Ecological Indicators 158 (2024) 111510

Contents lists available at ScienceDirect

Ecological Indicators

journal homepage: www.elsevier.com/locate/ecolind

# Vegetation quality assessment: A sampling-based loss-gain accounting framework for native, disturbed and reclaimed vegetation

Bradley L. Boyle [a,b,*], Warn Franklin [c], Alison Burton [c], Raymond E. Gullison [a,d]

[a] Hardner & Gullison Associates, LLC, 15 Woodland Drive, Amherst, NH 03031, USA
[b] Department of Ecology and Evolutionary Biology, University of Arizona, Biosciences West 310, Tucson, AZ 85721, USA
[c] Teck Coal Limited, P.O. Box 3000, Sparwood, BC V0B 2G1, Canada
[d] The Biodiversity Research Centre, University of British Columbia, 2212 Main Mall, Vancouver, BC V6T 1Z4, Canada

ARTICLE INFO

*Keywords:*
Impact assessment
Ecological monitoring
Vegetation restoration
Reclamation
Loss-gain accounting
Biodiversity offset

ABSTRACT

Governments and society increasingly are demanding that industrial projects result in a net positive impact (NPI) on biodiversity. Impacts are commonly measured in terms of losses and gains of area and quality of vegetation, where quality refers to how closely a site matches the condition of native vegetation in its undisturbed state. Existing vegetation quality frameworks share a number of limitations, including little or no replication, uncertain scope of inference, vulnerability to bias, and inability to measure error. Here we present the Vegetation Quality Assessment (VQA) framework, a sampling-based extension of Quality Hectares that measures vegetation quality in terms of overlap between the probability distributions of ecological indicators at a project site and in undisturbed (benchmark) vegetation of the same kind. Distribution overlap incorporates natural variation at the landscape scale and provides an intuitive measure of quality that varies between 0 and 1. Indicators are measured using a stratified-random sampling design that minimizes bias and supports inference at the scale of the project landscape. Confidence limits of quality and quality hectares are determined by bootstrapping; power and minimum sample sizes are estimated by Monte Carlo simulation. Multiple assessments track losses and gains of quality hectares and enable accurate accounting of progress to NPI. The VQA framework can be implemented using a variety of vegetation sampling methods, allowing existing vegetation databases to be leveraged as sources of data. We conclude by demonstrating the application of VQA at several mining operations in the Elk Valley of southeastern British Columbia, Canada.

## 1. Introduction

Increasing global awareness of the pervasive effects of human activity on the natural world (Ceballos et al., 2015; Pimm and Joppa, 2015) is leading to voluntary and mandated approaches to reduce the impacts on biodiversity from resource development and related economic activities (International Finance Corporation, 2012; Rainey et al., 2015; UN Environment Programme, 2020). The extent and condition or quality of native vegetation is the principal currency by which impacts to terrestrial ecosystems are measured (Gibbons and Freudenberger, 2006). In addition to their inherent value as major components of the world's biodiversity (Kier et al., 2005), plants and plant communities provide the habitats and ecosystems upon which other organisms depend, and deliver essential environmental services through their influence on soil fertility, hydrological cycles and the global carbon balance (Schlesinger and Andrews, 2000). In the context of environmental impact assessment, inaccurate estimates of vegetation extent and quality can lead to reduced restoration effectiveness at impacted sites, mismatched biodiversity offsets, and potentially costly compliance failures (Maron et al., 2012). An accurate, objective and cost-effective framework for measuring vegetation quality is arguably one of the most important tools of regulators, developers, reclamation practitioners, and other stakeholders involved in monitoring and managing the effects of industrial development on terrestrial biodiversity (Gibbons and Freudenberger, 2006; Parkes and Newell, 2003).

In many assessment and monitoring frameworks, vegetation "quality" refers to the degree to which vegetation at a site resembles native vegetation in the absence of human disturbance (Gibbons and Freudenberger, 2006). Vegetation structured by natural disturbances (such as flooding and avalanches) or maintained by pre-European human

* Corresponding author at: Hardner & Gullison Associates, LLC, 15 Woodland Drive, Amherst, NH 03031, USA.
*E-mail address:* bboyle@arizona.edu (B.L. Boyle).

https://doi.org/10.1016/j.ecolind.2023.111510
Received 20 October 2023; Received in revised form 20 December 2023; Accepted 26 December 2023
Available online 5 January 2024
1470-160X/

intervention (such as fire-managed grasslands in North American and Australia) are also generally regarded as native vegetation (Gibbons and Freudenberger, 2006; Lewis et al., 2018; Parkes and Newell, 2003). Although the value of native vegetation *per se* is uncontroversial (Gibbons and Freudenberger, 2006, Landres et al., 2014), debate still surrounds the questions of how to measure the condition of vegetation which has been disturbed by human activity (Cook et al., 2010; Gibbons and Freudenberger, 2006; McElhinny et al., 2005) and how to select the appropriate reference vegetation to which the disturbed vegetation is compared and restored (Demeo et al., 2018).

As part of a broader program of biodiversity monitoring and management at Teck Coal Limited's (Teck) Elk Valley steelmaking coal mining operations in southeastern British Columbia, Canada, we developed an accounting framework for quantifying impacts and improvements to native vegetation in terms of quality (as defined above) and quality hectares (quality x area; Rio Tinto, 2008; see Sahley et al., 2017, for an example application). The framework focuses on natural ecosystems, with quality measured empirically relative to undisturbed native vegetation, and reflecting not only average conditions but also variation at the landscape scale. In terms of sensitivity, the method needed to be capable of distinguishing major vegetation types and successional (seral) stages, as well as different types of disturbance—all at temporal and spatial scales relevant to large mining operations. Field methods needed to be efficient, cost-effective, and—ideally—compatible with standard protocols to allow the re-use of existing vegetation data. Most importantly, the framework needed to be repeatable, verifiable, capable of estimating uncertainty, and minimally vulnerable to observer bias. Although initially developed in the specific context of Teck's Elk Valley mining operations, the framework and associated methodology is intended to be widely applicable, and is currently in use at a variety of projects involving multiple companies in Canada, USA, Suriname, Chile and Greece, and in vegetation ranging from boreal forest to desert scrub to lowland tropical rainforest. Here we describe this monitoring framework, which we call Vegetation Quality Assessment (VQA).

## 2. Existing loss-gain vegetation quality frameworks

We identified several existing approaches for quantifying losses and gains in vegetation or habitat quality. These included Habitat Hectares (Parkes and Newell, 2003), BioCondition (Eyre et al., 2011), Biometric (Gibbons et al., 2008) and Quality Hectares (Rio Tinto, 2008), among others. These approaches typically measure vegetation quality by scoring ecological indicators relative to "benchmarks" representing the expected conditions in undisturbed native vegetation. Benchmark values are determined by sampling undisturbed vegetation adjacent to the project site or by reference to standard values for each vegetation or habitat type, as provided by regulatory agencies (e.g., The State of Queensland, 2014). Multiple indicator scores are combined in various ways to produce an index of overall vegetation quality (Gibbons and Freudenberger, 2006). Quality is then multiplied by area to provide a discounted measure of the amount and condition of vegetation to be compensated through restoration at the project site and offsets elsewhere. We follow Rio Tinto (Rio Tinto, 2008) in referring to quality-discounted area as "quality hectares" (QH).

All existing methods failed to meet one or more of our key requirements. The most pervasive shortcoming was lack of a statistical framework. Sample plots were often located deliberately rather than at random, and replication, when mentioned at all, was on the order of 1–3 plots per vegetation type, with no explicit justification for the sample sizes used (e.g., Eyre et al., 2011). In the absence of replication and randomization at the appropriate temporal and spatial scales, it is impossible to assess the degree to which the samples represent the site as a whole (Hurlbert, 1984). Furthermore, inadequate replication and lack of randomization increase vulnerability to observer bias; for example, field crews could accidentally or deliberately inflate reclamation quality scores by situating plots in microsites with better-than-average vegetation recovery. Another consequence of lack of replication is the inability to measure error or assess statistical significance. Quality is frequently reported without measures of variance (Eyre et al., 2011; Parkes and Newell, 2003) as are benchmark values provided by regulatory agencies (e.g., The State of Queensland, 2014).

A second shortcoming was the use of custom field protocols specific to each framework, thereby precluding the use of existing vegetation data. In many countries publicly-accessible repositories of vegetation data can be extensive, representing essentially all types of native vegetation (e.g., Peet et al., 2012). In the case of Teck's Elk Valley mining operations, the ability to leverage as reference data an existing provincial database of thousands of standardized vegetation plots (British Columbia Forest Service – Research Branch, 2016a) was a key goal of our assessment framework. Without this critical resource we would have faced years of costly data collection—a commitment of time and resources that many companies would find prohibitive. In addition, the sampling methodology commonly employed for vegetation plots within the provincial vegetation database (British Columbia Ministry of the Environment, 2021; Terrestrial Ecosystem Mapping Alternatives Task Force for the Resources Inventory Committee, 2010) was also used by Teck over many years for mapping vegetation within the Elk Valley. The ability to re-purpose such pre-existing data for vegetation assessment would represent a tremendous saving of time and expense.

Many of the methods we reviewed omitted important indicators of vegetation condition. For example, BioCondition omits species identifications in favor of "key attributes or surrogates of biodiversity values that can be rapidly measured in the field" (Eyre et al., 2011), with the goal of enabling surveys to be "undertaken rapidly by a range of natural resource managers…not just botanical ecologists" (Parkes and Newell, 2003). However, any potential gains in speed and ease enabled by such shortcuts must be weighed against the information lost by ignoring taxonomic composition. Species composition is a fundamental attribute by which vegetation units are defined in many classification systems (e.g., Jennings et al., 2009), and indicators based on taxonomic identification are sensitive barometers of forest health, succession and disturbance. For example, small-scale disturbances can leave imprints on species richness that remain detectable for 100 years or more (Curtin, 1995). Species composition discriminates at fine temporal scales among regenerating forests of different ages (Kappelle et al., 1995). Major classes of temperate and tropical vegetation are readily distinguished by taxonomic composition, even at coarse taxonomic levels such as genus and family (Gentry, 1995, 1993, 1988). Finally, field methods that discard the need for taxonomic expertise do not necessarily result in gains in efficiency. For example, the FS1333 Site Visit protocol employed extensively in British Columbia (Terrestrial Ecosystem Mapping Alternatives Task Force for the Resources Inventory Committee, 2010) and used for all vegetation plots in our example application (Section 4), includes identification and estimation of relative abundance of all plant species in the plot, in addition to structural and soil measurements. Yet the 1–2 h taken by experienced practitioners to complete a Site Visit vegetation survey (Terrestrial Ecosystem Mapping Alternatives Task Force for the Resources Inventory Committee, 2010) is comparable to the 2 h typically required for a BioCondition plot (Eddy et al., 2011), which omits most species-level identifications but measures numerous non-taxonomic indicators.

In developing a vegetation quality framework within the specific context of the Elk Valley, we sought a method that would allow us to use data collected previously as part of Teck's ongoing vegetation mapping and data collection program, and to leverage as benchmark data the vegetation inventories maintained by the British Columbia Ministry of Environment and Climate Change Strategy (BC ENV) and used to develop the provincial Biogeoclimatic Ecosystem Classification (British Columbia Forest Service – Research Branch, 2016b). In addition, as our ultimate goal is to deploy the framework worldwide, we needed a method that would be applicable to all vegetation types, temperate and

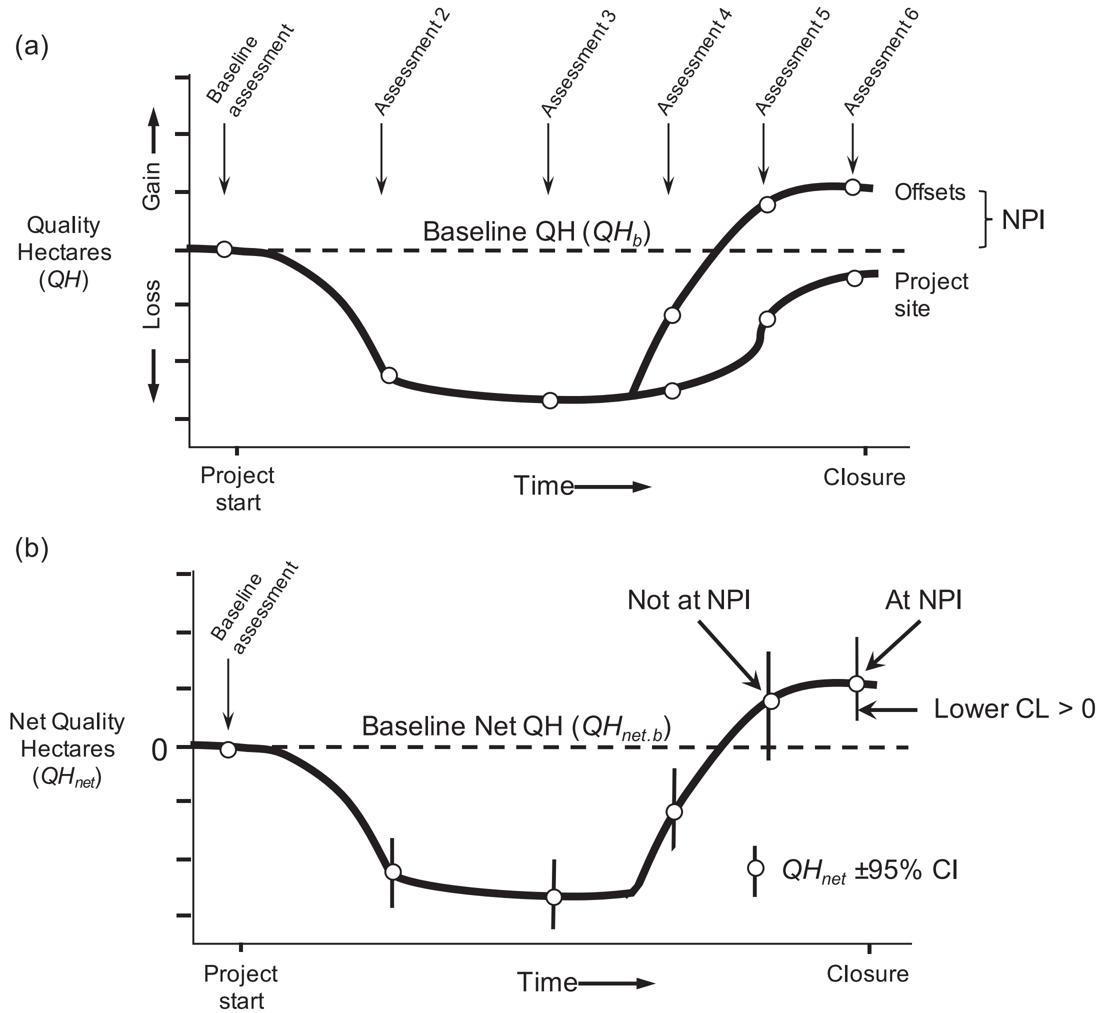


**Fig. 1.** The VQA workflow. (a) Change in Quality Hectares (QH) over time for a single vegetation type is measured by a series of QH assessments, beginning with a baseline assessment (Baseline QH) prior to any project impacts. Additional QH from offset sites (upper line) are added to QH at the project site to help achieve a Net Positive Impact (NPI). (b) Losses and gains expressed as Net QH (current QH - baseline QH, or $QH_{net}$), with 95% confidence intervals (CI) indicated by vertical bars. Baseline $QH_{net}$ is, by definition, zero, and NPI is achieved when the lower 95% confidence limit (CL) of $QH_{net}$ exceeds zero.

tropical, and compatible with a wide variety of commonly-used vegetation inventory methods.

## 3. Overview of VQA

### 3.1. Summary of key features

The framework we developed combines useful features of existing methods—including quality measured as similarity to undisturbed native vegetation and an accounting workflow based on QH—with a stratified-random sampling design that enables estimation of error and inference at the scope of the entire project site. Losses due to project impacts and gains from restoration and offsets are measured relative to a baseline of vegetation quality and area prior to project initiation (note that baseline is not same as benchmark; the baseline is simply whatever vegetation was present prior to project impacts, and may be pristine, disturbed, or even absent). Multiple later assessments at project and offset sites track net quality hectares ($QH_{net}$) over time, with the ultimate goal of detecting the point at which baseline QH have been exceeded and NPI has been achieved (Fig. 1). A key feature of the method is the measurement of quality in terms of overlap between indicator distributions of project site vegetation and in benchmark-quality vegetation of the same type. The approach can be applied to both forested or non-forested vegetation using widely-used vegetation sampling methods. Below, we outline the steps of the VQA workflow, followed by a trial implementation at Teck's Elk Valley operations.

### 3.2. Loss-gain accounting

The complete VQA workflow (Fig. 1) consists of a series of loss-gain assessments of quality hectares of each vegetation type present at the project or offset site. Quality hectares are calculated as

$$QH = Q\text{x}A \quad (1)$$

where $QH$ is quality hectares, $Q$ is vegetation quality and $A$ is the area of that vegetation in hectares. As $Q$ varies from 0 to 1, $QH$ ranges from 0 to $A$ hectares.

The workflow begins with a baseline assessment of QH for each class of vegetation prior to project start. At each subsequent assessment, the net change in QH relative to baseline ($QH_{net}$) is determined by subtracting baseline quality hectares ($QH_0$) from current quality hectares ($QH$) for the same vegetation:

$$QH_{net} = QH - QH_0 \quad (2)$$

Typically, with resource developments such as mines, not all losses at the project site can be restored. These residual impacts will need to be compensated by additional QH gained through conservation or restoration actions at one or more offset sites (Fig. 1a). Each offset site will require its own set of assessments, including a baseline assessment and separate calculations of net QH. Unlike the project site, offset assessments generally need to be adjusted for one or more counterfactual scenarios, such that only *additional* QH directly attributable to the offset program are included. In other words, existing QH that would have persisted into the future, or increases in QH that would have occurred in the absence of the offset program, must be excluded. The simplest counterfactual is complete elimination of an existing natural area. For example, if purchase and protection of an offset prevents a credible scenario of permanent conversion to agriculture resulting in complete loss of all native vegetation, then the baseline quality for the offset would be zero and net offset QH at each assessment would be simply the observed QH.

For projects that include offsets, overall net QH is the sum of net QH at the project site and at each offset site. For a project with two offsets, net quality hectares is

$$QH_{net} = QH_{net.p} + QH_{net.o1} + QH_{net.o2} \tag{3}$$

where $QH_{net}$ is overall net QH, $QH_{net.p}$ is net QH of the project site, and $QH_{net.o1}$ and $QH_{net.o2}$ are net QH of offset sites 1 and 2, respectively. Each assessment of overall net QH thus includes losses due to project impacts, gains due to restoration and natural succession at the project site, and gains from averted losses and restoration at offset sites.

In theory, attaining NPI requires that $QH_{net} > 0$ for all native vegetation types by the final assessment. However, each of the many measurements involved in measuring vegetation quality (Section 3.6.4) has its own associated measurement error that propagates to all derived values. In VQA, each estimate of Quality, QH and $QH_{net}$ is expressed as a mean with an associated 95 % confidence interval (CI; Section 3.6.4). In practice, therefore, NPI cannot not be assumed until the lower 95 % confidence limit (CL) of $QH_{net}$ exceeds 0 (Fig. 1).

### 3.3. Stratification of vegetation, disturbance and reclamation

Each assessment cycle requires a comprehensive classification and mapping of all land cover at the project site. Vegetation may be further subdivided into naturally occurring seral stages, if successional vegetation is a major component of natural landscapes in the region. For example, early successional forest can account for large areas in regions subject to frequent, naturally occurring fire (Fulé et al., 1997; Korb et al., 2003). Each class of vegetation (plus seral stage, if applicable) may be further subdivided into disturbance classes expected to follow distinct successional trajectories. Different restoration treatments should also be monitored separately if recovery outcomes and timelines are expected to differ, or if a goal of the program is to compare differences among treatments.

Each class of vegetation, disturbance, and reclamation treatment (if applicable) represents a different sampling stratum. Within each stratum, the required number of sampling points (plots) should be located at random using GIS software (the procedure for estimating minimum sample sizes is described in Section 3.6.5). If the initial classification results in a large number of strata, adequate sample sizes may be difficult or impossible to achieve given available time and resources. In such cases, it may be necessary to reduce the number of strata (thereby increasing per-stratum sample sizes) by combining ecologically-similar vegetation types.

Each assessment thus requires its own map of the project site, classified into strata of vegetation, disturbance and reclamation. Once the mapping process is complete, the actual area of each stratum is determined by summing the areas of all polygons belonging to that stratum and the resulting total multiplied by the quality of the stratum (Section 3.6) to determine the stratum's current QH for that assessment. Finally, the QH of all strata sharing the same benchmark vegetation are summed to provide the total current QH for each class of native vegetation. In general, a project pursuing NPI seeks to reestablish vegetation of the same type that existed prior to project impacts; QH, net QH and NPI are therefore assessed independently for each class of benchmark vegetation ("like-for-like"; see Text S1 for a discussion of exceptions).

Ideally, the baseline map and baseline assessment are completed prior to initiation of project-related impacts. However, if the VQA framework is implemented after the project has begun, modeling ("back-casting") can be used to reconstruct vegetation present prior to project impacts. An example of this approach is described in detail in Knopff and Franklin (2018a) and demonstrated in the Teck VQA application outlined below.

### 3.4. Vegetation sampling

Once mapping is complete and sampling strata have been defined, sampling localities for plots within strata are assigned randomly using GIS software. The number of plots per stratum can be estimated using Monte Carlo methods, as described below. We refer to all plots from the project or offset sites as "focal plots".

In addition to focal plots, determination of quality requires reference samples of plots in undisturbed examples of the same types of native vegetation present at the project site. These plots, which we call "benchmark plots", can be established in areas adjacent to the project site (if undisturbed native vegetation is present), or obtained from existing vegetation inventory databases. Regardless of source, benchmark plots must meet the following three requirements: (1) indicators and sampling methods should be identical with those of the focal plots, or they can be made compatible through transformation, (2) the vegetation sampled can be reliably matched to native vegetation present at the project site (i.e., the same vegetation classification was used for or can be applied to all plots), and (3) sampling locations are minimally affected by human disturbance, based on accompanying metadata or expert evaluation. Ideally, experts not involved with the project should be responsible for verifying that plots or localities sampled represent undisturbed native vegetation. In calculating quality, multiple samples of focal plots representing distinct classes of disturbance and reclamation of the same native vegetation type are compared to the same set of benchmark plots.

The VQA framework is compatible with a variety of vegetation sampling methods. In many cases it may be possible to combine different field methods, as long as indicators measured are compatible. Legacy data collected using different protocols may be used, but only if the relevant indicators can be transformed or otherwise normalized to produce equivalent measurements. For example, rarefaction (randomly subsampling individuals, down to the smallest number of individuals recorded among all plots; Magurran, 2004) can be used to normalize species richness among tree plots of different area—if the plots consist of measurements of individual trees and the minimum stem diameter is the same for all plots. However, rarefaction does not apply to cover plots, which record percent cover of each species at the level of the entire plot; species richness cannot be normalized and therefore should not be compared among cover plots of different sizes.

### 3.5. Indicator selection

Unlike some existing approaches, VQA does not mandate specific indicators. This indicator-agnostic approach allows reuse of existing data and encourages sharing of newly-collected reference data, ideally by depositing them in open access vegetation data repositories (e.g., VegBank; Peet et al., 2012). However, maintaining compatibility with existing data will often favor the use of long-established vegetation inventory methods, with indicators based on measurements of relative or absolute abundance of species, growth forms, and size classes (e.g., vegetation height or tree stem diameters). Fortunately, these attributes are also sensitive indicators of disturbance history (Gauthier et al., 2010; Ross et al., 2002) and successional age (Kappelle, 1995; Kappelle et al.,

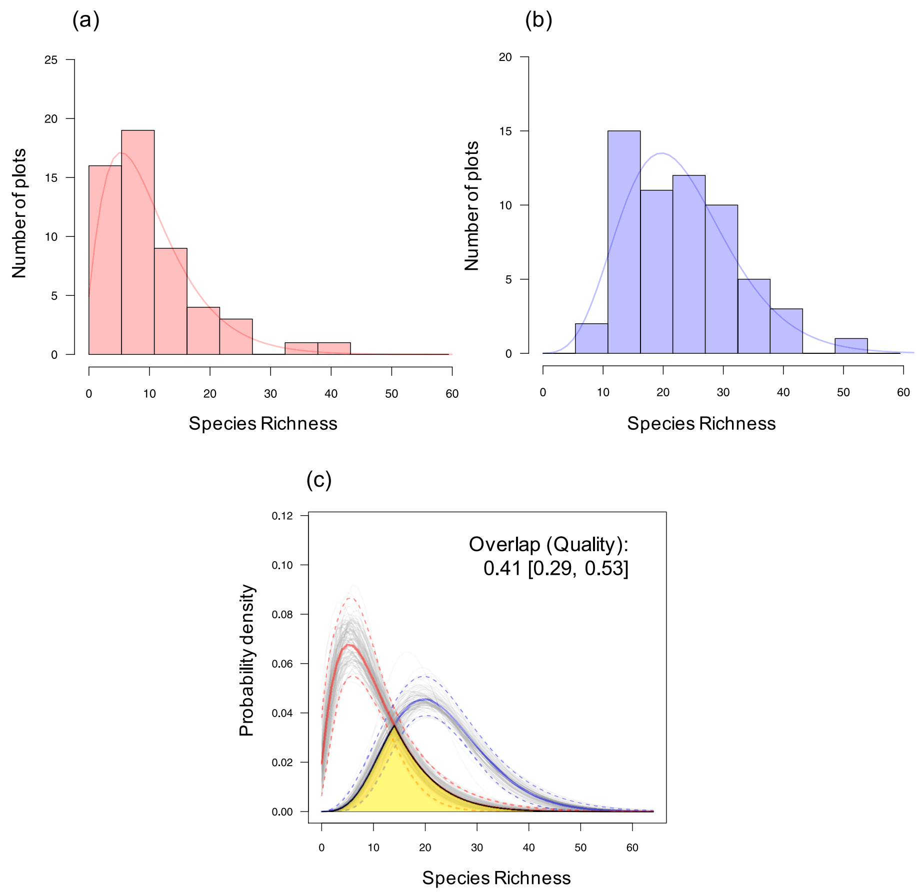


**Fig. 2.** Calculation of overlap-based quality. In (a) and (b), separate probability density functions (pdfs) are fit to the focal and benchmark sampling distributions, respectively, using maximum likelihood. In (c), quality [95% CLs in square brackets] is calculated as the overlap of the two pdfs. Dashed lines in (c) are the bootstrapped estimates of the 95% CLs of the pdfs; pale gray lines are a small sample of the individual bootstrap pdfs. CLs: confidence limits.

1996; see also Section 2). Furthermore, taxonomic composition and vegetation structure are defining characteristics of many vegetation classification systems (e.g., Jennings et al., 2009; Mucina et al., 2016) and, depending upon the classification used, may be essential for verifying that the reclaimed vegetation is the same vegetation type as the benchmark vegetation.

For applications where the goal is to measure similarity to undisturbed native vegetation, a small set of indicators of vegetation composition (e.g., native species richness, abundance-weighted taxonomic composition) and structure (e.g., percent cover of herbs, shrubs, trees, etc., percent cover by vegetation height strata, counts of tree stems by diameter classes) is generally sufficient to discriminate among different vegetation types, disturbance histories and successional ages (see citations in preceding paragraph). Indicators of invasion by non-native species (e.g., percent cover of exotic species) represent a different and important aspect of ecosystem integrity and may be required by local or national regulations (Weidlich et al., 2020). Others indicators may be included if well-supported in the scientific literature as discriminating among classes of disturbance and successional age, but the information gained should be weighed against increased per-sample effort, time and cost. With respect to the narrow goal of measuring similarity to natural communities, attributes of the physical environment, such as leaf litter, organic soil, soil nutrients, etc., are indirect indicators of vegetation condition and contribute little information beyond that provided by compositional and structural attributes of the plant community itself (see Karr, 1981; Karr et al., 1986). However, some environmental indicators may be requested or mandated by clients or regulators for other purposes, such as the provision of habitat features important for wildlife (e.g., standing dead trees; Parkes and Newell, 2003) or the establishment of organic soil during early reclamation (see example application, Section 4.2.4).

Finally, some applications may warrant the use of indicators other than those recommended above. For example, if the goal of restoration is to improve stand health of pine forests, indicators of infection by pine beetles, blister rust, and other disease agents may be more informative than indicators of composition and structure. Fixed benchmark values representing the disease-free state (e.g., 0 % pine beetle-infected trees) may be more practical or meaningful than comparisons to empirical benchmark distributions—especially if disease-free vegetation no longer exists (see Section 3.6.2). In summary, the final choice of indicators in VQA will be driven by the goals and constraints of the particular application.

### 3.6. Quality calculations

#### 3.6.1. Overlap-based quality

The principal measure of quality within the VQA framework is overlap between the probability distributions of indicators in the focal and benchmark vegetation. Measuring similarity in terms of overlap between probability distributions has a long history in economics (e.g., comparison of incomes; Gastwirth, 1975). The overlap method accommodates a wide variety of normal and non-normal distributions, and deserves to be more widely used—especially in ecology, where skewed and bounded distributions are the norm rather than the exception (Magurran, 2004). Overlap between focal and benchmark indicator distributions provides an intuitively meaningful index of similarity ranging from 0 to 1, where 1 (100 % overlap) means, effectively, "indistinguishable from benchmark". Furthermore, distribution overlap captures the full range of natural variation, including asymmetry not evident in comparisons of means (Fig. 2).

To calculate overlap, each indicator must first be assigned a distribution family based on the domain of the variable and visual inspection. Appropriateness of the distribution can be formally assessed using a goodness-of-fit test such as the one-sample Kolmogorov-Smirnov test (Berger and Zhou, 2014). Commonly measured indicators generally fall into one of three distributions: (1) negative binomial (counts such as species richness), (2) gamma (zero-bounded continuous measurements such as taxonomic similarity/dissimilarity), and (3) beta (percentages or proportions such as percent cover of species or growth forms). As the negative-binomial is a discrete distribution, it is approximated by the gamma distribution when calculating distribution overlap (Best and Gipps, 1974).

Overlap is calculated in a two-step process. In step one, separate univariate probability density functions (pdfs) are fit to the focal and benchmark samples using maximum likelihood estimation (Fig. 2a; see Venables and Ripley, 2002). In step two, the area of the intersection of the two distributions is determined by integration. Confidence limits of the overlap-quality estimate are determined by bootstrapping these calculations, using the bootstrap deviance approach described below (Section 3.6.4).

### 3.6.2. Alternative quality methods

Although the "overlap quality" method described above accommodates the majority of indicators, three alternative approaches are available to handle special cases. The first of these, "one-tailed quality", calculates quality using values either above or below the benchmark mean, but not both. Depending on the goals of the application, one-tailed quality may be appropriate for indicators such as Species Richness if high values of richness are considered desirable; in this case, quality would be calculated using values below the benchmark mean only (i.e., the "lower tail"). Details of the one-tailed quality algorithm are provided in Text S2 and Fig. S1.

"Fixed-benchmark quality", which applies to beta-distributed indicators only, uses a single fixed benchmark value. Quality is calculated as the absolute value of the difference between the benchmark value and the focal mean, divided by the maximum possible difference on the [0,1] domain (Fig. S2a), with an additional, non-linear transformation that causes quality to drop steeply with increasing distance from benchmark (Fig. S2b). Fixed-benchmark quality is appropriate when a single, ideal benchmark value is readily apparent and appropriate. Examples of indicators for which a fixed benchmark might be appropriate include percent cover of invasive species and infection rates of diseases (e.g., "Percent of trees infected white pine blister rust"); in both cases, the benchmark is 0. Details of the calculation of fixed-benchmark quality are provided in Text S3 and Fig. S2.

"Overlap-means quality" combines features of the overlap and fixed-benchmark algorithms to resolve issues caused by zero and one inflation in beta distributed indicators. Zero- and one-inflated distributions are common among ecological indicators measured as percentages (Magurran, 2004). Mixed distributions can distort measures of quality based purely on distribution fitting. The overlap-means algorithm calculates quality twice, once based on distribution overlap and a second time based on the difference between the focal and benchmark mean, and uses a weighted average of the two. The result is a robust quality score which provides an intuitively accurate representation of the similarity of two distributions across the entire beta domain, even for mixed distributions. For details of the calculation of overlap-means quality, see Text S4 and Figs. S3 and S4.

### 3.6.3. Overall quality

Most existing vegetation quality methods combine indicator scores into a single index of overall quality. To reduce the influence of correlated indicators and minimize subjective weightings (see critique in McCarthy and Parris, 2004), we group related indicators into "functional groups" representing key aspects of ecological function. For example, indicators based on taxonomic composition, such as "Species Richness", are assigned to functional group "Composition", whereas indicators that measure the relative abundance of different size classes or growth forms, such as "Percent Cover Herbs" and "Percent Cover Trees", are assigned to functional group "Structure". For each functional group, functional group quality is calculated as the arithmetic mean of all indicator qualities within that functional group. Overall quality is then calculated as the geometric mean of the functional group qualities, as follows:

$$Q = \left( \prod_{i=1}^{N} Qf_i \right)^{\frac{1}{N}} \tag{4}$$

where $Q$ is overall quality, $Qf_i$ is the arithmetic mean of indicator qualities in functional group $i$, and $N$ is the number of functional groups.

The use of the geometric mean addresses the criticism that existing approaches are vulnerable to manipulation of quality scores through the substitution of one indicator for another (McCarthy and Parris, 2004). Under the VQA approach, a single functional group whose mean drops to zero will force overall quality to zero as well, regardless of the values of the other functional group means. This behavior ensures that all key components of ecological function have non-zero quality. At the same time, the use of the arithmetic mean for functional group quality prevents indicators within the same group from forcing overall quality to zero during early succession, when cover (and quality) of herbs and shrubs recovers long before cover (and quality) of large trees becomes non-zero.

### 3.6.4. Confidence limits and significance testing

In VQA, the 95 % CLs of indicator quality, functional group quality, overall quality and QH are estimated using the bootstrap deviance method of Efron (1979). Each quality calculation is repeated 10,000 times, sampling the actual data with replacement. This method accurately represents asymmetric confidence intervals (CIs) that commonly result from overlap of distributions such as the beta distribution.

Under certain circumstances, the 95 % CLs can be used directly to evaluate significant differences. For example, the difference between two estimates of quality or QH is significant if the CIs of the difference of the estimates do not overlap 0 (Cousineau, 2017). In VQA, we assess progress to NPI by examining the position of the lower 95 % CL of the difference between current and baseline QH (i.e., Net QH). Once the lower CL exceeds 0, we can be 95 % confident that $QH_{current} > QH_{baseline}$ and NPI has been attained, given the variability of our estimates of Quality and QH (Fig. 1b).

### 3.6.5. Effect size, power, and minimum sample size

The difference between the mean and the 95 % CL, or margin of error (MOE), is an intuitive measure of effect size, precision and error which deserves to be more widely used (Cousineau, 2017). Because MOEs calculated using the bootstrap deviance method are generally asymmetrical, we use the largest of the two MOEs of quality as the "minimum detectable effect size" (MDES) attainable at a given sample size. Defining effect size in this way, we can use Monte Carlo methods to estimate the minimum sample size ($n$.min) required to detect a given target MDES ($MDES_{exp}$) at $\alpha = 0.05$ and a conventional power of 0.80 (see Text S5 for details). This approach is also illustrated in the example application below. Throughout, we follow the recommendations of Cumming (2014) and Cousineau (2017) in presenting upper and lower CLs in square brackets.

**Table 1**
Benchmark and focal plot sample sizes for all vegetation classes (ecosystem groups) within the study area, for the baseline and current (reclamation) assessments. Sample sizes will be smaller than shown for individual indicators not measured in all plots.

| | | Focal plots | |
|---|---|---|---|
| Ecosystem group | Benchmark plots | Baseline native vegetation | Current reclaimed vegetation |
| Alpine | 10 | 81 | |
| Alpine Dwarf Shrub | 46 | 30 | |
| Alpine grassland | 4 | 41 | |
| Alpine meadow | 2 | 15 | |
| Avalanche feature | 100 | 24 | |
| Brushland/Grassland | 62 | 61 | 98 |
| Deciduous floodplain | 0 | 4 | |
| Dry forest, Early-mid | 79 | 95 | 105 |
| Dry forest, Mature | 55 | 70 | |
| Dry forest, Old | 14 | 4 | |
| Herb meadow | 0 | 6 | |
| Intermediate forest, Early-mid | 148 | 220 | 10 |
| Intermediate forest, Mature | 83 | 216 | |
| Intermediate forest, Old | 28 | 22 | |
| Krummholz | 15 | 9 | |
| Rock/Talus | 64 | 60 | |
| Shrubland | 2 | 6 | |
| Wet forest, Early-mid | 70 | 46 | 9 |
| Wet forest, Mature | 77 | 54 | |
| Wet forest, Old | 28 | 11 | |
| Wetland | 72 | 46 | |

## 4. Example application: A VQA accounting of baseline & reclaimed vegetation at Teck Elk Valley mine operations

### 4.1. Goals of the application

As a trial application of the VQA framework, we performed assessments of baseline and current reclaimed vegetation quality for Teck's steelmaking coal mining operations in the Elk Valley of British Columbia, Canada. The baseline assessment estimated quality and extent of all vegetation prior to mining-related impacts within the Elk Valley. The current assessment measured present-day extent and condition of reclaimed vegetation within the study area, across all five Teck Elk Valley operations. The goals of the application were four. The first goal was to demonstrate the retroactive application of the VQA framework, using modeling approaches to reconstruct terrain and vegetation originally present within an area already impacted by mining activities (see Knopff and Franklin, 2018, for details). The second goal was to develop a best estimate of the baseline extent and condition of all native vegetation impacted by Teck's operations in the Elk Valley over the period 1950 – 2018. The third goal was to demonstrate the use of existing vegetation data within the VQA framework by combining focal data from vegetation surveys previously collected from the project site with benchmark data from a public vegetation inventory database. The fourth goal was to estimate the current contribution of reclamation to NPI, with the aspirational goal of ensuring that gains from reclamation, revegetation and biodiversity offsets ultimately exceed impacts to native vegetation within the combined mine footprints.

### 4.2. Methods

#### 4.2.1. Study site

The study area encompassed the combined past and current footprints of all Teck steelmaking coal mining operations within the Elk River Valley ("Elk Valley") of southeast British Columbia. The Elk Valley is located along the western edge of the Rocky Mountains, in terrain dominated by steep slopes and narrow valley bottoms. The area is mostly forested, with shrubby and herbaceous subalpine and alpine vegetation occurring at the highest elevations and on steep slopes subject to snow avalanches. The most common vegetation types include the Engelmann Spruce–Subalpine Fir Dry Cool variant and Engelmann Spruce–Subalpine Fir Dry Cool Woodland subzone (Meidinger and Pojar, 1991). Teck's steelmaking coal operations within the Elk River Valley include four active mines and one under care and maintenance and have a combined area of c. 19,000 ha. This total includes areas currently disturbed by mining plus areas that were disturbed in the past but are now reclaimed or undergoing reclamation, through 2018.

#### 4.2.2. Vegetation mapping and classification

For the baseline assessment, vegetation of the mine footprint was reconstructed using a combination of archival aerial photos, satellite imagery, and Predictive Ecosystem Mapping (PEM; Terrestrial Ecosystem Mapping Alternatives Task Force Resource Inventory Committee (RIC) 1999) as well as terrain reconstruction and hydrological and soil models. The resulting back-cast vegetation polygons were combined with maps of existing vegetation to produce a comprehensive baseline map of pre-mine vegetation conditions within all Elk Valley operations administered by Teck (Knopff and Franklin, 2018). All vegetation units were classified to the site series level, using the Biogeoclimatic Ecosystem Classification (BEC) version 10 (British Columbia Forest Service – Research Branch, 2016b; MacKenzie, 2012). Site series, defined under the BEC system as "all sites within a biogeoclimatic subzone or variant that are capable of producing the same mature or climax vegetation unit" (British Columbia Ministry of Forests, 2023), are roughly equivalent to plant associations in other systems, such as the US National Vegetation Classification (e.g., Jennings et al., 2009). Seral stage classes were assigned to forested vegetation using BC Ministry of Environment and Climate Change Strategy standard terminology (Terrestrial Ecosystem Mapping Alternatives Task Force for the Resources Inventory Committee, 2010).

As detailed information on vegetation condition at the baseline year was not available, we used the condition of current vegetation outside the mine footprint to represent vegetation condition within the mine footprints at baseline. We believe this assumption is conservative as land management practices have generally improved over time and anthropogenic fire along with other large-scale land disturbance practices have decreased; therefore, current condition should be as good as, or better than, condition at the baseline year (although we acknowledge that fire may have contributed to increased quality for drier forest types structured by fire in pre-European contact times). For much of the 20th century, the extent of old growth forest within the Elk Valley was considerably less than under historical disturbance regimes prior to European settlement in the mid-1850 s. In particular, the lack of old growth Montane Spruce (MS) and Interior Cedar Hemlock (ICH) forest reflects a combination of very large and severe fires in 1919 and in the 1930 s, forest clearing for urban/rural development and agriculture, and timber harvesting that has been ongoing since the early 1900s (Holmes et al., 2018). As we did not subdivide native vegetation into subcategories of disturbance, each vegetation type is represented by exactly one focal land cover class only.

The resulting vegetation and land cover map for the study site consisted of 133 site series (see definition above). Including structural stages of forested vegetation, the final number of distinct land cover classes (site series + structural stages) was 396. As this number of sampling strata was too large to be sampled at adequate sample sizes, we revised the baseline map, combining site series into a smaller number of classes (hereafter, "ecosystem groups") representing vegetation of similar structure, composition and moisture availability. For example, site series representing above-treeline alpine vegetation were combined into the ecosystem group "Alpine". In addition, for forested vegetation, the seven BEC structural stages were combined into the three seral stages "Early-mid", "Mature" and "Old". We used separate categories for early-mid and mature forest to support restoration and maintenance of

**Table 2**
Ecological indicators used for the example VQA application, their probability distributions and quality scoring methods.

| Indicator | Distribution | Quality algorithm | Test tail | Functional group |
|---|---|---|---|---|
| Species Richness | Negative binomial | Overlap | Lower | Composition |
| Taxonomic Distance | Gamma | Overlap | Both | Composition |
| Percent Cover Exotic Species | Beta | Fixed (0) | n/a | Composition |
| Percent Cover Herbs | Beta | Overlap-means | Both | Structure |
| Percent Cover Moss | Beta | Overlap-means | Both | Structure |
| Percent Cover Shrubs | Beta | Overlap-means | Both | Structure |
| Percent Cover Trees | Beta | Overlap-means | Both | Structure |
| Percent Cover Decomposing Wood | Beta | Overlap-means | Both | Environment |
| Percent Cover Organic Soil | Beta | Overlap-means | Both | Environment |
| Percent Cover Surface Water* | Beta | Overlap-means | Both | Environment |

*Indicator "Percent Cover Surface Water" used for "Wetland" vegetation only.

naturally-structured successional forest, in addition to old growth. This reflects Teck's broader goal of restoring the historical range of variability within the Elk Valley, including the landscape scale distribution of fire-, disease- and avalanche-maintained patches of early seral forest (Davis, 2009; Keane et al., 2009). Forested seral stages were therefore assigned benchmarks of the same seral stage, rather than comparing all seral stages to old growth—as would be the case if old growth forest was the only restoration objective. Comparison to intermediate seral stage benchmarks also enables early detection of arrested succession and other departures from expected trajectories toward old growth vegetation (Soto and Puettmann, 2020).

After combining site series into ecosystem groups, the final vegetation and land cover map consisted of 21 ecosystem groups plus seral stage sampling strata (Table 1). Five sampling strata (Alpine grassland, Alpine meadow, Deciduous floodplain, Herb meadow and Shrubland) had focal or benchmark sample sizes that were too low (0–4 plots) to allow analysis. The total area of all units within the footprint study area was 18,978.9 ha.

For the current vegetation assessment, all reclaimed vegetation types within the footprint were assigned a target benchmark vegetation and the total area of each reclamation class was determined. Reclaimed vegetation fell into four ecosystem groups: "Brushland/Grassland", "Early-mid Dry Forest", "Early-mid Intermediate Forest" and "Early-mid Wet Forest". As our accounting encompassed past, present and future impacts through 2035, including planned expansions, vegetated areas currently falling within the footprint of future mine expansions were coded as "non-vegetated" and included with the un-reclaimed areas within the current mine footprint. Total area of reclaimed vegetation was 2,109.0 ha, with the remaining non-vegetated areas totaling 16,869.9 ha.

#### 4.2.3. *Vegetation plots*

The baseline focal plots were sampled over a 41-year period postdating the baseline year, from 1975 to 2016, in native vegetation adjacent to the project site (i.e., the current mine footprint). These samples provide a conservative estimate of the condition of native vegetation present prior to mining impacts (Section 4.2.2). These data were collected by multiple consulting companies as part of Teck's ongoing vegetation mapping and inventory program within the Elk Valley. Only plots collected using the standard BC ENV Site Visit methodology (Terrestrial Ecosystem Mapping Alternatives Task Force for the Resources Inventory Committee, 2010) were used; all others were removed.

Focal plots for the current reclamation assessment were sampled in reclaimed vegetation within the project footprint. These plots are located in multiple areas representing different reclamation treatments. For many of them, reclamation was initiated more than a decade ago. Unlike current practices, many historical reclamation activities did not have restoration of native vegetation as a primary goal, and included planting of non-native agronomic species, particularly in grassland ecosystems targeted to support an ungulate end land use. Despite potential differences in reclamation methods and initiation times, we did not distinguish among different reclamation treatments. Our goal for this preliminary VQA application was to gain a general understanding of the current state of historical reclamation for broad classes of vegetation. This information will be used to plan more fine-grained sampling in future assessments, and enable adaptation of reclamation practices to better serve Teck's current focus on native vegetation. All reclamation focal plots were collected using the Site Visit methodology described above.

Benchmark plots were obtained from a database of inventories developed by the BC ENV as reference data for the BEC vegetation classification standard (British Columbia Forest Service – Research Branch, 2016a; MacKenzie, 2012; Meidinger and Pojar, 1991). As with the focal plots, all benchmark plots were collected using the Site Visit methodology.

All plots were originally entered into a Microsoft Access database (Microsoft Corporation, 2020) using a common schema and data entry tool provided by the Province of British Columbia (British Columbia Forest Service – Research Branch, 2018). The separate plot datasets were merged and imported to an SQL Server database (Microsoft Corporation, 2022) used by Teck to manage land cover data for the Elk Valley. Vegetation classifications of both plots and land cover units were then cross-checked and standardized within the database. Only benchmark plots matching at the site series level to native vegetation present within the Elk Valley baseline map were retained (Section 4.2.2).

For the VQA analysis, the combined plot and land cover data were exported as a single Microsoft Access file and imported to R (R Development Core Team, 2011) using the mdb.get function of package Hmisc (Harrell, Frank E et al., 2020). After checking for errors, standardizing vocabulary such as growth form names, converting species codes to scientific names, and updating taxonomy to conform to the BC provincial plant species list (British Columbia Ministry of the Environment, 2016), the data were exported to the set of comma-delimited input files required by the VQA analysis pipeline (Section 4.2.7). Plots with missing data or errors that could not be reliably corrected were flagged and removed from the analysis. The final normalized data set consisted of 1,121 baseline focal plots, 222 reclamation focal plots, and 959 benchmark plots. Table 1 shows the distribution of the benchmark, baseline and reclamation vegetation plots among the 21 ecosystem groups.

With respect to the sampling design of VQA, the use of existing vegetation data means that random sampling cannot be assumed. However, the focal baseline plots were collected over many years by multiple contractors and agencies for various vegetation mapping projects, generally in support of Terrestrial Ecosystem Mapping (TEM) (Resources Inventory Committee – Ecosystems Working Group, 1998). The requirements of the TEM standard and the varied origins of these data reduce—but do not eliminate—the potential for pervasive patterns of observer bias. The focal reclamation plots were located within reclamation treatment polygons using a systematic (grid-based) design, which minimizes observer bias and in some cases provides comparable accuracy to stratified-random sampling (Olufadi et al., 2012; Su-Yim et al., 2009). The existing data therefore provide useful initial estimates of baseline vegetation extent and condition, and the contribution of early reclamation to ongoing efforts to restore and exceed baseline through a combination of onsite reclamation and offsets. Future

**Table 3**
Baseline and current (reclaimed) vegetation quality for the combined Teck Elk Valley Operations footprint. For vegetation for which quality could not be calculated due to insufficient sample size (indicated by asterisk *), a benchmark quality of 1 (100 %) is assumed. Total footprint area (all ecosystem groups combined) is 18,978.9 ha. CLs: confidence limits.

| Ecosystem group | Baseline quality [95 % CLs] | Reclaimed (current) quality [95 % CLs] |
|---|---|---|
| Alpine | 0.79 [0.72–0.88] | 0 |
| Alpine dwarf shrub | 0.76 [0.68–0.85] | 0 |
| Alpine grassland* | 1 | 0 |
| Alpine meadow* | 1 | 0 |
| Avalanche feature | 0.86 [0.81–0.95] | 0 |
| Brushland/Grassland | 0.82 [0.78–0.88] | 0.35 [0.32–0.40] |
| Deciduous floodplain* | 1 | 0 |
| Dry forest, early-mid | 0.81 [0.76–0.90] | 0.42 [0.39–0.46] |
| Dry forest, mature | 0.84 [0.79–0.94] | 0 |
| Dry forest, old* | 1 | 0 |
| Herb meadow* | 1 | 0 |
| Intermediate forest, early-mid | 0.79 [0.76–0.91] | 0.31 [0.28–0.36] |
| Intermediate forest, mature | 0.87 [0.83–0.95] | 0 |
| Intermediate forest, old | 0.76 [0.69–0.83] | 0 |
| Krummholz | 0.74 [0.64–0.88] | 0 |
| Rock/Talus | 0.82 [0.77–0.89] | 0 |
| Shrubland* | 1 | 0 |
| Wet forest, early-mid | 0.75 [0.70–0.83] | 0.26 [0.19–0.32] |
| Wet forest, mature | 0.81 [0.75–0.90] | 0 |
| Wet forest, old | 0.82 [0.74–0.90] | 0 |
| Wetland | 0.83 [0.78–0.90] | 0 |

sampling will adhere to the stratified-random design described in Sections 3.3 and 3.4.

#### 4.2.4. Ecological indicators

From the many measurements collected under the Site Visit plot protocol, we selected ten attributes that were recorded with sufficient consistency across all plots to be used as ecological indicators (Table 2). Three of these (“Species Richness”, “Taxonomic Distance” and “Percent Cover Exotic Species”) were taxonomic-based indicators which we assigned to functional group “Composition”. Four (“Percent Cover Herbs”, “Percent Cover Moss”, “Percent Cover Moss”, “Percent Cover Trees”) reflected aspects of vegetation height and complexity and were therefore placed in functional group “Structure”. As discussed previously (Section 3.5) all of the preceding indicators are supported in the literature as sensitive indicators of disturbance history and successional age. The remaining three indicators (“Percent Cover Decomposing Wood”, “Percent Cover Organic Soil” and “Percent Cover Surface Water”) measured attributes of the physical environment, and were assigned to functional group “Environment”. “Percent Cover Decomposing Wood” was included because large woody debris is regarded by many researchers as an important resource for many plant and wildlife species (Feller, 2003) and an indicator of old growth forest (Lombardi et al., 2012). “Percent Cover Organic Soil” was included at the request of Teck to enable measurement of progress during early reclamation, when organic soil is established and restored through management interventions. “Percent Cover Surface Water”, a critical environmental feature for wetland vegetation, was included in quality calculations of ecosystem group “Wetland” only.

“Species Richness” was the count of all native species recorded in the

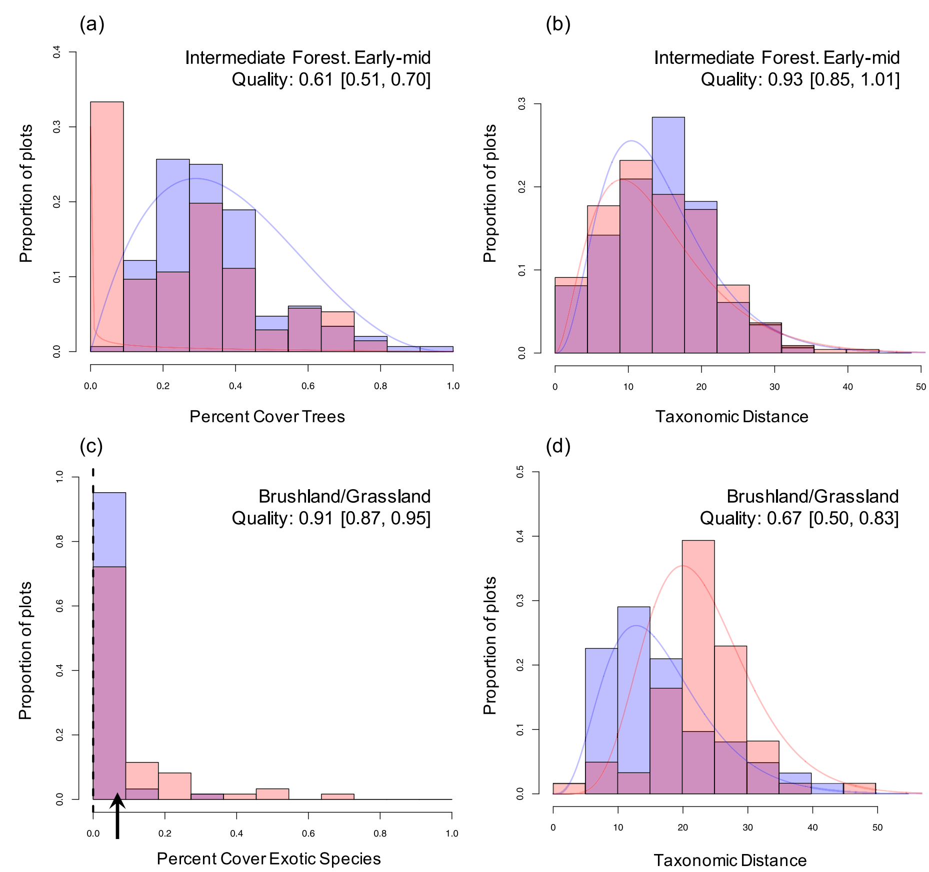


**Fig. 3.** Observed (histograms) and fitted (curves) focal and benchmark distributions of selected indicators for baseline Intermediate Forest, Early-mid (top panels) and Brushland/Grassland (bottom panels), and their quality scores [lower, upper 95% CLs]. Vertical dashed line in (c) indicates the fixed benchmark value of 0 used for this indicator. Arrow in (c) marks the focal mean of 0.77. Pink: focal; blue: benchmark; purple: overlap.

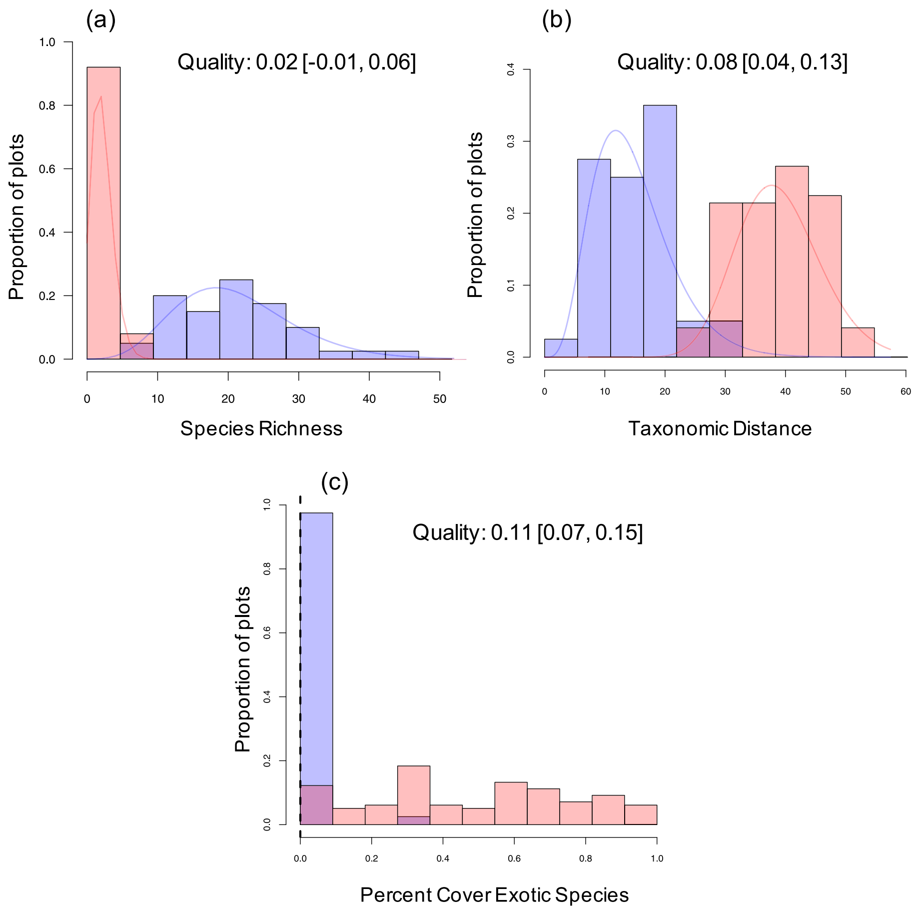


**Fig. 4.** Observed (histograms) and fitted (curves) focal and benchmark distributions of composition indicators for reclaimed Brushland/Grassland, and their quality scores [lower, upper 95% CLs]. Pink: focal; blue: benchmark; purple: overlap.

plot. "Percent Cover Exotic Species" was the aggregate, plot-level cover of all non-native species combined. Native and exotic species status was assigned using the 2016 edition of the BC Flora Checklist (British Columbia Ministry of the Environment, 2016). Indicator "Taxonomic Distance" (TD) measured dissimilarity of species composition of the focal plots relative to the benchmark plots, as determined using non-metric multidimensional scaling (NMDS) of abundance-weighted species compositions; details of the calculation of this univariate indicator from the original multivariate species data are provided in Text S6 and Fig. S5. The remaining indicators were measured as described in BC Ministry of the Environment (2010).

#### 4.2.5. *Quality and quality hectares*

We calculated indicator quality and overall quality for sampling strata with at least 6 focal and 6 benchmark plots. Quality of "Percent Cover Exotic Species" was calculated using a fixed benchmark value of 0. All other indicators used the overlap or overlap-means quality algorithms (Table 2). The 95 % CLs of quality were calculated using 10,000 bootstrap iterations. QH and QH.net were calculated following the procedure described in Section 3.2. Vegetation types without at least 6 focal and 6 benchmark plots were excluded from quality calculations. For these vegetation types, we conservatively assumed a baseline quality of 1 (100 %) and therefore used their actual areas as QH in all subsequent calculations.

#### 4.2.6. *Power-sample size simulations*

We performed Monte Carlo simulations for each vegetation type, estimating the relationship between sample size ($n$; equal for focal and benchmark) and power to detect a significant change in overall quality at $\alpha = 0.05$ and an effect size (MDES) of 0.1 (Section 3.6.5). For each class of vegetation, the required minimum sample size, $n_{min}$, was the value of $n$ at which a logistic fit to a plot of $n$ and power for all simulations crossed the conventional power line of 0.80. We also performed power-sample size simulations for selected indicator-vegetation combinations. As some indicators required very large minimum sample sizes to achieve an MDES of 0.1, we also ran simulations at MDESs of 0.15 and 0.2.

#### 4.2.7. *Software*

All VQA calculations were implemented in the R programming language, version 3.5.2 (R Development Core Team, 2011). Univariate pdfs were fit to benchmark and focal samples by maximum likelihood estimation using function fitdistr of package MASS, version 7.3.51.1 (Ripley et al., 2013). NMDS (used for indicator Taxonomic Distance) was implemented using the metaMDS function of package vegan, version 2.5.4 (Dixon, 2003). Also required are tidyverse, version 1.3.1 (Wickham et al., 2019) and Hmisc, version 4.3.1 (Harrell, Frank E et al., 2020).

### 4.3. *Results and discussion*

#### 4.3.1. *Baseline assessment*

Baseline overall quality [lower, upper 95 % CLs] of the assessed vegetation types ranged from a low of 0.74 [0.64, 0.88] for Krummholz to a high of 0.87 [0.83–0.95] for Intermediate forest, Mature (Table 3; SM-1). The below-benchmark overall quality scores of forested vegetation, plus details of individual indicators (see below) were consistent with a history of logging in the early 1900s, prior to the initiation of the

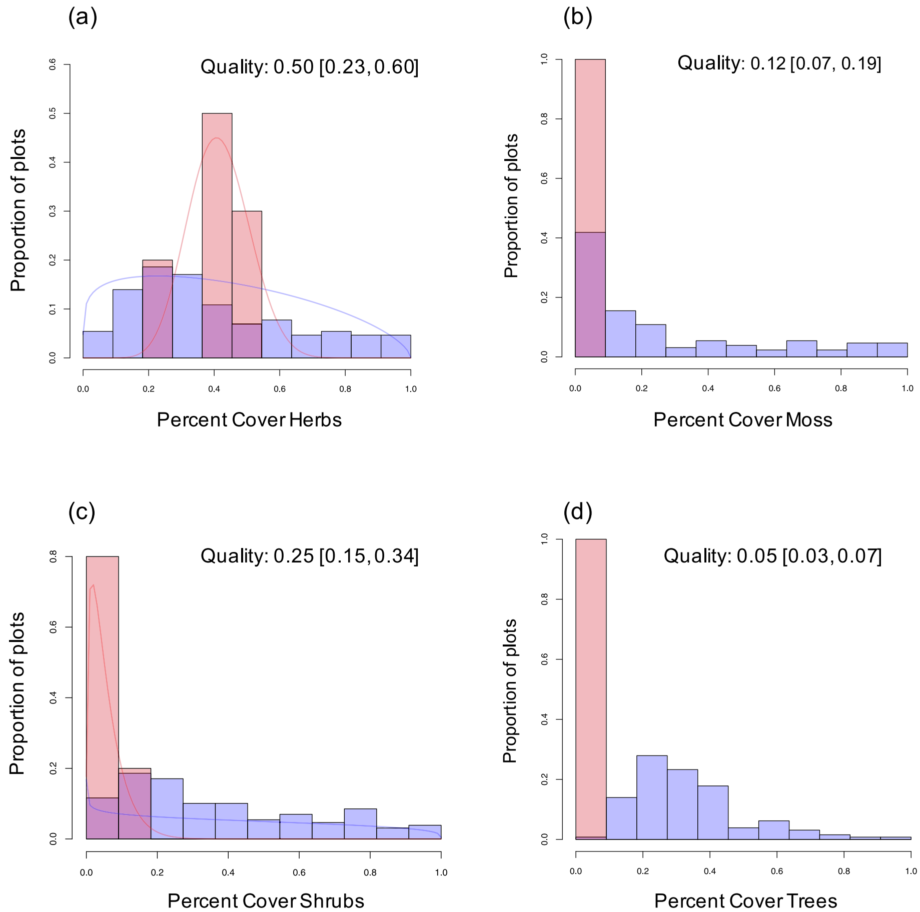


**Fig. 5.** Observed (histograms) and fitted (curves) focal and benchmark distributions and quality scores [lower, upper 95% CLs] of structural indicators for reclaimed Intermediate Forest, Early-mid. Pink: focal; blue: benchmark; purple: overlap.

mines. Distributions of individual indicators and their quality scores (Tables S1 & S2) support this interpretation. For early-mid seral stages of all three forest types, a disproportionate number of focal sites had little to no large tree cover and consequently low quality scores for indicator Percent Cover Trees (Fig. 3a for Intermediate Forest, Early-mid; results for Early-mid Dry Forest and Wet Forest similar). These anomalies were structural only; taxonomic composition of early-mid seral forests was almost identical to benchmark (e.g., Fig. 3b). In addition, strongly bimodal focal distributions of indicator Percent Cover Trees suggest that the Early-mid seral forest class in the Elk Valley may represent mixtures of naturally regenerating forest and recovering forest harvest. If true, the accuracy of the baseline assessment would be improved by splitting each stratum into two disturbance classes, one natural and one disturbed, sharing the same benchmark vegetation, and assessing each class independently with a separate set of focal plots. These observations highlight the value of examining individual indicator distributions and quality scores in addition to overall quality.

Non-forested vegetation classes also had overall quality scores < 1 (i.e., CIs not overlapping 1), suggesting that factors other than forest harvesting were responsible for the below-benchmark baseline vegetation quality. At lower elevations, quality of indicator “Percent Cover Exotic Species” for Brushland/Grassland (0.77 [0.66, 0.85]) was lower than for other forested and non-forested vegetation type (all of which had CIs close to or overlapping 1; Table S1). Although the majority of plots in Brushland/Grassland had zero cover of exotic species, a few had high exotic cover, with some sites exceeding 60 % (Fig. 3c). The prevalence of exotic species in Brushland/Grassland is also reflected in its somewhat anomalous species composition and low score for Taxonomic Distance (Fig. 3d). These observations are consistent with a general pattern of invasion of western North American grasslands and shrublands by exotic species (Dukes and Mooney, 2004).

#### 4.3.2. *Reclamation assessment*

Of the original 18,978.9 ha within the study area mine footprint, 2,109.0 ha had been re-vegetated at the time of this analysis to the four reclamation vegetation classes shown in Table 3. Overall quality of reclaimed vegetation ranged from 0.26 [0.19–0.32] for Wet forest, Early-mid to 0.42 [0.39–0.46] for Dry Forest, Early-mid, with no upper CLs exceeding 0.46.

Quality of the individual indicators paints a clearer picture of the ways in which each type of historical reclamation differed from undisturbed vegetation. Brushland/Grassland had fewer species, different taxonomic composition and much higher cover of non-native species, relative to the benchmark condition (Fig. 4, Table S3). These results are consistent with the widespread use of a small number of non-native agronomic species in historical grassland reclamation, which was guided largely by provincially-approved standards for ungulate forage production. Reclaimed forest differed strongly from naturally occurring early-mid seral forest in both structure (Fig. 5) and composition (Tables S3 & S4), with anomalously low cover of mosses, shrubs and trees (Fig. 5b-d), lower species richness and different taxonomic composition (Fig. S6).

#### 4.3.3. *Quality hectares and progress to NPI*

Comparison of current to baseline QH shows the contribution of historical reclamation to the overall goal of NPI (Table 4). Although impacts to most classes of vegetation will need to be compensated via offset actions elsewhere, on-site restoration has made major

**Table 4**
Baseline area and quality hectares (QH), current (reclaimed) area and QH, and net QH of vegetation within the combined Teck Elk Valley Operations footprint. Baseline and current QH are the product of the areas shown here and the quality scores in Table 3. Net QH is current QH - baseline QH. For vegetation for which quality could not be calculated due to low sample size (indicated by asterisk), baseline QH = actual area (ha). CLs: confidence limits.

| Vegetation (ecosystem group) | Baseline | | Current (reclaimed) | | Net QH [95 % CLs] |
|---|---|---|---|---|---|
| | Area (ha) | QH [95 % CLs] | Area (ha) | QH [95 % CLs] | |
| Alpine | 385.0 | 304.2 [277.2, 338.8] | 0 | 0 | −304.2 [-338.8, −277.2] |
| Alpine dwarf shrub | 0.6 | 0.5 [0.41, 0.51] | 0 | 0 | −0.5 [-0.5, −0.4] |
| Alpine grassland* | 67.1 | 67.1 | 0 | 0 | −67.1 |
| Alpine meadow* | 13.0 | 13.0 | 0 | 0 | −13 |
| Avalanche feature | 741.5 | 637.7 [600.6, 704.4] | 0 | 0 | −637.7 [-704.4, −600.6] |
| Brushland/ Grassland | 862.0 | 706.8 [672.4, 758.6] | 1257.7 | 440.2 [402.5, 503.1] | −266.6 [-269.9, −255.5] |
| Deciduous floodplain* | 157.3 | 157.3 | 0 | 0 | −157.3 |
| Dry forest, early-mid | 920.4 | 745.5 [699.5, 828.4] | 800.8 | 336.3 [312.3, 368.4] | −409.2 [-460, −387.2] |
| Dry forest, mature | 521.4 | 438.0 [411.9, 490.1] | 0 | 0 | −438 [-490.1, −411.9] |
| Dry forest, old* | 94.1 | 94.1 | 0 | 0 | −94.1 |
| Herb meadow* | 0.3 | 0.3 | 0 | 0 | −0.3 |
| Intermediate forest, early-mid | 7,771.00 | 6139.1 [5,906.0, 7,071.6] | 3.5 | 1.1 [1.0, 1.3] | −6,138 [-7,070.3, −5,905] |
| Intermediate forest, mature | 3,931.00 | 3420 [3,262.7, 3,734.5] | 0 | 0 | −3,420 [-3,734.5, −3,262.7] |
| Intermediate forest, old | 1,272.00 | 966.7 [877.7, 1,055.8] | 0 | 0 | −966.7 [-1,055.8, −877.7] |
| Krummholz | 55.6 | 41.1 [35.6, 48.9] | 0 | 0 | −41.1 [-48.9, −35.6] |
| Rock/Talus | 233.4 | 191.4 [179.7, 207.7] | 0 | 0 | −191.4 [-207.7, −179.7] |
| Shrubland* | 9.4 | 9.4 | 0 | 0 | −9.4 |
| Wet forest, early-mid | 923.0 | 692.3 [646.1, 766.1] | 47.0 | 12.2 [8.9, 15.0] | −680.1 [-751.1, −637.2] |
| Wet forest, mature | 565.0 | 457.7 [423.8, 508.5] | 0 | 0 | −457.7 [-508.5, −423.8] |
| Wet forest, old | 317.0 | 259.9 [234.6, 285.3] | 0 | 0 | −259.9 [-285.3, −234.6] |
| Wetland | 138.8 | 115.2 [108.3, 124.9] | 0 | 0 | −115.2 [-124.9, −108.3] |

contributions toward NPI for Brushland/Grassland and Early-mid Dry Forest; 62 % and 45 %, respectively, of the baseline QH of these two vegetation types had been restored by the time these data were collected. In addition to extending re-vegetation efforts to areas not currently undergoing reclamation, additional gains in QH could be achieved by improving the quality of existing and future reclamation. This is particularly true of Brushland/Grassland, where replacement of non-native grasses with native species has the potential to provide substantial improvements in quality for multiple indicators (Section 4.3.2). In the case of Early-mid Dry Forest, this vegetation is expected to transition over time to Mature Dry Forest, and restoration should therefore aim to restore substantially more than the number of QH originally present of the early-mid seral stage. Over-planting of the earliest seral stages will ensure a more balanced restoration of seral stages at and beyond closure. Teck has recently refocused the current reclamation program to target native species as well as aiming towards natural ecosystem assemblages for end land use objectives (Franklin and Burton, 2018). In this context, VQA provides a mechanism for integrating the contribution of native species and natural ecosystem assemblages into the goal of NPI.

Finally, it is important to emphasize that CLs, not means, determine when NPI has been attained. As illustrated in Fig. 1b, only when the lower 95 % CL of Net QH exceeds zero (QH.net.lcl > 0) are we justified in concluding that our estimate of QH.net also exceeds zero. It should also be apparent that excessively broad CIs can increase restoration obligations beyond what would be required given a more accurate estimate of QH.net. The incorporation of measurement error into the VQA workflow thus creates a built-in incentive for estimating quality, QH, and net QH with the greatest possible accuracy, subject to the opposing constraint of increased sampling and monitoring costs.

#### 4.3.4. *Sample size requirements*

Simulations of power versus sample size indicated that sample sizes ranging from 9 to 17 were required to detect a change in overall quality of 0.1 in the baseline vegetation (e.g., Fig. 6a). By contrast, much smaller sample sizes were needed to detect a similar effect size in the reclaimed vegetation (Fig. 6b). This difference reflects the narrower CIs of overall quality for the reclaimed vegetation (Table 3), which in turn are a consequence of the low variance of many reclaimed vegetation indicators. The low minimum sample sizes shown in Fig. 6b should not be taken literally; considerations of power and effect size aside; sample sizes of n = 3 are unlikely to be representative of an entire class of reclamation. Furthermore, as reclaimed vegetation matures and the variances of its indicators and overall quality converge upon values observed in the benchmark vegetation, *n*.min should increase to the values of *n*.min required for the baseline vegetation. In practice, smaller samples of plots can be installed initially for monitoring reclamation quality, and additional plots added incrementally as the vegetation matures.

Sample size requirements for individual indicators were generally higher than for overall quality. For example, samples sizes of 18, 27 and 42 were required to achieve effect sizes of 0.2, 0.15 and 0.1, respectively, for Species Richness of vegetation "Avalanche Feature", (Fig. S7). Given the importance of the indicator distributions and indicator quality for diagnosing and resolving causes of low overall quality scores, we recommend larger sample sizes than the minimum required for the measurement of overall quality. An even more compelling reason for targeting higher minimum sample sizes is to minimize the amount by which the estimated mean current QH must exceed baseline QH to ensure that the lower 95 % CL of Net QH is greater than zero (Fig. 1b).

## 5. Conclusions

The VQA approach provides an objective, repeatable and verifiable framework for measuring the extent and quality of pre-impact vegetation, monitoring the pace and effectiveness of restoration, and determining the point at which NPI has been achieved. By using distribution overlap as the main measure of quality, the method incorporates natural, landscape-scale variation not captured by other methods. The ability to quantify uncertainty increases confidence in monitoring results and creates built-in incentives for maximizing accuracy of quality estimates.

A potential shortcoming of the method is the need for relatively large sample sizes. This shortcoming is balanced in part by the relatively small number of indicators required, and the use of sampling methods familiar to most field crews experienced in vegetation surveys. In addition, the ability to use existing vegetation data can reduce the number of new

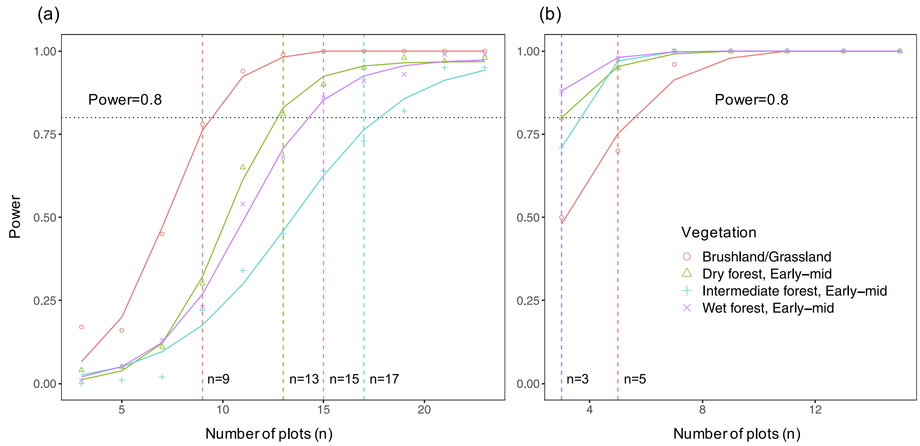


**Fig. 6.** Monte Carlo simulations of power vs. sample size for overall quality for (a) four type of baseline vegetation and (b) their corresponding classes of reclaimed vegetation. Each curve is a logistic fit to a series of power-sample size simulations for a single vegetation type. Vertical dotted lines indicate the minimum sample sizes required to detect of change in overall quality of 0.1 at α = 0.05 and a conventional power of 0.8.

plots needed. Over time, the growth of repositories of vegetation inventory data will lower the number of new benchmark plots that need to be established. Regulators should encourage companies to deposit samples of benchmark-quality vegetation in publicly available databases; companies themselves may voluntarily contribute such data in the interest of reducing future sampling burdens.

## CRediT authorship contribution statement

**Bradley L. Boyle:** Conceptualization, Methodology, Investigation, Data curation, Formal analysis, Visualization, Software, Writing – original draft, Writing – review & editing. **Warn Franklin:** Funding acquisition, Methodology, Validation, Writing – review & editing, Project administration. **Alison Burton:** Methodology, Validation, Writing – review & editing. **Raymond E. Gullison:** Conceptualization, Funding acquisition, Methodology, Validation, Writing – review & editing.

## Declaration of competing interest

The authors declare the following financial interests/personal relationships which may be considered as potential competing interests: Bradley L. Boyle reports financial support and article publishing charges were provided by Teck Resources Limited. Raymond E. Gullison reports financial support was provided by Teck Resources Limited. Warn Franklin reports financial support was provided by Teck Resources Limited. Allison Burton reports financial support was provided by Teck Resources Limited. If there are other authors, they declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

## Data availability

The authors do not have permission to share data.

## Acknowledgments

We thank Jared Hardner, Garry Luini, Katrina Lukianchuk, and Justin Straker and for suggestions that helped improve aspects of the VQA framework, and Christina Small for comments that strengthened the final manuscript. Dan Vasiga prepared the Elk Valley database exports of plots, vegetation and land cover used for the example application. Will Mackenzie and Deb MacKillop (BCENV) assisted with identification and selection of benchmark plot data from the BCENV provincial vegetation database. This work was supported in part by funding from Teck Coal Limited.

## Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ecolind.2023.111510.

## References

Berger, V.W., Zhou, Y., 2014. Kolmogorov–Smirnov Test: Overview, in: Wiley StatsRef: Statistics Reference Online. John Wiley & Sons, Ltd. https://doi.org/https://doi.org/10.1002/9781118445112.stat06558.

Best, D.J., Gipps, P.G., 1974. An improved gamma approximation to the negative binomial. Technometrics 16, 621–624.

British Columbia Forest Service – Research Branch, 2016a. BECMaster ecosystem plot database [WWW Document]. accessed 5.15.18. http://www.for.gov.bc.ca/hre/becweb/resources/information-requests/index.html.

British Columbia Forest Service – Research Branch, 2016b. Biogeoclimatic ecosystem classification, version 10 [WWW Document]. accessed 5.15.18. https://www.for.gov.bc.ca/hre/becweb/resources/maps/versionhistory.html#version10.

British Columbia Forest Service – Research Branch, 2018. VPro [WWW Document]. accessed 5.15.18. https://www.for.gov.bc.ca/hre/becweb/resources/software/vpro/index.html.

British Columbia Ministry of Forests, 2023. BEC WEB [WWW Document]. accessed 12.15.23. https://www.for.gov.bc.ca/hre/becweb/system/how/site.html.

British Columbia Ministry of the Environment, 2016. BC Flora checklist 2016 [WWW Document]. accessed 7.18.16. https://www.for.gov.bc.ca/hre/becweb/resources/codes-standards/standards-species.html.

British Columbia Ministry of the Environment, 2021. Terrestrial ecosystem data standards & guidelines [WWW Document]. accessed 12.15.23. https://www2.gov.bc.ca/gov/content/environment/plants-animals-ecosystems/ecosystems/tei-standards.

Ceballos, G., Ehrlich, P.R., Barnosky, A.D., García, A., Pringle, R.M., Palmer, T.M., 2015. Accelerated modern human – induced species losses: entering the sixth mass extinction. Sci. Adv. 1, 1–5. https://doi.org/10.1126/sciadv.1400253.

Cook, C.N., Wardell-Johnson, G., Keatley, M., Gowans, S.A., Gibson, M.S., Westbrooke, M.E., Marshall, D.J., 2010. Is what you see what you get? Visual vs. measured assessments of vegetation condition. J. Appl. Ecol. 47, 650–661. https://doi.org/10.1111/j.1365-2664.2010.01803.x.

Corporation, M., 2020. Microsoft Access [WWW Document]. accessed 12.20.23. https://www.microsoft.com/en-us/microsoft-365/access.

Corporation, M., 2022. SQL Server [WWW Document]. accessed 12.20.23. https://www.microsoft.com/en-us/sql-server.

Cousineau, D., 2017. Varieties of confidence intervals. Adv. Cogn. Psychol. 13, 140–155. https://doi.org/10.5709/acp-0214-z.

Cumming, G., 2014. The new statistics: why and how. Psychol. Sci. 25, 7–29. https://doi.org/10.1177/0956797613504966.

Curtin, C.G., 1995. Can montane landscapes recover from human disturbance? Long-term evidence from disturbed subalpine communities. Biol. Conserv. 74, 49–55. https://doi.org/10.1016/0006-3207(95)00014-U.

Davis, R., 2009. Simulation of fire dynamics and the range of natural variability of forest stand structure in the cranbrook and invermere timber supply areas, southeastern british columbia. Cranbrook, BC.

Demeo, T., Haugo, R., Ringo, C., Kertis, J., Acker, S., Simpson, M., Stern, M., 2018. Expanding our understanding of forest structural restoration needs in the pacific northwest. Northwest Sci. 92, 18–35. https://doi.org/10.3955/046.092.0104.

Dixon, P., 2003. VEGAN, a package of R functions for community ecology. J. Veg. Sci. 14, 927–930. https://doi.org/10.1111/j.1654-1103.2003.tb02228.x.
Dukes, J.S., Mooney, H.A., 2004. Disruption of ecosystem processes in western North America by invasive species. Rev. Chil. Hist. Nat. 77, 411–437.
Eddy, D., Hall, R., Rehwinkel, R., Baines, G., Dorrough, J., Austin, M., Kelly, A.L., Franks, A.J., Eyre, T.J., 2011. Assessing the assessors: Quantifying observer variation in vegetation and habitat assessment 12, 144–148.
Efron, B., 1979. Bootstrap Methods: Another Look at the Jackknife. Ann. Stat. 7, 1–26. https://doi.org/10.1214/aos/1176344552.
UN Environment Programme, 2020. Convention on Biological Diversity: Update of the zero draft of the post-2020 global biodiversity framework.
Eyre, T.J., Kelly, A., Neldner, V.J., Wilson, B.A., Ferguson, D.J., Laidlaw, M.J., Franks, A. J., 2011. BioCondition: A Condition Assessment Framework for Terrestrial Biodiversity in Queensland. Assessment Manual. Version 2.1, Department of Environment and Resource Management (DERM). Biodiversity and Ecosystem Sciences, Brisbane.
Franklin, C.W., Burton, A., 2018. End land use objective planning : integrating an ecosystem based approach into biodiversity and reclamation planning. In: Fort Williams, B.C. (Ed.), 41st Annual TRCR Mine Reclamation Symposium. Canada, pp. 1–11.
Fulé, P.Z., Covington, Moore, W.W., 1997. Determining reference conditions for ecosystem management of southwestern ponderosa pine forests. Ecol. Appl. 7, 895–908.
Gastwirth, J.L., 1975. Statistical measures of earnings differentials. Am. Stat. 29, 32–35. https://doi.org/10.1080/00031305.1975.10479109.
Gauthier, S., Boucher, D., Morissette, J., De Grandpré, L., 2010. Fifty-seven years of composition change in the eastern boreal forest of Canada. J. Veg. Sci. 21, 772–785. https://doi.org/10.1111/j.1654-1103.2010.01186.x.
Gentry, A.H., 1988. Changes in plant community diversity and floristic composition on environmental and geographical gradients. Ann. Missouri Bot. Gard. 75, 1–34.
Gentry, A.H., 1995. Diversity and floristic composition of neotropical dry forests. In: Bullock, S.H., Mooney, H.A., Medina, E. (Eds.), Seasonally Dry Tropical Forests. Cambridge University Press, Cambridge, pp. 146–194.
Gentry, A.H., 1993. Patterns of diversity and floristic composition in neotropical montane forests, in: Churchill, S.P., Balslev, H., Forero, E., Luteyn, J. (Eds.), Proceedings of the Neotropical Montane Forest Biodiversity and Conservation Symposium, The New York Botanical Garden, 21-26 June 1993. New York Botanical Garden, Bronx, N.Y., pp. 103–126.
Gibbons, P., Ayers, D., Seddon, J., Doyle, S., Briggs, S., 2008. Biometric 2.0: A Terrestrial Biodiversity Assessment Tool for the NSW Native Vegetation Assessment Tool - Operational Manual, NSW Department of Environment and Climate Change. Canberra.
Gibbons, P., Freudenberger, D., 2006. An overview of methods used to assess vegetation condition at the scale of the site. Ecol. Manag. Restor. 7, S10–S17. https://doi.org/10.1111/j.1442-8903.2006.00286.x.
Harrell, Frank E, J., Dupont, C., et al., 2020. Hmisc: Harrell Miscellaneous R package [WWW Document]. URL https://cran.r-project.org/package=Hmisc (accessed 6.1.22).
Holmes, P., Stuart-Smith, K., MacKillop, D., Lewis, D., Machmer, M., Franklin, W., MacDonald, R., McGuinness, K., Chow, E., van Rensen, C., Ayele, T., 2018. Old and mature forest cumulative effects assessment report. Kootenay-Boundary Region, Elk Valley.
Hurlbert, S.H., 1984. Pseudoreplication and the design of ecological experiments. Ecol. Monogr. 54, 187–211.
International Finance Corporation, 2012. IFC sustainability framework: policy and performance standards on environmental and social sustainability [WWW Document]. accessed 6.1.20. https://www.ifc.org/content/dam/ifc/doc/2010/2012-ifc-performance-standards-en.pdf.
Jennings, M.D., Faber-Langendoen, D., Loucks, O.L., Peet, R.K., Roberts, D., 2009. Standards for associations and alliances of the U.S. national vegetation classification. Ecol. Monogr. 79, 173–199. https://doi.org/10.1890/07-1804.1.
Kappelle, M., 1995. Changes in diversity along a successional gradient in a Costa Rican upper montane Quercus forest. Biodivers. Conserv. 4222, 10–34.
Kappelle, M., Kennis, P.A.F., de Vries, R.A.J., 1995. Changes in diversity along a successional gradient in a Costa Rican upper montane Quercus forest. Biodivers. Conserv. 4, 10–34. https://doi.org/10.1007/BF00115312.
Kappelle, M., Geuze, T., Leal, M.E., Cleef, A.M., 1996. Successional age and forest structure in a Costa Rican upper montane Quercus forest. J. Trop. Ecol. 12, 681–698.
Karr, J.R., 1981. Assessment of biotic integrity using fish communities. Fisheries 6, 37–41. https://doi.org/10.1577/1548-8446(1981)006.
Karr, J.R., Fausch, K.D., Angermeier, P.L., Yant, P.R., Schlosser, I.J., 1986. Assessing biological integrity in running waters: a method and its rationale. Champaign, Special Publication, Illinois Natural History Survey, p. 5.
Keane, R.E., Hessburg, P.F., Landres, P.B., Swanson, F.J., 2009. The use of historical range and variability (HRV) in landscape management. For. Ecol. Manage. 258, 1025–1037. https://doi.org/10.1016/j.foreco.2009.05.035.
Kier, G., Dinerstein, E., Ricketts, T.H., Küper, W., Kreft, H., Wilhelm Barthlott, J.M., 2005. Global patterns of plant diversity and floristic knowledge. J. Biogeogr. 32, 1107–1116.
Knopff, K., Franklin, W.E., 2018. Biodiversity management: establishing pre-existing baseline conditions on mature and historical mining disturbances to derive back-casted wildlife habitat suitability model metrics for ten wildlife species, in: 41st Annual TRCR Mine Reclamation Symposium. Fort Williams, BC, Canada.
Korb, J.E., Covington, W.W., Fulé, P.Z., 2003. Sampling Techniques Influence Understory Plant Trajectories After Restoration : An Example from Ponderosa Pine Restoration 11, 504–515.
Landres, P.B., Morgan, P., Swanson, F.J., Applications, S.E., Nov, N., 2014. Overview of the Use of Natural Variability Concepts in Managing Ecological Systems. Ecol. Appl. 9, 1179–1188.
Lewis, M., Christianson, A., Spinks, M., 2018. Return to flame: Reasons for burning in Lytton First Nation, British Columbia. J. for. 116, 143–150. https://doi.org/10.1093/jofore/fvx007.
Lombardi, F., Lasserre, B., Chirici, G., Tognetti, R., Marchetti, M., 2012. Deadwood occurrence and forest structure as indicators of old-growth forest conditions in mediterranean mountainous ecosystems. Ecoscience 19, 344–355. https://doi.org/10.2980/19-4-3506.
MacKenzie, W., 2012. Biogeoclimatic ecosystem classification of non-forested ecosystems in British Columbia (Tech. Rep. 068). Province of British Columbia, Victoria, B.C.
Magurran, A.E., 2004. Measuring biological diversity. Blackwell Publishing Ltd, Oxford, UK.
Maron, M., Hobbs, R.J., Moilanen, A., Matthews, J.W., Christie, K., Gardner, T. a., Keith, D. a., Lindenmayer, D.B., McAlpine, C. a., 2012. Faustian bargains? Restoration realities in the context of biodiversity offset policies. Biol. Conserv. 155, 141–148. https://doi.org/10.1016/j.biocon.2012.06.003.
McCarthy, M., Parris, K., 2004. The habitat hectares approach to vegetation assessment: an evaluation and suggestions for improvement. Ecol. Manag. Restor. 5, 24–27.
McElhinny, C., Gibbons, P., Brack, C., Bauhus, J., 2005. Forest and woodland stand structural complexity: its definition and measurement. For. Ecol. Manage. 218, 1–24. https://doi.org/10.1016/j.foreco.2005.08.034.
Meidinger, D. V, Pojar, J., 1991. Ecosystems of British Columbia. Special Report Series 6. British Columbia Ministry of Forests, Victoria, B.C.
Mucina, L., Bültmann, H., Dierßen, K., Theurillat, J.P., Raus, T., Čarni, A., Šumberová, K., Willner, W., Dengler, J., García, R.G., Chytrý, M., Hájek, M., Di Pietro, R., Iakushenko, D., Pallas, J., Bergmeier, E., Santos Guerra, A., Ermakov, N., Valachovič, M., Schaminée, J.H.J., Lysenko, T., Didukh, Y.P., Pignatti, S., Rodwell, J. S., Capelo, J., Weber, H.E., Solomeshch, A., Dimopoulos, P., Aguiar, C., Hennekens, S.M., Tichý, L., 2016. Vegetation of Europe: hierarchical floristic classification system of vascular plant, bryophyte, lichen, and algal communities. Appl. Veg. Sci. 19, 3–264. https://doi.org/10.1111/avsc.12257.
Olufadi, Y., Oshungade, I.O., Adewara, A.A., 2012. On allocation procedure using systematic sampling. J. Interdiscip. Mathermatics 15, 23–40.
Parkes, D., Newell, G., 2003. Assessing the quality of native vegetation: the "habitat hectares" approach. Ecol. Manag. Restor. 4, 29–38.
Peet, R.K., Lee, M.T., Jennings, M.D., Faber-Langendoen, D., 2012. VegBank: a permanent, open-access archive for vegetation plot data. Biodivers. Ecol. 4, 233–241.
Pimm, S.L., Joppa, L.N., 2015. How many plant species are there, where are they, and at what rate are they going extinct? Ann. Missouri Bot. Gard. 100, 170–176. https://doi.org/10.3417/2012018.
R Development Core Team, 2011. R: A Language and Environment for Statistical Computing.
Rainey, H.J., Pollard, E.H.B., Dutson, G., Ekstrom, J.M.M., Livingstone, S.R., Temple, H. J., Pilgrim, J.D., 2015. A review of corporate goals of No net loss and net positive impact on biodiversity. Oryx 49, 232–238. https://doi.org/10.1017/S0030605313001476.
Resources Inventory Committee – Ecosystems Working Group, 1998. Standard for terrestrial ecosystem mapping in British Columbia. Province of British Columbia, Resources Inventory Committee, Vancouver, BC.
Rio Tinto, 2008. Rio Tinto and biodiversity: achieving results on the ground [WWW Document]. URL www.riotinto.com/documents/ReportsPublications/RTBidoversitystrategyfinal.pdf (accessed 3.23.19).
Ripley, B., Venables, B., Bates, D.M., Hornik, K., Gebhardt, A., Firth, D., 2013. Package 'MASS' [WWW Document]. URL https://cran.r-project.org/web/packages/MASS/index.html (accessed 4.25.19).
Ross, K.A., Fox, B.J., Fox, M.D., 2002. Changes to plant species richness in forest fragments: fragment age, disturbance and fire history may be as important as area. J. Biogeogr. 29, 749–765. https://doi.org/10.1046/j.1365-2699.2002.00722.x.
Sahley, C.T., Vildoso, B., Casaretto, C., Taborga, P., Ledesma, K., Linares-Palomino, R., Mamani, G., Dallmeier, F., Alonso, A., 2017. Quantifying impact reduction due to avoidance, minimization and restoration for a natural gas pipeline in the Peruvian Andes. Environ. Impact Assess. Rev. 66, 53–65. https://doi.org/10.1016/j.eiar.2017.06.003.
Schlesinger, W., Andrews, J., 2000. Soil respiration and the global carbon cycle. Biogeochemistry 48 (1), 7–20.
Soto, D.P., Puettmann, K.J., 2020. Merging multiple equilibrium models and adaptive cycle theory in forest ecosystems: implications for managing succession. Curr. for. Reports 6, 282–293. https://doi.org/10.1007/s40725-020-00128-1.
Su-Yim, J., Kleinn, C., Ho-Kim, S., Jin-Hyun, J., Yong-Shin, M., 2009. A comparison of systematic sampling designs. J. Korean for. Soc. 98, 133–141.
Terrestrial Ecosystem Mapping Alternatives Task Force for the Resources Inventory Committee, 1999. Standards for Predictive Ecosystem Mapping: Inventory Standard (Version 1.0). British Columbia Ministry of Forests, Resources Inventory Branch, Victoria, B.C.
Terrestrial Ecosystem Mapping Alternatives Task Force for the Resources Inventory Committee, 2010. Field Manual for Describing Terrestrial Ecosystems (2nd Edition). British Columbia Ministry of Forests & British Columbia Ministry of Environment, Victoria, B.C.
The State of Queensland, 2014. BioCondition Benchmarks [WWW Document]. URL https://www.qld.gov.au/environment/plants-animals/biodiversity/benchmarks/.

Venables, W.N., Ripley, B.D., 2002. Modern Applied Statistics with S, 4th ed.,. Springer Science & Business Media.
Weidlich, E.W.A., Flórido, F.G., Sorrini, T.B., Brancalion, P.H.S., 2020. Controlling invasive plant species in ecological restoration: A global review. J. Appl. Ecol. 57, 1806–1817. https://doi.org/10.1111/1365-2664.13656.
Wickham, H., Averick, M., Bryan, J., Chang, W., McGowan, L., François, R., Grolemund, G., Hayes, A., Henry, L., Hester, J., Kuhn, M., Pedersen, T., Miller, E., Bache, S., Müller, K., Ooms, J., Robinson, D., Seidel, D., Spinu, V., Takahashi, K., Vaughan, D., Wilke, C., Woo, K., Yutani, H., 2019. Welcome to the Tidyverse. J. Open Source Softw. 4, 1686. https://doi.org/10.21105/joss.01686.