or obligations. However, national reporting requirements for Convention on Biological Diversity, led by DOC, include tier one monitoring nationally that can be used to report on changes in indigenous plant species dominance and extent (see template for that attribute), and which are considered internationally to be an Essential Biodiversity Variable (EBV; Pereira et al. 2013; see also Bellingham et al. 2020).

Similarly, forthcoming efforts to apply the IUCN of ‘red listing’ of ecosystems is in progress (mid-2024) with DOC and MfE, which will include indicators of ecosystem state and change for international reporting obligations to the IUCN.

Part D—Impact analysis

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

Few to no negative changes are likely given the lack of current management, or it’s effective for few species at local scales only. However, high-profile tree species dieback of foundation species (structural and functional dominant species) can have major impacts on both environmental condition and human well-being. This is exemplified by concerns about the progression of tree diseases such as kauri dieback, whose declines could cause major negative impacts on forest ecosystem integrity as well as multiple communities. For Māori, declines of taonga species can have multiple impacts including undermining the mauri of the ngahere and multiple connections to mana whenua (Waipara et al. 2013, Black et al. 2019).

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 from tree canopy declines, and have not generally been considered for indigenous tree species. There is potential that landscape values would decline in

some areas for vulnerable species, potentially having impacts on tourism or property values. Growing interest in claiming carbon credits from management of indigenous forests could also create a direct economic impact (through inability to claim C credits because of dieback sufficient to); this could be national in scale for vulnerable forest types (i.e., as suggested for forests containing kāmahi in Hackwell and Robinson 2021). Consequences of canopy dieback on ranges that are critical for water supplies to urban areas or to horticulture (e.g., drought-induced dieback of high-elevation forests in the Kaimai Range; Jane and Green 1983a,b), such as water interception, stem flow, and hydrology in soil are unknown but potentially important.

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

Several climate-change-driven effects are likely:

- Sudden decline in cabbage tree is suggested to move south as climate changes (Brockie 2020), although the underpinning evidence or forecasts for this are lacking.
- Multiple new or emerging pathogens are likely with climate change and increased transportation/trade frequency (Sturrock et al. 2011).
- Major likely or known drivers of canopy declines are likely to increase in the coming decades, including environmental weeds, climate-induced drought (Jane and Green 1983a,b; Grant 1984), increased frequency and intensity of fires, and extreme weather events (Wyse et al. 2018).

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