Review of the Lake Trophic Level Index

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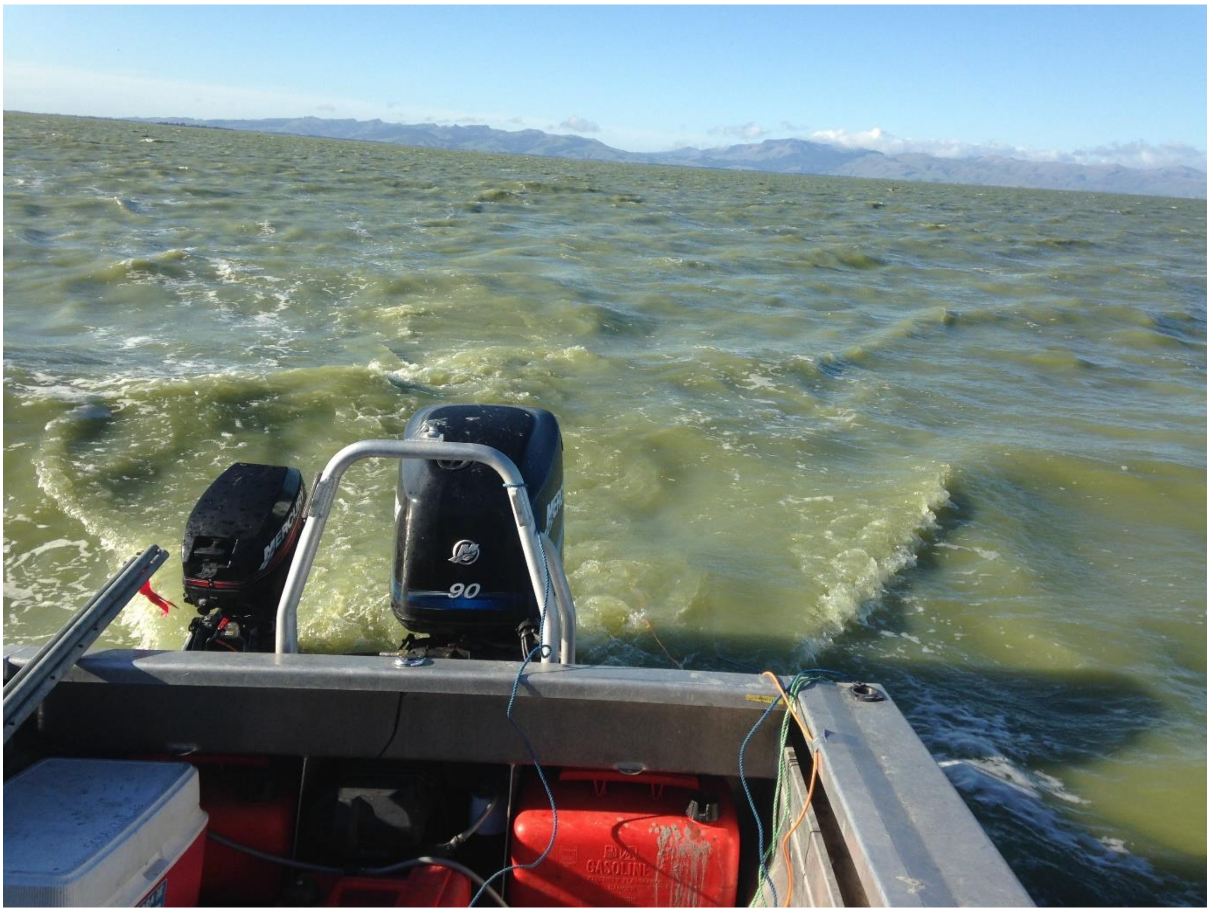


The supertrophic Lake Ellesmere/Te Waihora, Canterbury. Photo: Marc Schallenberg

Prepared for the Regional Council Lakes Focus Group

September 2020 (revised March 2021)

Executive summary

The concept of lake trophic state has been adopted by limnologists and lake managers for almost one hundred years. Trophic state is a normative concept that has some conceptual value, but is not strictly definable (Peters 1991). Quantitatively, it has been described in various ways, sometimes with reference to a single attribute, but most often in terms of multimetric indices.

The trophic level index (TLI; Burns et al. 2000) was developed specifically for New Zealand lakes; it derives to some extent from Carlson's trophic state index (Carlson 1977) and Chapra and Dobson's "Naumann index" (Chapra & Dobson 1981). As the phytoplankton communities of many New Zealand lakes had been demonstrated to be nitrogen-limited, neither of the two overseas trophic level indices were deemed appropriate for New Zealand lakes because they don't account for lake nitrogen concentrations.

In the TLI, chlorophyll *a* (Chl) was considered to be the primary trophic state variable. Data from New Zealand lakes were used to construct secondary trophic level indices for Secchi disk depth, total nitrogen and total phosphorus concentrations by calibrating them against the TLI(Chl). The overall TLI is calculated using the average (i.e., equal weighting) of the four separate TLI sub-indices. The TLI is scaled on a log(10) scale such that each TLI integer unit describes a separate trophic state category, producing a framework with 7 trophic state classes from microtrophic to supertrophic.

The Burns et al. (1999; 2000) TLI has been the main tool for the assessment of lake water quality and eutrophication in New Zealand for the past 20 years. It is used by regional councils and the Ministry for the Environment to monitor and report on the trophic state and trends of over one hundred lakes. In addition, TLI targets have been set in regional plans for many of the monitored lakes (e.g., Burns et al. 2009), indicating that TLI reflects an important value that many New Zealanders perceive in lakes. Furthermore, TLI has also been used by researchers and modellers to hindcast trophic states and to simulate the outcomes of various climate change and land use scenarios on future lake trophic states (e.g., Trolle et al. 2011; Abell et al. 2020).

Since the TLI's introduction and uptake as a lake monitoring framework by regional councils, it has become apparent that different variations of the original TLI protocol have been applied in sampling lakes and in calculating the TLI. The effects of the inconsistencies between different methodologies have not been assessed to date.

This report addresses the following questions regarding the TLI:

- How does TLI relate to lake ecological health and integrity?
- What is the management utility of TLI?
- What are the effects of inconsistencies in the way TLI is implemented?
- Should the TLI be updated?

The key findings of this report are summarised below:

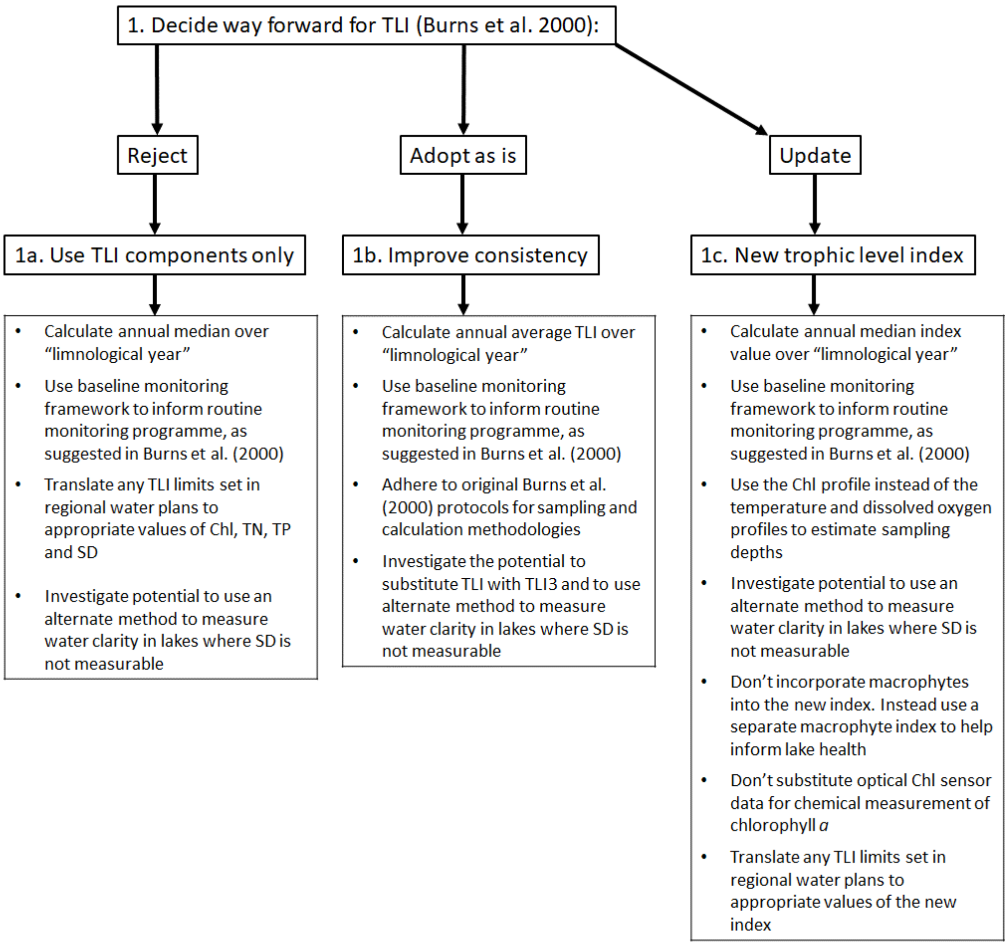
1. While the TLI is not an attribute in the current National Policy Statement for Freshwater Management, three of its components are included (Chl, total phosphorus and total nitrogen). The main reason that water clarity was not included is that a variety of natural substances in the water of some lakes precludes the development of a national-scale water clarity attribute linked to anthropogenic effects.
2. Anthropogenic eutrophication (as indicated by TLI or its components), is a key indicator of the pristineness component of lake ecological integrity, as defined by Schallenberg et al. (2011). Reference conditions for TLI and its components have been derived for New Zealand lakes,

allowing for estimates of departures from reference conditions to be made for TLI (Abell et al. 2020) and its components (Schallenberg 2019).

3. TLI is conceptually useful because it reflects a normative value related to lake health which integrates information on the current trophic state and the potential for further eutrophication.
4. The management utility of the TLI depends on whether the management goals are values-oriented or ecosystem-oriented. It has more value as a tool for monitoring perceived lake health state and trends and for prioritising investment in restoration than it does in providing a nuanced understanding of ecosystem functioning. It has limited functionality for limit-setting.
5. While taking the annual average of the monthly (logged) TLI data may have statistical advantages (especially if the data are log-normal distributed), the Burns et al. (2000) protocol was developed and calibrated by calculating the TLI on the annual average of the un-logged monthly data. Thus, regional councils should adhere to the Burns et al. (2000) protocol so that TLI values are properly calibrated and are comparable.
6. To address spatial variation in TLI, Burns et al. (2000) recommended sampling multiple sites per lake as part of a "baseline monitoring" programme. After analysis of the correlation among sites, the number of sites sampled may be reduced (as part of the "routine monitoring" programme). Large lakes (i.e. > 100 km^2 and deeper than 100m) have substantial spatial variability and require individually-designed monitoring programmes to satisfactorily account for spatial variability in TLI and its components.
7. While the protocol recommended in Burns et al. (2000) for determining sampling depths may be useful for sampling the phytoplankton habitat in most lakes, it will underestimate the trophic state of lakes which have deep chlorophyll maxima (i.e., lakes with phytoplankton layers in, or below, the thermocline).
8. Different time periods are used for the calculation of the annual TLI assessment (e.g., January 1 - December 31, July 1 - June 30, September 1 - August 31). This should be standardised to avoid apparently different TLI assessment values being reported for the same lake in the same year. As the Burns et al. (2000) protocol recommended, for most lakes, the "limnological year", running from September 1 to August 31, defines the most appropriate time period within which to calculate TLI .
9. While it is essential that the TLI be used in a consistent manner, there have been inconsistencies in its use. Thus, monitoring data for many lakes have been excluded from national assessments of water quality. Whenever possible, the Burns et al. (2000) protocol is quite clear and should be adhered to in assessing the TLI; however, the recommended method of taking the log of the annual average concentrations of TLI components is not the most statistically robust approach to calculating the TLI.
10. The TLI should not be calibrated differently for different lake types. It is more sensible to set different TLI limits/guideline values for different lake types. This way, the TLI and the definition of trophic state have consistent meanings for all lakes of all types.
11. While many factors can directly and indirectly affect the trophic state of lakes, the calculation of the TLI should not be adjusted for these factors. Rather, these factors could be accounted for in setting limits/guideline values for lakes which are affected by the co-variates. No accounting for sediment resuspension was originally recommended, nor is required, for the calculation of the TLI because strong benthic-pelagic coupling is a normal feature of many shallow lakes.
12. The inclusion of submerged macrophytes within the concept of trophic state, as has been suggested by some overseas researchers, has not received widespread acceptance and is not a feature of the TLI. However, lake submerged macrophyte indicators can be useful, complementary indicators of lake health, as is acknowledged by the inclusion of lake macrophyte indices in the NPS-FM (2020). In Waikato lakes, the two indices are only weakly correlated, indicating there is little redundant information between them.

13. New sensor technologies are now available which can provide much more detailed information on phytoplankton and water clarity than what was available at the time the TLI was developed. Chl sensor profiling could improve the determination of sampling depths for TLI calculation, but the optical Chla sensors should not replace the solvent-extract Chl method recommended for the TLI by Burns et al. (2000).
14. The Burns et al. (2000) TLI protocol ignores deep chlorophyll maxima, which occur in some lakes. If the TLI were to be updated, to remedy this problem the determination of sampling depths should be made from Chl profiles rather than from profiles of temperature and dissolved oxygen concentrations, as are recommended in Burns et al. (2000).
15. As a component of TLI, Secchi depth can be problematic to measure in some lakes. If the TLI were to be updated, an alternative method of measuring visual clarity should be investigated for TLI calculations.
16. TLI3 (TLI calculated without the inclusion of Secchi depth information) is sometimes used when researching or reporting lake trophic state. Before the substitution of TLI3 for TLI can be recommended, further statistical work should be done to determine the potential errors and biases that this substitution could result in.

This critical review of the TLI highlights strengths and weaknesses of the TLI, as it is used today. A number of recommendations are outlined for three possible ways forward regarding the future of the TLI: 1. Ceasing use of the TLI in favour of measuring its components separately, 2. Continuing to use the TLI, but with improved consistency and adherence to the original Burns et al. (2000) protocol, and 3. Updating the TLI to create a new, improved trophic state index.



This report aims to stimulate thought and discussion regarding the TLI and hopefully assist in improving policy and practice regarding the management of eutrophication in our lakes

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"People know what they do; frequently people know why they do what they do; but what they don't know is what they do does." – Michel Foucault[1]

1. Background

1.1. Trophic state and trophic state indices

The popular usage of normative concepts such as lake health and ecological integrity (Schallenberg et al. 2011) suggest that such concepts are of value. The scientific expression of such normative concepts is usually in the form of multivariate metrics which place lakes along a gradient of environmental quality or desirability. From a research/academic perspective, such concepts may have limited value, but they appear to serve important functions in the realm of environmental management and restoration (Schallenberg et al. 2011). Normative concepts such as trophic state may represent emergent phenomena (or epiphenomena) of complex ecosystems, that are widely perceived by humans. While the epistemological basis for such concepts can be debated, the fact remains that the trophic state of lakes has been in use in New Zealand limnology for over half a century, and therefore carries some "weight" (e.g., Jolly1959; McColl 1972; Spenser 1978; White 1983; Vant 1987a; Burns et al. 2000; Abell et al. 2020).

The terms eutrophic and oligotrophic were first used by Auguste Thienemann and Einar Naumann in the early part of the twentieth century. Naumann defined the trophic state as indicative of the quantitative production of phytoplankton as well as the nutrient differences among lakes (Hutchinson 1967; Carlson 2009). At the time, there was no useful quantitative information on nutrient concentrations in waters, but in 1923 Naumann reported that the fertilisation of lake water resulted in algal proliferations (Naumann (1923), as cited in Hutchinson (1967)).

The concept of trophic state conceptually relates to the idea that lakes have a generalised "ontogeny" or "succession", which results in natural changes in lake productivity over time (Vant 1987a). This concept states that lakes progress over time (i.e., millennia or longer) from a state of low biological productivity soon after lake formation (e.g., post-glaciation) to a state of higher productivity as more fertile soils develop in the catchments increasingly contributing nutrients and organic matter to lakes via hydrological flows from land to water. In the absence of geological catastrophes which could reset this succession, lakes are thought to undergo this type of slow, natural eutrophication process. This process may be naturally facilitated by microbial processes such as nitrogen fixation as well as by increasing ecological complexity, which can increase lake fertility and the cycling rates of nutrients in catchments. Human modifications to catchments and lakes, such as agricultural intensification, can be seen as artificially accelerating the process of nutrient transfer from land to water (Leopold 1949), thus accelerating the slow, natural process of eutrophication of aquatic ecosystems.

In the absence of chemical data, Thienemann and Naumann distinguished oligotrophic from eutrophic lakes based on qualitative criteria such as whether the dominant genus of chironomid larvae in a lake was *Tanytarsus* or *Chironomus* (Moss 2018). In the 1960's and 70's, quantitative indices to classify the trophic status of lakes were developed. These indices were either based on single or multiple biotic and/or abiotic variables.

Substantially different definitions of trophic state have been proposed, including: (1) an estimate of "the quantitative production of phytoplankton" (Naumann as referenced in Carlson 2007), (2) an estimate of "the potential food base of an ecosystem" (Dodds 2006), (3) the "degree of nutrient enrichment" of an

[1] Foucault, M (1964) Madness and civilisation: A history of insanity in the age of reason. Union Générale d'Éditions. Paris.

ecosystem (Taylor et al. 2007) and (4) “the life-supporting capacity per unit volume of a lake” (Burns et al. 2000).

Assessments of lake trophic state based on single variables included phosphorus retention (Miller, 1995), chlorophyll-a concentration (Hakanson & Boulion 2001), and organic matter supply (Baban 1996); there is no consensus on which single variable best expresses trophic state. On the other hand, where data have been available, multimeric indices have also been developed. For example, Shannon and Brezonik’s (1972) trophic state index included chlorophyll-a (Chl), Secchi disk depth (SD), total nitrogen (TN), total phosphorus (TP), primary productivity, conductivity and the divalent:monovalent cation ratio. The index was used for 10 years in Florida lakes.

Perhaps the most well-known trophic state index (TSI) is Carlson’s Trophic State Index (Carlson 1977; Sharma et al. 2010), which comprises three single variable assessments: the TSI(SD), TSI(Chl) and TSI(TP). Carlson constructed the TSI(SD) in such a way that a halving of SD corresponds to an increase of 10 points in the index (calibrated so that most lakes score between 0 and 100). Using data from Minnesota lakes, Carlson performed log-log linear regressions of SD against Chl and TP. Based on these regressions he created TSI(Chl) and TSI(TP) (Carlson, 1977). Sometimes the three indices are averaged, for example in Melcher (2013) and Sharma et al. (2010). However, for conceptual and mathematical reasons this has been discouraged (Osgood 1984). Carlson recommends primarily using TSI(Chl), and calculating TSI(SD) and TSI(TP) only as a substitute for, or in addition to, TSI(Chl) (Carlson, 2007).

Several adaptations of Carlson’s TSI were developed, such as the Kratzer-Brezonik TSI for Florida lakes (Kratzer & Brezonik 1981). Whereas Carlson’s index only applies to phosphorus-limited lakes, the Kratzer-Brezonik TSI can also be used for nitrogen-limited and nutrient-balanced lakes. Furthermore, the Kratzer-Brezonik TSI was based on Chl instead of SD and Florida lake data were used for regression against SD, TP and TN (Brezonik, 1984). Also Chapra and Dobson (1981) created a TSI called the “Naumann index” for the Laurentian Great Lakes. Like Carlson’s TSI, the Naumann index is based on SD and incorporates Chl and TP (Chapra & Dobson 1981).

Classifications of lake trophic state based on oxygen concentration and oxygen depletion have existed since the 1930s (e.g., Hutchinson 1938). More recently, Chapra and Dobson developed their “Thienemann index” for the Laurentian Great Lakes as a complement to their Naumann index; the former is calculated from the volumetric oxygen depletion rate, the duration of the stratified period and the oxygen concentration of the hypolimnion at the onset of stratification (Chapra & Dobson 1981). A TSI based on oxygen saturation (Jones & Barnes 2005) was developed for Indiana lakes and Walker (1979) modified Carlson’s TSI for Connecticut lakes to include the rate of hypolimnetic volumetric oxygen depletion (HVOD). As yet another extension to Carlson’s TSI, Dunalska (2011) developed a TSI for lakes based on the total organic carbon concentration of the lake water.

Carlson’s TSI only takes into account conditions in the pelagic zone of lakes. Canfield and Jones (1984) developed a TSI that accounts for nutrients present in submerged macrophytes.

In 1982, the OECD developed global trophic state criteria by undertaking a survey in which one hundred limnologists and water quality experts from around the world were asked to classify lakes using any criteria they thought appropriate. Based on the results of this survey, OECD (1982) developed a probability distribution for how lakes with specific Chl concentrations would be classified according to the experts. The OECD criteria were developed based on this probability distribution, together with relationships between Chl concentrations and SD and TP. The reader is referred to Lee et al. (1995) and Nürnberg (1996), who reviewed diverse criteria for the trophic classification of lakes.

A number of attempts have been made to produce phytoplankton community indices for trophic state. In 1931, Naumann identified phytoplankton indicator taxa representative of eutrophic and oligotrophic lakes and this indicator work was extended by Nygaard in 1949 and 1955, and then by Järnefelt in 1956 and Hutchinson in 1967 (Hutchinson 1967). More recently, Marchetto et al. (2009) identified two types of phytoplankton indices, which are used by European countries. The first type is based on the abundance of specific taxa and the trophic state scores for these taxa. The scores were originally determined based on correlations between taxa and TP. The second type is based either on ratios between biovolumes of different algal groups, or on the percent of the total algal biovolume made up by indicator algal groups. According to Marchetto et al. (2009), phytoplankton indices should be used with caution because of difficulties in identifying taxa to species level; therefore, such phytoplankton taxonomic indicators may lack sensitivity.

It has been suggested that zooplankton are useful in assessing the trophic state of waterbodies (Haberman & Haldna 2014; Duggan et al. 2001; Duggan 2008). Based on studies of Estonian lakes Haberman & Haldna (2014) identified several options for constructing a zooplankton trophic index. These include the presence of indicator species for eutrophic and oligotrophic waters and ratios of specific zooplankton groups. Furthermore, variables including the grazing rate of herbivorous zooplankton, the ratio of zooplankton:phytoplankton biomass, and the ratio of planktonic filterers:primary production were also proposed to reflect the trophic state of lakes (Haberman & Haldna 2014). The rotifer trophic state index developed from 33 lakes from the North Island of New Zealand is based on a multivariate weighted averaging algorithm that relates relative abundances of rotifer taxa to trophic state, attributing trophic state indicator scores to various rotifer taxa (Duggan et al. 2001).

The new NPS-FM requires regional councils to report on “ecosystem health”, which is defined as a multimetric incorporating water quality, water quantity, habitat, aquatic life and ecological processes (NPS-FM 2020). A water body is seen as healthy if these five components are “suitable to sustain the indigenous aquatic life expected in the absence of human disturbance or alteration.” Trophic state indices are not mentioned in New Zealand’s National Policy Statement for Freshwater Management (NPS-FM) 2020; however, the lake attributes Chl, TN and TP are referred to as “trophic state” attributes (NPS-FM 2020). Thus, when the NPS-FM attributes are eventually fully implemented in regional plans, regional councils will be mandated to measure variables related to lake ecosystem health and trophic state, although they will not be required to report on a trophic state index.

Similarly, the European Union (EU) does not mandate the use of trophic state indices. The EU requires member states to evaluate “ecological status” based on assessments of “biological quality elements” (BQEs): phytoplankton, aquatic flora, benthic invertebrates and fish. For each BQE, the ecological status is classified as “high”, “good”, “moderate”, “poor” or “bad”, in consideration of observed values in relation to a “reference” value. The member states can choose their own methods to do this assessment and at least 297 methods are used (Birk et al. 2012). Although not specifically aiming to assess “trophic state”, many methods are focused on the assessment of eutrophication. In some of the method descriptions (e.g., Barbe et al. 2003; STOWA 2018) the terms “oligotrophic”, “mesotrophic”, “eutrophic” and “hypereutrophic” are used, although the reporting of trophic state (using specific trophic state indices) is not mandated.

In summary, the concept of lake trophic state has been used by limnologists and lake managers for almost one hundred years. It is a normative concept that limnologists have quantitatively formulated in various ways, sometimes in terms of a single attribute, but most often in terms of multimetric indices. It appears that trophic state has some conceptual value but is not satisfactorily definable (Peters 1991). Trophic state is a concept, like lake health and ecological integrity, which has inherent value and which can impart information regarding the human value of lakes; however, it is not easily defined or generalised, except within an agreed set of human perspectives about the value of lake ecosystems. Peters stated that, "Thoughtful limnologists try to avoid the [trophic state] terms in precise discourse, using instead quantitative measurements..." (Peters 1991).

Above, several options for quantifying the trophic state of lakes are described. The NPS-FM (NPS-FM 2020) and the EU policy frameworks show that while monitoring and reporting on lake trophic state is not explicitly mandated, some components of a general concept of trophic state are nevertheless included in their policy frameworks. Thus, despite the confusing plethora of definitions, formulations and thresholds that have been applied to trophic state classification (Peters 1991), the concept of trophic state continues to indirectly influence the setting of water quality standards for lakes and assessment of the effectiveness of lake management.

1.2. The New Zealand trophic level index (TLI)

From a review of the scientific literature, it appears that most trophic state indices divide the gradient of trophic states into quantifiable classes based on variables related to open water transparency, algal biomass and nutrient levels. New Zealand limnologists have used a variety of trophic state classification frameworks. For example, McColl (1972) used the variables dissolved oxygen, water transparency, the summer alkalinity differential between surface and bottom waters, chlorophyll-a, phosphorus, nitrogen, iron and manganese concentrations to classify the trophic status of seven North Island lakes. Spencer (1978) explored the potential to use heterotrophic glucose uptake rates, heterotrophic bacteria direct counts and adenosine triphosphate measurements to classify 21 Canterbury high country lakes into trophic states. White (1983) carried out a comparative analysis of the trophic status of 27 New Zealand lakes as compared to the mean trophic states of 150 lakes in OECD countries (OECD 1982), as determined by TN, TP, Chl and SD.

Burns et al. (1999; 2000) developed a trophic state index specifically for New Zealand lakes, based on Carlson's TSI (Carlson 1977) and Chapra and Dobson's "Naumann index" (Chapra & Dobson 1981). Burns et al. (1999; 2000) developed their lake trophic level index (TLI) because Carlson's index was deemed to be too coarse in its higher trophic levels, while Chapra and Dobson's was too fine, having five separate levels within the mesotrophic range. Furthermore, the phytoplankton communities of many New Zealand lakes had been demonstrated to be nitrogen-limited and neither of the two overseas trophic level indices takes nitrogen concentrations into account.

Burns et al. (1999; 2000) initially considered using Bill Vant's trophic state classification (Davies-Colley et al. 1993; Table 5.5.), but this classification had only four trophic categories and it was felt that more than one trophic category was required for the range of chlorophyll-a concentrations spanning between 5 and 30 mg m^{-3}. However, Vant's (1987a) trophic state, which comprised the attributes Chl, SD, TP, TN, HVOD (hypolimnetic volumetric oxygen depletion rate) and phytoplankton species and biomass, was adopted as a working definition for a New Zealand TLI. However, Burns et al. (1999; 2000) eventually decided not to include phytoplankton community structure attributes in their formulation of the TLI due to insufficient data available on the phytoplankton communities in their study lakes.

As opposed to Carlson's TSI, which was designed around SD, the TLI was constructed with Chl as the primary trophic state variable. Data from 17 New Zealand lakes (sampled over multiple years using the same sampling and analytical protocols) was then used to construct the secondary trophic level indices TLI(SD), TLI(TN) and TLI(TP), which were calibrated to TLI(Chl). The overall TLI is calculated using an average (i.e., equal weighting) of the four separate TLI sub-indices. The TLI sub-indices are transformed on a log(10) scale such that each log(10) unit defines a trophic state category, producing a framework with 7 trophic state classes, from microtrophic to supertrophic. Burns et al. (1999; 2000) also developed a percent annual change (PAC) metric which, assesses temporal changes in trophic state using TLI data along with HVOD (Burns et al., 1999).

While Burns et al. (2000) defined trophic state rather vaguely as “the life-supporting capacity per unit volume of a lake”, their TLI has become the main tool for assessing lake water quality in New Zealand for the past 20 years. It is used by regional councils and the Ministry for the Environment to monitor and report on the trophic state and trends in over one hundred lakes. In addition, TLI targets for many of the monitored lakes have been set into regional plans (e.g., Burns et al. 2009), indicating that the TLI reflects important lake values for many New Zealanders. Furthermore, the pristine, reference state of TLI has been modelled by researchers (e.g., Abell et al. 2020) and future changes in TLI have been simulated in relation to various climate change and land use scenarios (e.g., Trolle et al. 2011).

2. Scope of this TLI review

Since the TLI was developed over 20 years ago, its use as an index of lake water quality by regional councils and by the Ministry for the Environment has become ubiquitous. However, variations of the original protocol have been applied in the sampling for, and calculation of, the TLI. The effects of these inconsistencies have not been commented on or assessed, formally.

Along with providing the above discussion regarding the conceptual development and historical application of the concept of trophic state, this report also addresses the following questions:

- How does TLI relate to lake ecological health and integrity?
- What is the management utility of TLI?
- What are the effects of inconsistencies in the way TLI is implemented?
- Should the TLI be updated?

The report makes a number of recommendations and concludes by exploring three potential avenues for the future use of the TLI in New Zealand.

3. How does TLI relate to lake ecological health and integrity?

In this section, we discuss how the lake trophic state index, the TLI, relates to other normative concepts and indices, which are sometimes used in lake management.

3.1. The TLI and ecosystem health as defined in the NPS-FM (2020)

Although it does not list TLI as an attribute, the national policy statement for freshwater management (NPS-FM 2020) includes “ecosystem health” as one of its four compulsory freshwater values. Ecosystem health is defined as a normative concept that integrates water quality, water quantity, habitat, aquatic life and ecological processes (NPS-FM 2020):

Appendix 1A – Compulsory values

1 Ecosystem health

This refers to the extent to which an FMU or part of an FMU supports an ecosystem appropriate to the type of water body (for example, river, lake, wetland, or aquifer).

There are 5 biophysical components that contribute to freshwater ecosystem health, and it is necessary that all of them are managed. They are:

Water quality – the physical and chemical measures of the water, such as temperature, dissolved oxygen, pH, suspended sediment, nutrients and toxicants

Water quantity – the extent and variability in the level or flow of water

Habitat – the physical form, structure, and extent of the water body, its bed, banks and margins; its riparian vegetation; and its connections to the floodplain and to groundwater

Aquatic life – the abundance and diversity of biota including microbes, invertebrates, plants, fish and birds

Ecological processes – the interactions among biota and their physical and chemical environment such as primary production, decomposition, nutrient cycling and trophic connectivity.

In a healthy freshwater ecosystem, all 5 biophysical components are suitable to sustain the indigenous aquatic life expected in the absence of human disturbance or alteration (before providing for other values).

While the aggregation of information into an ecosystem health index may be desirable for simplifying the communication of lake health status to the public, the disadvantage of excessive aggregation is that nuanced insights into ecological condition, function and drivers/stressor can be lost in the aggregation of multiple attributes (Verburg et al. 2010). For example, substantial changes in components of lake health may occur over time, but compensatory changes by other components could result in no change in overall lake health, which may obscure important information about lake development over time. The same criticism could be made of the TLI index. Instead of including it, the NPS-FM lists three of the TLI components as attributes, encouraging the reporting of the attributes separately, instead of aggregating them into the TLI index.

The lake scientist advisory groups that assisted with the development of the NPS-FM attributes were not able to recommend a national-scale attribute for water clarity (i.e., Secchi disk depth). This was due to the influence of light-altering substances such as humic acids and suspensoids (e.g., chemical precipitates from geothermal inputs, glacial flour, resuspended lake bed sediment) on water clarity in many New Zealand lakes (Howard-Williams & Vincent 1985). These substances which do not result from eutrophication can substantially reduce water clarity in lakes. Therefore, a national-scale water clarity attribute could not easily be recommended and, as a result, the water clarity component of the TLI is absent from the NPS-FM attributes.

Table 1. NPS-FM (2020) lake attributes, highlighting those referred to as “trophic state” attributes. Also shown are the associated ecosystem health components, the sub-attributes, how the attributes are guided by a lake classification, the applicable measurement time scale and key attribute statistic, and whether the attributes mandate either limit setting or action plans from regional councils.

Attribute	Health component	Sub- attributes	Classification	Time scale and statistic	“Trophic state”	“Limit setting”	“Action plan”
Phytoplankton biomass	**Aquatic life**			**Annual median and maximum**	**Yes**	**Yes**	
Total nitrogen	**Water quality**		**i) seasonally stratified; ii) polymictic or brackish**	**Annual median**	**Yes**	**Yes**	
Total phosphorus	**Water quality**			**Annual median**	**Yes**	**Yes**	
Ammonia toxicity	**Water quality**			**Annual median and maximum**		**Yes**	
Submerged plants	**Aquatic life**	**i) invasive impact; ii) native condition**		**At least once per 3-year period**			**Yes**
Dissolved oxygen	**Water quality**	**i) bottom water; ii) mid-hypolimnion**	**Mid-hypolimnion for seasonally stratifying lakes only**	**Annual minimum**			**Yes**

Key point: *While the TLI is not an attribute in the NPS-FM, three of its components are included (Chl, TP and TN). The main reason that water clarity was not also included is that a variety of natural (non-anthropogenic) light-altering substances in the water of some lakes precluded the development of a national-scale water clarity attribute that could be related to anthropogenic eutrophication.*

3.2. The TLI and lake health as indicated by LakeSPI (Clayton & Edwards 2006)

In many countries, lake macrophyte communities are assessed to determine ecological condition, “lake health” and the conservation value of lakes (Schallenberg & Schallenberg 2018). Submerged macrophyte communities can play an important role in lake functioning (Pokorný & Kvêt 2004), especially in shallow lakes (Scheffer 2004), where they support lake productivity, biodiversity, and where they stabilise the lake bed against wind-induced sediment resuspension. Thus, macrophytes generally provide important ecosystem services to lakes (Schallenberg et al. 2013).

LakeSPI (submerged plant indicators), is a macrophyte-based lake health monitoring protocol and bioindicator, developed specifically for New Zealand lakes (Clayton and Edwards 2006). It is based on assessments of lake macrophyte communities and coverage. It assesses the departure of lake submerged vegetation from a pristine state, based on a range of vegetation features common to New Zealand lakes (McDonald et al. 2013).

In a LakeSPI assessment, three indices are evaluated: (1) the condition of native submerged plants (native condition index), (2) the impact of invasive submerged plants on the lake (invasive impact index) and (3) the overall LakeSPI index (Clayton & Edwards 2006). These are normative indices, whereby higher native condition and LakeSPI indices reflect a healthier lake, while a higher invasive impact index is representative of a degraded lake, reflecting poorer ecological health. Indices 1 and 2 are combined to give the overall LakeSPI index number (Figure 1), which is calibrated for each lake and scaled to a percentage whereby a LakeSPI of 100 corresponds to a pristine lake condition (Clayton

& Edwards 2006). The quantification of the departure from a pre-human reference condition and the scaling of the index scores to allow across-lake comparisons are useful for comparatively assessing and reporting lake macrophyte community health. The maximum potential LakeSPI score is also adjusted for lake depth (Clayton & Edwards 2006). Other information can also be used to assist, adjust or calibrate LakeSPI assessment, including: (1) a correction for naturally turbid waters (Clayton & Edwards 2006), (2) the use of historical information on the macrophyte community of a lake prior to substantial human impacts, when available (Edwards et al. 2007; 2010), and (3) expert opinion (Edwards et al. 2010).

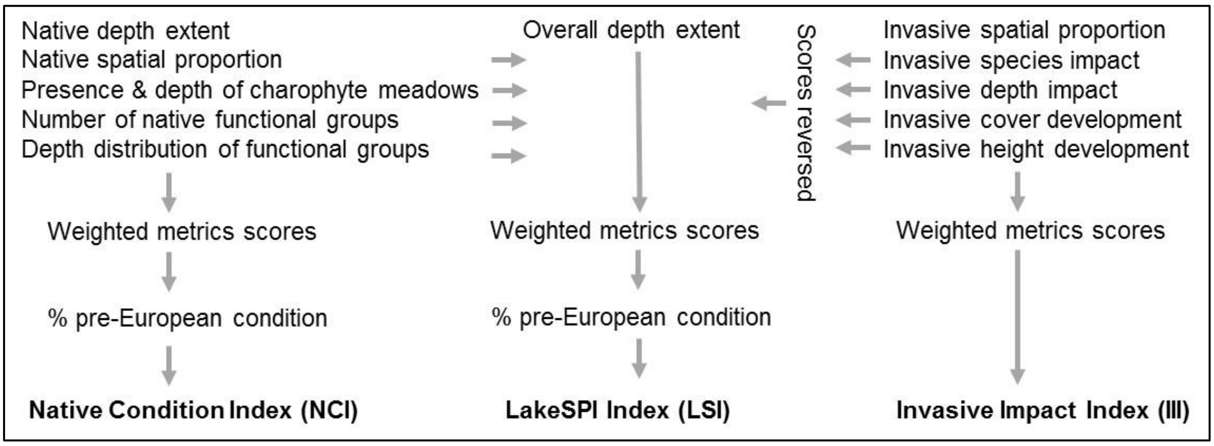

Figure 1. The LakeSPI method showing the various vegetation elements that contribute to the three index scores (modified from de Winton et al. 2012).

It is generally accepted that the health of the macrophyte community is a key indicator of lake health, especially in shallow lakes (Scheffer 2004), but also in deep lakes (Kelly & Hawes 2005 and references therein). The alternative stable state theory (Scheffer 2004) suggests that in shallow lakes, a healthy macrophyte community generally inhibits severe phytoplankton blooms. However, a study of the relationship between LakeSPI scores and TLI in Waikato lakes showed at best only weak correlations between the two indices of lake health (Verburg et al. 2010; Schallenberg & Schallenberg 2018). This indicates that the information in LakeSPI assessments is not redundant to that in TLI assessments and that both indices potentially provide important complementary information about lake health.

The NPS-FM (2020) acknowledges the contribution of LakeSPI to assessments of lake health by including the native condition index and the invasive impact index as lake attributes. As such, regional councils will be mandated to measure and report on three components of TLI as well as two components of LakeSPI (Table 1). However, limits will not be required to be set in relation to the macrophyte attributes. Instead, action plans will need to be instated to demonstrate a commitment to achieving the guideline values in future.

Key point: *TLI and LakeSPI are complementary, normative lake health indices, together covering aspects of the condition of the pelagic and littoral zones of lakes. In Waikato lakes, the two indices are only weakly correlated, indicating there is little redundant information in them.*

3.3. The TLI and ecological integrity (Schallenberg et al. 2011)

The use of concepts such as "life-supporting capacity" (RMA 1991) and "ecosystem health" (NPS-FM 2020) in central government policies and acts of Parliament mandates the safeguarding of diverse lake values, beyond safeguarding only the trophic state of lakes. In support of this, a large body of academic work has attempted to develop more holistic ecological value concepts such as 'ecosystem health' (Steedman 1994; Scrimgeour & Wicklum 1996; Rapport et al. 1998), 'biotic integrity' (Karr & Dudley 1981; Karr 1996) and 'ecological integrity' (Miller 1991; Barbour et al. 2000; Bunn & Davies 2000). These concepts may be closely aligned to the Māori concept of *mauri*, which can be translated as the embodiment of an 'essential life force" (Tipa & Teirney 2006).

Such concepts have been criticised as being subjective and normative (e.g., Peters 1991; Sagoff 2000). However, normative concepts like ecological integrity (EI) may be useful ion some domains precisely because they do incorporate values and value judgements and, therefore, connect science more directly to policy goals and objectives. Furthermore, such concepts may better reflect how humans perceive lake conditions than merely some information on water quality.

The concept of ecological integrity was developed and refined for New Zealand's terrestrial (Lee et al. 2005) and freshwater environments (Schallenberg et al. 2011). Schallenberg et al. (2011) proposed that freshwater ecological integrity be defined as:

> *The degree to which the physical, chemical and biological components (including composition, structure and process) of an ecosystem and their relationships are present, functioning and maintained close to a reference condition reflecting negligible or minimal anthropogenic impacts.*

Therefore, aquatic ecological integrity is a "wilderness-normative" concept (see Manuel-Navarette et al. 2004) that places a measure of pristineness (i.e., distance from a reference condition) at the core of the ecological integrity concept. In addition to pristineness, Schallenberg et al. (2011) also proposed three other quantifiable components of freshwater EI: nativeness, (bio)diversity and ecological resilience (Table 2).

Table 2. Suggested list of attributes for the assessment of ecological integrity in lakes (from Schallenberg et al. 2011).

Component of EI	Indicator of attribute	Examples of related stressors
Nativeness	**Catch per unit effort (CPUE) of native fish**	**Invasion by/introduction of exotic species**
	Percentage of species native (e.g., macrophytes, fish)	**Invasion by/introduction of exotic species**
	Absence of invasive fish and macrophytes	**Invasion by/introduction of exotic species**
	Proportion of shoreline occupied by native macrophytes	**Invasion by/introduction of exotic species**
Pristineness		
a. Structural	**Depth of lower limit of macrophyte distribution**	**Eutrophication (benthic effects)**
	Phytoplankton community composition	**Eutrophication**
b. Functional	**Intactness of hydrological regime**	**Connectedness, abstraction, irrigation, artificial human barriers**
	Continuity of passage to sea for migratory fish (potentially indicated by diadromous fish)	**Connectedness, artificial human barriers**
	Water column DO fluctuation	**Eutrophication**
	Sediment anoxia (or rate of change of redox state with depth)	**Anoxia, eutrophication (benthic effects)**
c. Physico-chemical	**TLI (or its components)**	**Eutrophication**
	Non-nutrient contaminants	**Depends on pressures**
Diversity	**Macrophytes, fish, invertebrate diversity indices**	**Loss of biodiversity**
Resilience	**Number of trophic levels**	**Loss of top predators**
	Euphotic depth compared to macrophyte depth limit	**Macrophyte collapse**
	Instance/frequency of macrophyte collapse or recorded regime shifts between clear water and turbid states	**Macrophyte collapse**
	Compensation depth compared to depth of mixed layer	**Potential for light or nutrient limitation of phytoplankton growth**
	N:P nutrient balance (DIN:TP)	**Risk of cyanobacterial blooms**
	Presence of potentially bloom-forming cyanobacteria	**Risk of cyanobacterial blooms**

"TLI (or its components)" is an indicator that appears in Table 2 (from Schallenberg et al. 2011) under the sub-component of ecological integrity called physico-chemical pristineness. This recognises that anthropogenic eutrophication of lakes relates to their departure from pristineness, and hence, to lake ecological integrity. Three other pristineness indicators (depth of deepest extent of macrophytes, phytoplankton community composition, and water column dissolved oxygen fluctuations) also describe aspects of eutrophication (see Table 2), further affirming that eutrophication relates to the pristineness component of ecological integrity.

The concept of aquatic ecological integrity has been considered by some regional councils in setting lake health targets and has also been used to derive reference conditions for different lake types in New Zealand (Schallenberg 2019; Abell et al. 2020).

Key point: *Eutrophication (as indicated by TLI or its components), is a key indicator of the pristineness component of lake ecological integrity. Reference conditions for TLI and its components have been derived for New Zealand lakes, allowing for estimates of departures from reference conditions to be made for TLI (Abell et al. 2020) and some of its components (Schallenberg 2019).*

4. What is the management utility of TLI?

Vant (1987b) suggested that "Rather than relating nutrient concentrations in a lake to trophic state and thence to suitability for beneficial uses, lake managers can simply relate the nutrient concentration to the suitability for beneficial uses.". Similarly, Peters (1991) stated that "thoughtful limnologists" avoid the use of trophic state and instead deal in quantifiable measurements. While the current freshwater policies in both New Zealand (NPS-FM 2020) and Europe (EU 2000) mention trophic state, they refrain from mandating the use of trophic level indices for monitoring and reporting on lake status and trends, but instead require the reporting of nutrient concentrations and phytoplankton biomass. Thus, the utility of tropic state and trophic state indices is not supported unanimously by limnologists. In Section 4, we discuss some of the strengths and weaknesses of trophic state as a concept and the TLI as a tool for lake management.

4.1. Conceptual advantages

In his critique of the trophic state concept, Carlson (1984) discussed the plethora of diverse definitions of trophic state as being a major problem hindering its general utility. One of the reasons given for the flourishing of definitions was the intermingling by some proponents of trophic state assessments of causes vs effects, drivers vs responses. This confusion may have contributed to the inclusion of disparate measures such as nutrient inputs, lake condition and lake productivity into more, increasingly divergent trophic state indices. Carlson (1984) suggested that a refocusing of the concept of trophic state to refer to phytoplankton biomass only could be a solution to the increasing vagueness which had begun to be associated with the trophic state concept. However, the problem with focusing trophic state only on biomass (e.g., Chl concentration) is that this can lead to the misclassification of some lakes because nutrient inputs, biomass, and productivity may be decoupled in some lakes by a variety of factors such as nutrient limitation (N- and/or P-limitation), light-limitation (turbidity), biomass-limitation (grazing), toxicants, and other factors (Carlson 1984; Carlson 1991). Therefore, Carlson (1984) proposed that a suitable alternative to a biomass-based trophic classification could involve the use of a dual classification scheme assessing both the actual biomass condition of the lake as well as its potential biomass condition.

TLI includes four indicators which are highly correlated among lakes. This could be considered to be inefficient, introducing redundant information into the index. But in the case of the TLI, inclusion of the nutrient and SD indicators appears to allow for some accounting for the risk that a lake could reach a higher potential phytoplankton biomass than is indicated by the observed phytoplankton biomass indicator (i.e., TLI(Chl)) alone. For example, Osgood (1984), Carlson (1991), Burns et al. (1999; 2000) and Verburg et al. (2010), were able to compare the scores of the various TLI sub-indices for individual lakes to help infer their nutrient limitation status. The inclusion of the three correlated lake health attributes, Chl, TP and TN, in the NPS-FM (2020) also highlights the utility of assessing multiple, correlated indicators of trophic state to avoid potential misclassifications of some lakes. The information in TP, TN and SD is not redundant if it accounts for eutrophication risk reflected by current in-lake conditions - conditions which could lead to higher phytoplankton biomass with a change in just one growth-limiting factor. For example, in cases where phytoplankton biomass is not limited by P availability, it could be limited by the availability of N or light (or by any of a number of other factors such as temperature, grazing, etc.). The inclusion of both N and P as well as Secchi disk depth (utilised in the TLI as 1 / SD, or light attenuation in the water column) allows for the state of these potentially growth-limiting factors to be assessed in addition to the actual phytoplankton biomass (Carlson 1991). For example, by comparing the differences in TLI(Chl) vs TLI(TP), TLI (TN) and TLI (SD), it is possible to assess whether the potential phytoplankton biomass could be higher than it is, given the nutrient concentrations and water clarity in the lake (e.g., Burns et al. 1999; Burns et al. 2009). Thus, the inclusion of the nutrient and SD variables in the TLI acknowledges, and can account for, the risk that phytoplankton biomass would increase should the concentrations of either nutrient or SD increase.

According to von Liebig’s resource-limitation theory, plants will grow until a certain resource becomes limiting, at which point that resource becomes the growth-limiting resource. If for example, the phytoplankton in a lake are N-limited while P is in plentiful supply, the phytoplankton biomass concentration will reflect the availability of N, rather than the surplus of P. Thus, in such a lake, TLI(TP) will overestimate the actual trophic state as defined by phytoplankton biomass (i.e., TLI(Chl)). However the relatively elevated TLI(TP) indicates that the phytoplankton biomass could increase if there were an increase in N availability. This is not an inconsequential risk, because if an N-fixing phytoplankter were to bloom in the lake, it could utilise atmospheric N and in so doing could also utilise the surplus P. This could allow the proliferation of phytoplankton to a biomass level higher than that indicated by measured TLI(Chl) and TLI(TN). A similar case could occur with regard to light limitation as indicated by the TLI(SD).

However, in complex, multi-species ecosystems such as lakes, the situation is more complicated and there is the potential for resource co-limitation (i.e., multiple resources limiting phytoplankton production; Lewis & Wurtsbaugh 2008), as opposed to growth limitation by a single resource. Thus, by accounting for Chl, TN, TP and light attenuation (potential light limitation; Carlson 1991), the multimetric TLI accounts both for actual phytoplankton biomass as well as for potential phytoplankton biomass. In other words, it appears to account to some degree for the increased risk of eutrophication that results from unbalanced resource supply ratios as well as for co-limitation. This risk is lessened in lakes where the resource supply ratios are relatively balanced - where the four TLI sub-indicators should give similar TLI scores.

Burns et al. (2000) provided a method to remove the effects of wind-induced sediment resuspension from the calculation of percent annual change (PAC) in the TLI. Wind-induced sediment resuspension not only temporarily reduces water clarity, but also temporarily increases water column Chl and nutrient concentrations (Schallenberg & Burns 2004). This is an issue that is particularly important in shallow, devegetated lakes. The recommended method involves measuring total suspended solids (TSS) in addition to the TLI components and then using the residuals of each TLI component plotted against TSS as a way to normalise the TLI components for the effect of wind-induced sediment resuspension. In lakes in which the correlations with TSS are statistically significant, this “deweathering” must be done instead of deseasonalising the time series data. This methodology doesn't affect the TLI; it only normalises the calculation of the PAC.

Thus, while the Burns et al. (2000) TLI protocol adjusts for intermittent wind effects on the TLI time trend calculation (the PAC), it doesn't remove the effect of sediment resuspended from the lake bed on the TLI itself. Thus, the TLI considers resuspended sediment to contribute to the trophic state of the pelagic zone. Therefore, algae growing on the lake bed and/or phytoplankton recently sedimented on to the lake bed as well as nutrients associated with the sediments all contribute to the TLI if/when they are resuspended by turbulence into the water column. In support of this, it has been shown that both nutrients and sediments entrained by turbulence into the water column can stimulate the growth phytoplankton (Schallenberg & Burns 2004) and, hence, resuspended lake bed sediments can have both direct and indirect effects on the trophic state of a lake.

The management utility of the TLI is highlighted by its current wide use by regional councils for lake objective-setting, monitoring and reporting. The problem of multiple, diverse definitions and methodologies can be overcome by the consistent use of one trophic state definition and methodology, such as strict adherence to the Burns et al. (1999; 2000) TLI protocol. By using Chl as the primary TLI indicator, the TLI is primarily an index reflecting the phytoplankton biomass in lakes (Burns et al. 1999; 2000), which is generally understood by limnologists and lake managers to be an integral component of lake health. In addition, the TLI index is useful in lake management because it accounts to some degree for eutrophication risk, or the potential phytoplankton biomass that could accrue with changes to the lake conditions.

Key point: *The TLI is conceptually useful because it reflects a normative value related to lake health which integrates information on both the current trophic state and the risk of phytoplankton proliferation as indicated by unbalanced resource supply ratios.*

4.2. Practical utility

While the TLI has some advantages in relation to lake management, the TLI is not an ideal ecological response variable for understanding ecosystem functioning, for modelling/scenario testing, or for limit-setting (Table 3). This is because the TLI integrates four indicator variables, which may change in opposing directions, resulting in the potential to obscure important changes in individual indicators. This is a general issue with multimetric indices: components of TLI may exhibit synergistic or compensatory effects which would be difficult to ascertain from the TLI index value alone or from changes in the index over time. This can be remedied by also analysing the indicator components of the TLI, which can provide more insight into lake functioning than simply assessing the overall TLI index (Osgood 1984; Carlson 1991; e.g., Burns et al. 2000; e.g.,Verburg et al. 2010). For example, a disproportionate increase in TLI(TP) over TLI(TN) could suggest conditions favourable to blooms of N-fixing cyanobacteria, which are of management concern for their potential to produce cyanotoxins.

The TLI may also have limited utility as an ecological response variable for limit-setting purposes. For example, if a lake TLI objective were used to set limits on catchment land uses, the fact that TLI aggregates four indicators means that the relationship between TLI and land use is likely to be complex - certainly not as direct as if limits were set for individual drivers of trophic state.

There are demonstrated relationships between land uses and associated nutrient and sediment impacts on rivers and lakes (Larned et al. 2019). As a result, the TLI responds directly and indirectly to land use and land use change. Climate change (warming, etc.) is generally thought to increase the trophic state of lakes (Hamilton et al. 2013; Schallenberg & Hamilton 2016), but expected changes in various climate drivers and in in-lake responses, such as Chl, can be quite complex and therefore hard to predict, both at the scale of an individual lake and in relation to effects on lakes at regional and national scales. For example, the effects of warming on water column stability may be offset by the effects of increasing wind energy on water column stability (Bayer et al. 2013). Similarly, the stimulatory effects of warming and increasing pCO_2 on phytoplankton productivity may be offset by intensified zooplankton grazing pressure and infection rates of algal pathogens, also due to warming.

Similarly, the impacts of invasive species on TLI is complex, depending on the specific invasive species in question. For example, if lakes are invaded by a diatom (Novis et al. 2020) or by planktivorous fish (Jeppesen et al. 2000), then TLI could increase. On the other hand, if lakes are invaded by *Daphnia* (Burns 2013), then grazing pressure on phytoplankton may increase, resulting in a decreased TLI.

In general, the management utility of the TLI depends on whether one is managing a lake to meet a values-oriented policy, or for improving ecological understanding. Being a normative index, it has more value as a tool for monitoring lake health state and trends and for prioritising investment in restoration than it does in providing understanding of ecosystem functioning, optimisation of restoration actions, or in limit-setting.

Table 3. Assessment of the utility of the TLI for various lake management tasks.

Task	Utility	Notes
1. State and trend monitoring	**Useful if sampled and calculated consistently**	**Has been used in this capacity by regional councils for 20 years**
2. Restoration prioritisation	**Useful if sampled and calculated consistently**	**As the TLI reflects a lake health value that is intuitively, widely understood, it is a useful tool for prioritising investment in lake/catchment restoration**
3. Understanding ecosystem functioning	**Only broad scale functioning - emergent property of lakes**	**Due to it being an integrative index, the TLI doesn't facilitate the understanding of the dynamics of specific drivers or ecological responses, nor of relationships between the two**
4. Indicator of risk of algal blooms	**Useful, by comparing the levels of different TLI components**	**High TLI(TP):TLI(TN) could indicate conditions favourable to cyanobacterial blooms. High TLI(SD) in relation to the other TLI components could indicate light-limitation of phytoplankton production and a risk of proliferation if water clarity improves.**
5. General ecological response variable, e.g., for limit-setting	**Not very useful**	**Due to it being an integrative index, the TLI doesn't facilitate the understanding of the dynamics of specific drivers or ecological responses, nor of relationships between the two**
a. Sensitivity to land use change	**Sensitive, in general**	**Land use change often results in fluxes of nutrients and sediments from the catchment to lakes**
b. Sensitivity to climate change	**Somewhat sensitive**	**Climate change impacts are complex and varied and may offset one another**
c. Sensitivity to invasive species	**Sensitive or not depending on the invasive species and severity of invasion**	**The effects of invasive species on TLI depend on whether a particular invasive species enhances nutrients, sediment and phytoplankton biomass or suppresses these**

Key point: *Being a normative index, the TLI has more value as a tool for monitoring lake health state and trends and for prioritising investment in restoration than it does in providing understanding of ecosystem functioning, optimisation of restoration actions, or in limit-setting.*

5. What are the effects of inconsistencies in the way the TLI is implemented?

The Burns et al. (2000) TLI manual describes in detail the protocols for determining the TLI, including sampling techniques and equipment, laboratory analysis of samples, management and surveillance of data, data processing, and the optimisation of sampling strategies. A software product called "LakeWatch" was also produced to facilitate the correct calculation of TLI. Nevertheless, it is apparent from surveying regional council TLI data and methods, that some inconsistencies have occured in the way TLI has been implemented.

Two different trophic level monitoring strategies were recommended by Burns et al. (2000): baseline monitoring and routine monitoring. Baseline monitoring involves measurement of a larger suite of limnological variables in addition to the TLI variables. There were three stated purposes to baseline monitoring: i) to set up a robust baseline value of TLI, ii) to better understand the basic limnology of the lake being monitored, and iii) to help develop an efficient and economic system of routine monitoring of the lake. It was recommended that lakes have at least 2 sampling stations and that sampling should occur monthly and should be carried out for at least 3 years. After this period, a careful analysis of the data should inform the routine monitoring programme, which could then implement an optimal number of sampling sites, optimal sampling frequency, etc.

Using this two-tiered sampling strategy, different numbers of sites and different sampling frequencies could have been utilised for some years, until a routine monitoring strategy was eventually implemented. This could produce inconsistencies in the historical TLI data that has been reported and archived.

Some potential problems have been report with the LakeWatch software. For example, the algorithm in LakeWatch for calculating the TLI(SD) is different to that in the Burns et al. (2000) TLI protocol (Fig. 2). In addition, some confusion has been expressed as to how LakeWatch averages samples across sites and carries out some other calculations. These issues have likely resulted in some inconsistencies in the way TLI has been calculated because some people have used the algorithms and methodology of the Burns et al. (2000) protocol, while others used LakeWatch.

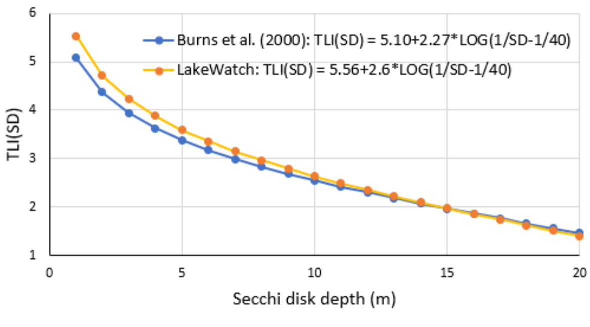

Figure 2. Comparison of the TLI(SD) calculated by the Burns et al. (2000) algorithm and by LakeWatch software.

5.1. Secchi disk depth and TLI3

Secchi disk depths are not reported for a number of monitored lakes and, as a result of this, Sorrell (2006) and Verburg et al. (2010) used TLI3 (i.e., TLI calculated without SD data) instead of TLI (or as they named it, TLI4) in their national-scale analyses of lake water quality. Verburg et al. (2010) indicated that TLI3 and TLI4 were highly correlated in a dataset of 70 lakes, but they didn't provide information on potential bias by using TLI3, and it is unknown how well TLI3 would correlate with TLI4 in the lakes which didn't have SD data. One reason that SD data are not recorded is that for some shallow lakes with moderate or high water clarity, the Secchi disk may still be visible where it interacts with macrophytes or lies on the lake bed, invalidating the SD measurement. Indeed, for some lakes the Secchi disk is not an appropriate tool for measuring water clarity. TLI3 has also been used in research studies instead of the TLI (e.g., Abell et al. 2020). While the substitution of TLI3 for TLI may be adequate for broad scale comparisons of lake state and trends, a deeper statistical analysis of the relationship between TLI3 and TLI should be undertaken, examining not only the correlation between the two, but the magnitude of the residuals as well as any bias in inferred TLI values that this substitution could cause. For example, could the relationship of SD to the TLI differ systematically in lakes in which it is difficult to estimate SD (i.e., lakes where only TLI3 can be calculated)? See Section 6.2.7 for more information on statistical comparisons between TLI3 and TLI.

Given our current level of understanding, the omission of SD from the TLI calculation could undermine the TLI because phytoplankton in some lakes, such as those with consistently impaired water clarity, may be light-limited (Carlson 1991). The inclusion of the TLI(SD) accounts for the risk of eutrophication in lakes where phytoplankton biomass is only constrained by low water clarity - lakes which are "primed" for phytoplankton proliferation should water clarity increase. However, this particular risk of light-limitation could also be indicated by a disproportionately low TLI(Chla) compared to TLI(TN) and TLI(TP). This requires a further step in the analysis of TLI data, beyond merely calculating the TLI index.

Two ways forward through this SD dilemma seem apparent, given the current state of knowledge: (1) consistently use only TLI3, or (2) adhere to TLI by developing a alternate method for measuring water clarity in shallow lakes and substitute this measure for SD in the TLI calculation. If option 1 is preferred, it could be useful to first undertake a thorough, national-scale analysis of the influence of TLI(SD) on TLI in a wide variety of lakes with different concentrations of various light-attenuating substances. This would provide a fuller picture of the errors and biases that could result from dropping SD as a component of TLI. If option 2 is preferred, an alternative method (e.g., horizontal black disk, etc.) should be agreed upon and a standard protocol should be developed for its use. For the substitution to work, a strong correlation (e.g., R^2 > 0.80) should exist between the alternate method and SD measurements. The readings by the alternate method should then be calibrated to SD so that the measurements can be converted to estimated SD. SD_{est} could then be used in TLI4 for lakes in which it is difficult to measure SD.

Key points: *SD has been problematic in the calculation of TLI for some lakes. Either TLI3 should be adopted as the standard TLI formulation (i.e., SD should be dropped in the standard TLI calculation) or TLI should be adhered to by developing an alternate way of measuring water clarity in lakes where SD measurements are problematic. Both of these suggestions require further analysis.*

5.2. Logarithmic transformation of the data

The Burns et al. (2000) protocol states, "In each lake for each year [the TLI] regression models were used to calculate the trophic level indices TL(SD), TL(TP) and TL(TN) from the annual average of variables SD, TP and TN." Thus, the protocol transforms the averages into log units rather than averaging the logged data (as has been done by some regional councils). If the data collected over the year are normally distributed, then the resultant TLI will be the same either way it is calculated. However if the data are log-normally distributed (i.e., skewed), then taking the log of the averages results in an elevated TLI, as compared to taking the average of the logged data. This is because the mean and median are the same value in a normal distribution, whereas the mean is greater than the median in a log-normal distribution.

Because the TLI was developed and calibrated using the calculation procedure of Burns et al. (2000), this procedure should always be used.

> ***Key Point:*** *While taking the annual average of the monthly (logged) TLI data may have statistical advantages (especially if the data are log-normally distributed), the Burns et al. (2000) protocol was developed and calibrated by calculating the TLI on the annual average of the monthly data. This could result in differences in the calculated TLI. Thus, regional councils should adhere to the Burns et al. (2000) protocol so that TLI values are properly calibrated and are comparable.*

5.3. Sampling optimisation

5.3.1. Sites

For baseline monitoring (to establish a trophic state baseline for a lake), as opposed to routine monitoring, Burns et al. (2000) recommended sampling at least 2 sites per lake. They stated that an analysis of the correlation between the sites should be carried out after 2 (but preferably 3) years of baseline monitoring, to determine whether having multiple sites provides valuable additional information about the lake's trophic state.

The Burns et al. (2000) protocol examined within-lake variation in TLI by examining the differences in TLI estimates between two sites in each of two lakes (length = 3.1 and 2.8 km, surface area = 2.03 and 3.46 km^2). While the within-lake differences in TLI estimates among sites were found to be negligible, this is unlikely to be the case for larger lakes. In fact, no spatial sampling regime was recommended in the TLI protocol for large lakes (lakes they defined as being >100 m depth and >100 km^2 in surface area). Instead, Burns et al. (2000) recommended individually designed monitoring programmes for large lakes. In any lakes that have embayments, where water exchange with the main lake may be reduced, additional water quality monitoring sites should be considered to provide greater representativeness of the trophic state of such lakes. In addition, the TLI protocol recommended including extra monitoring sites in lakes which may have relevant features such as shoreline urban areas which may locally influence the lake trophic state via stormwater discharges and other pollution point sources. No guidance is provided in Burns et al. (2000) as to how to aggregate the results of trophic state monitoring from more than one monitoring site per lake.

Some regional council TLI data uploaded to the LAWA website were from samples collected from the lake shore, rather than mid-lake. This is not recommended in the Burns et al. (2000) TLI protocol because sampling from shore cannot allow depth-integration over the mixed layer depth. Furthermore, vertically migrating phytoplankton such as *Ceratium hirundinella* are less likely to be sampled near the shore, whereas scum-forming phytoplankton such as *Dolichospermum* spp. can accumulate on lake shores due to redistribution and focusing by wind. Therefore, sampling from shore could underestimate or overestimate the trophic state of a lake, especially in lakes with vertically migrating or floating phytoplankton.

Key point: *To address within-lake spatial variation in TLI, Burns et al. (2000) recommended sampling multiple sites as part of a "baseline monitoring" programme. After analysis of the correlation among sites, the number of sites sampled may be reduced, as part of the "routine monitoring" programme. Large lakes have substantial spatial variability and require individually-designed monitoring programmes to satisfactorily account for spatial variability in TLI.*

5.3.2. Water layers

The TLI's effectiveness in assessing the trophic state of lakes depends on whether it effectively samples the phytoplankton biomass in a lake. The Burns et al. (2000) sampling protocol focuses sampling on the mixed layer, as determined by the analysis of temperature and dissolved oxygen profiles. For seasonally stratified lakes, Burns et al. (2000) suggested that the appropriate water mass in which to assess TLI is the mixed layer, which should be sampled at four depths: 0.2 m depth, one-quarter, one-half, and three-quarters of the mixed layer depth. These samples are pooled for the analysis of Chl, TN and TP. For polymictic lakes, the TLI protocol recommends the sample be collected from a depth that is one-quarter of the maximum depth of the lake.

In lakes where the phytoplankton are distributed only within the mixed layer, the recommended sampling strategy will integrate effectively across the range of depths that contains the phytoplankton. However, in some lakes, vertically migrating phytoplankton (e.g., dinoflagellates such as *Ceratium hirundinella*; James et al. 1992) and those which can regulate their depth in the water column (e.g., cyanobacteria such as *Planktothrix* spp.; Dokulil & Teubner 2012) may occur in, or even below, the thermocline (Hamilton et al. 2010; Leach et al. 2017). As such, the recommended sampling protocol may not account adequately for the phytoplankton biomass in lakes which have deep chlorophyll maxima. Chlorophyll profiles, either (i) peaking at or below the thermocline and/or (ii) extending into the hypolimnion, have been reported for Lakes Taupo, Rotoma and Tarawera (Hamilton et al. 2010), Hayes and Johnson (Mitchell & Burns 1981) and Wanaka (Bayer et al. 2015). So this phenomenon is likely to be common, at least in lakes where the euphotic zone extends into the thermocline (Hamilton et al. 2010). However no estimates have been published so far of the percentage of monitored lakes in New Zealand that deep chlorophyll maxima occur in. Sampling using an integrated tube sampler may or may not capture deep chlorophyll maxima, depending on whether the tube extends into the metalimnion and upper hypolimnion – zones where deep chlorophyll maxima tend to occur.

Key points: *While the sampling depths recommended in Burns et al. (2000) may be useful for sampling the phytoplankton habitat in most lakes, such sampling will underestimate the trophic state of lakes with deep chlorophyll maxima (i.e., lakes with phytoplankton layers in, or below, the thermocline). Samples for TLI assessment should not be taken from shore.*

5.3.3. Frequency of sample collection

Burns et al. (2000) discussed the potential utility of various sampling frequencies for TLI monitoring, both in terms of the optimal number of samplings per year and in terms of the permitted omission of some sampling years within long-term monitoring programmes. This advice may have caused some inconsistencies in how the TLI has been estimated for different lakes and/or at different times.

Such inconsistencies in sampling frequency for monitoring were deemed serious enough for NIWA to exclude around half of New Zealand's monitored lakes from their snapshots of lake water quality carried out for the Ministry for the Environment in 2015 and 2019; only 65 lakes were assessed in the Ministry for the Environment's Aotearoa 2015 report (MfE 2015) and only 58 lakes in Aotearoa 2019 (MfE 2019). Burns et al. (2000) stated that "Optimum sampling for routine [TLI] monitoring would be monthly if possible...". In addition, for trend detection, it is advisable that monthly sampling be continuous among years, without gap years. Large, deep, oligotrophic lakes may not require monthly sampling, but as Burns et al. (2000) stated, such lakes should have individually-designed monitoring programmes informed by initial high frequency baseline monitoring. The resultant routine monitoring programmes for these lakes would presumably have been informed by observed patterns of seasonal variability over time.

Very dynamic shallow lakes, or lakes with short water residence times may benefit from sampling at shorter than monthly intervals to ensure short-lived blooms and other events are captured by the monitoring strategy.

Key points: *A monthly TLI sampling frequency is adequate for most lakes. The frequency could potentially be lower for large, deep oligotrophic lakes and higher for eutrophic shallow lakes, or lakes with short water residence times. Once the appropriate sampling frequency is determined, sampling should occur continuously, without gap-years, to allow for accurate trend detection.*

5.4. Time period of TLI calculation and the "limnological year"

Burns et al. (2000) recommended that TLI be calculated for the "limnological year" (starting and ending on 1 September and 31 August, respectively) because most lakes were likely to be isothermal (vertically mixed) at this time of year. Thus, the annual growth period for phytoplankton is not divided among two years. This is a sensible rationale as it allows comparative interannual analysis of trophic states that account for complete growing seasons.

The annual peak of phytoplankton productivity in deep, oligotrophic lakes (e.g., Lakes Taupo, Wakatipu and Coleridge) often occurs in winter (Vincent 1983; Schallenberg & Burns 1997; James et al. 2001); but even for these lakes, the end of August/beginning of September is a period of low phytoplankton biomass and is an appropriate time to define a "limnological year".

It is known that different regional councils and organisations that assess TLI (i.e., LAWA, NIWA) use different time frames for the calculation of TLI. This can lead to inconsistencies in reporting.

Key point: *Different time periods are used by different organisations for the calculation of the annual TLI assessment (e.g., January 1 - December 31, July 1 - June 30, September 1 - August 31). This should be standardised and for most lakes, the period September 1 to August 31 is most appropriate integration period.*

5.5. Can there be a consistent approach?

TLI has been used by regional councils for around 20 years and national TLI data have been collated, analysed and reported by the Ministry for the Environment for almost 15 years. Thus, the TLI has played an important role in the monitoring and reporting of the condition and trends of water quality of New Zealand lakes. In NIWA's collation and analysis of national-scale TLI data for the Ministry for the Environment, NIWA rejected the data for approximately half of the lakes because they were not complete or consistent. Thus, since 2015, the assessment of TLI condition and trends by the Ministry for the Environment has not been assessed for around half of New Zealand's monitored lakes (MfE 2015; MfE 2019). This highlights the importance of appropriate, consistent and comprehensive TLI data collection and analysis.

Although the Burns et al. (2000) TLI protocol is comprehensive and detailed, there have been inconsistencies in the way TLI has been measured, calculated and reported. Strict adherence to the TLI protocol is recommended to improve the consistency of the TLI data collected and reported (Table 4).

Table 4. Suggestions for reducing inconsistencies in the implementation of TLI.

Issue	Recommendation
Baseline monitoring to inform routine monitoring programme	**Follow the Burns et al. (2000) protocol and institute 3 years of monthly baseline monitoring for at least 2 sites per lake. Analyse correlations among sites and temporal dynamics to inform an appropriate routine monitoring programme.**
Logarithmic transformation	**Follow the Burns et al. (2000) protocol and calculate the TLI based on the annual average data.**
Time period of calculation	**Follow the Burns et al. (2000) protocol and calculate and report TLI for the "limnological year".**
Water depths	**Follow the Burns et al. (2000) protocol. However, see Section 6 for suggested improvement in water layer selection.**
Use of TLI3	**Option 1: Use TLI3 consistently for all lakes. Option 2: Use the original TLI consistently for all lakes. For lakes in which it is problematic to measure SD, develop an alternate method for measuring water clarity, calibrate it to SD, and use the estimated SD in TLI calculations for those lakes for which it is problematic to measure SD.**

Key points: *While it is essential that TLI be used in a consistent manner, there have been inconsistencies in its use. Thus, monitoring data for many monitored lakes has been excluded from national assessments of water quality. The Burns et al. (2000) protocol is clear and should be used whenever possible.*

6. Should the TLI be updated?

6.1. TLI: a robust indicator of a normative concept

Trophic state is a widely used normative concept and the New Zealand trophic level index (Burns et al. 2000) is an indicator that has been used to assess the trophic state of the mixed layer of New Zealand lakes for the past 20 years. The TLI is a useful index as it integrates four broadly correlated indicators of trophic state. In doing so, it provides information about likely imbalances in the resources that phytoplankton require for growth and the potential for phytoplankton proliferation. As the TLI is a carefully developed, expert-defined, and well-calibrated index of trophic state, it is not advisable to alter the equations or weightings that underpin the TLI calculation, as this would redefine the TLI. Rather than adjust the TLI, it would be more transparent and more prudent to create a new lake trophic state index, if modification to the TLI is deemed necessary.

This review has discussed some of the TLI's strengths and weaknesses and of the concept of trophic state, upon which it is based. Regarding its future use, there seem to be three options to carefully consider:

1. Continue using the TLI in its current form, allowing an extension of the 20-year TLI dataset and the improving ability to discern temporal trends in TLI that the historical data affords,

2. Follow the lead of the NPS-FM and EU Water Framework Directive and stop using trophic level indices, instead focusing on individual measurements of trophic level indicators,

3. Continue using a modified, potentially recalibrated trophic state index, but giving it a different name to avoid confusion with the TLI.

Regardless of the pathway chosen, at least some of the sampling, measurement and calculation inconsistencies described in Table 4 need to be addressed to allow nation-wide comparisons of lake water quality. It is strongly recommended that if TLI is dropped, then the constituent trophic state indicators Chl, TN, TP and water clarity should continue to be measured and reported on in our lakes.

Key point: *The TLI has been used as a key assessment of lake water quality for over 20 years. There are some advantages and disadvantages to continuing to use it. It may also be advantageous to update the TLI by making some improvements to it.*

6.2. Considerations for updating the TLI

6.2.1. Use of annual averages

The limits set for trophic state variables in the NPS-FM are annual medians and annual maxima. In contrast, the TLI is expressed as the log of annual means. As discussed in Section 5.2, the use of means and logarithmic transformation can be problematic and could be remedied by use of medians instead of means. If the TLI were to be updated, the use of annual medians instead of annual means could be beneficial.

The TLI is primarily calibrated against Chl data, an indicator of phytoplankton biomass. In lakes susceptible to phytoplankton blooms, Chl can peak episodically and rapidly, markedly changing the instantaneous trophic state of the lake. The susceptibility of lakes to severe phytoplankton blooms is an important human-perceived value of lakes and, for this reason, the NPS-FM includes an annual maximum Chl limit for lakes. If the TLI were updated, then it could be beneficial to calculate an annual maximum (or a 95^{th} percentile) and median TLI, instead of basing the TLI on the annual means.

Key points: *The Burns et al. (2000) protocol is clear as to how the TLI should be calculated. However, the recommended method of taking the log of the annual mean concentrations of TLI components is not the most statistically robust approach to calculating the TLI. Instead, calculating and reporting the annual median is preferable. In addition, adding an annual maximum (or 95^{th} percentile) TLI to the reported TLI data could be beneficial by indicating the severity of episodic deficits in trophic state (e.g., caused by episodic algal blooms).*

6.2.2. Scaling the TLI for different lake types

The NPS-FM (2020) doesn't include TLI in its list of lake attributes. However, it includes the trophic state attributes, Chl, TN and TP, which requires regional councils to set limits in relation to these. The NPS-FM attribute bottom lines apply to all lakes across New Zealand, but different TN limits have been set for seasonally stratifying and brackish vs polymictic lake types. This suggests that some consideration should be given to whether the TLI should be scaled differently for different lake types.

It is likely that for some lake types a more eutrophic condition is naturally more likely, or naturally more common (e.g., lowland lakes with large catchments), than for others (e.g., alpine lakes with small catchments). However, in the substnatial literature on trophic state, it is not scaled differently for different lake types. Thus, trophic state is invariant in relation to lake type, while the prevalence of different trophic states may vary among lake types.

If TLI were to be used widely for limit setting, then it could be appropriate to scale the TLI limits (i.e., the NPS-FM bands A, B, C, and D) differently for different lake types, as was done with TN in the NPS-FM (2020), rather than scaling TLI differently for different lake types. Various lake typologies could be considered, including that by Vant (1987a), who identified six lake classes and that by Sorrell (2006), who identified seven lake classes.

Key points: *The minimally-impacted trophic state may vary for different lake types. Rather than scale the TLI differently for different lake types, it is more sensible to set different TLI limits/guideline values for different lake types. This way, the TLI and the definition of trophic state have consistent meanings for all lakes.*

6.2.3. Scaling the TLI to account for co-variates

Factors such as lake size, depth, water residence time and mixing regime have been identified as important characteristics that can influence lake management (Vant 1987a). To this list of characteristics, climate (Schallenberg & Sorrell 2010), salinity (Schallenberg et al. 2003; 2010) and non-algal light attenuation (Carlson 1979; 1991) can be added. These characteristics very likely interact with, and influence, the trophic state of lakes. Thus, one could consider such factors as important co-variates, mediating the relationship between catchment inputs and TLI. Should the TLI be updated to account for the effects of such covariates?

Implementing such covariates in the calculation of the TLI would then scale the TLI differently for lakes which differed in some, or all, of these characteristics. As with scaling for lake type (Section 6.2.2), this would result in modification of the trophic state category attributed to different lakes, such that TLI becomes weighted for additional factors. Such adjustment of the TLI may be relevant to scientific study of trophic state, but this may be undesirable when assessing the lakes for human-perceived values, ecosystem services and uses. Adjusting the trophic state classification for co-variates could undermine the management utility of trophic level indices, such as the TLI. This may be why adjustments of trophic state classes for co-variates is not typically undertaken for any published trophic level indices.

The only co-variate which is accounted for in the Burns et al. (1999; 2000) TLI protocol is the effect of TSS on the percent annual change trend estimator (the PAC). Consistent with other trophic state indices, TSS is not used to adjust the TLI, itself.

> ***Key point:*** *While many factors can both directly and indirectly affect the trophic state of lakes, the calculation of TLI should not be adjusted for these factors. Rather, these factors could be accounted for in setting limit/guideline values for lakes in which co-variates clearly influences the relationship be anthropogenic activities and TLI. No accounting for sediment resuspension is required in the calculation of TLI.*

6.2.4. Submerged macrophytes as a contributor to trophic state

Canfield and Jones (1984) constructed a trophic state index which accounted for submerged plant biomass as well as phytoplankton biomass; however, the inclusion of macrophytes has not been widely accepted into the concept of eutrophication and trophic state. One difficulty with redefining trophic state to include submerged macrophytes is that plant biomass can vary substantially across the lake bed and, as such, it is difficult to estimate the lake-wide contribution of submerged plants to the total autotrophic biomass or nutrient content of a lake.

While submerged macrophytes contribute important lake ecological benefits and ecosystem services, proliferations of some invasive submerged macrophyte species can be detrimental to these. For example, the colonisation of lakes by *Egeria densa* promotes the collapse of the lake macrophyte community, often resulting in subsequent severe phytoplankton blooms (Schallenberg & Sorrell 2010). Thus, the relationship between lake health and macrophyte biomass may not be a positive monotonic relationship where lake health increases continuously with macrophyte biomass. Therefore, the inclusion of macrophyte biomass in the assessment of trophic state and TLI would be complex, requiring some accounting for the relative abundance of invasive macrophyte species and the potential negative effects of invasive macrophyte proliferations.

The importance of macrophytes in the health of lake ecosystems and in regulating lake water quality has been acknowledged by the inclusion of two submerged macrophyte indices as attributes in the NPS-FM (2020). These are the invasive impact index and the native condition index, both being sub-indices of the LakeSPI index (Clayton & Edwards 2006). Thus, these components of LakeSPI are seen as being complementary to water column trophic state indicators in the assessment of lake ecosystem health, not a replacement for trophic state monitoring (see Schallenberg & Schallenberg 2018). The NPS-FM (2020) recognises that the monitoring of both trophic state and macrophyte indices improves assessments of lake health.

> ***Key point:*** *The inclusion of submerged macrophytes as a component of trophic state has not received widespread acceptance. However, lake submerged macrophyte indicators can be complementary indicators of lake health. This is acknowledged by the inclusion of lake macrophyte indices in the NPS-FM (2020), which, together with trophic state attributes, contribute towards assessing lake ecosystem health.*

6.2.5. Sampling water layers, deep chlorophyll maxima, and the use of areal units to measure trophic state

When undertaking sampling of a lake for a trophic state assessment, the selection of sampling depths depends on whether the lake is deemed to be thermally stratified or isothermal. However, determination of this condition isn't always easy. The Burns et al. (2000) protocol suggests that when a lake exhibits > 3˚C difference between bottom water and surface water, then the lake should be considered to be thermally stratified. However, the protocol points out that this rule of thumb can be misleading in some circumstances. For example, there could be a transient, diel thermocline in the lake. Furthermore, in deep lakes, temperature differences of a fraction of a degree can prevent mixis during calm periods (e.g., Bayer et al. 2015). Thus, the consistent and accurate determination of the state of mixis of a lake at any point in time requires a sophisticated protocol that can be used rapidly, in the field. A software application is available for estimating mixed layer depth, thermocline depth, Schmidt stability, etc. based on temperature profiles (Read et al. 2011). It is possible such an app may be adapted specifically to assist with determining sampling depths for the TLI.

To obtain a depth-integrated estimate of the TLI, the Burns et al. (2000) protocol recommends sampling the lake's mixed layer down to a maximum of three-quarters of the mixed layer depth. To be able to do this in a stratified lake, pre-examination of the temperature and dissolved oxygen profiles is recommended to determine the thickness of the mixed layer. However, this sampling protocol is not guaranteed to sample the zone in which phytoplankton can grow and accumulate. In some lakes with high water clarity and shallow mixed layers, phytoplankton may grow within the thermocline or in the upper hypolimnion (Hamilton et al. 2010). Phytoplankton taxa that are either able to migrate vertically (e.g., dinoflagellates such as *Ceratium hirundinella*; James et al. 1992) or maintain buoyancy in specific water layers (e.g., cyanobacteria, such as *Planktothrix* spp.; Dokulil & Teubner 2012) are sometimes found growing in high concentrations below the mixed layer (Leach et al. 2018).

Therefore, if the TLI were to be updated, the sampling depths could be determined based on the Chl profile, as determined by vertical profiling with a sonde. This would redefine the sampled layer as the trophogenic zone (zone where phytoplankton can grow), rather than as the mixed layer. In some cases, the trophogenic zone may fall entirely within the mixed layer, but as indicated above, in some cases the trophogenic zone can extend into the thermocline and hypolimnion. Nowadays, many regional councils use CTDs or lake monitoring buoys, which contain *in vivo* Chl fluorescence sensors, for profiling lakes. These are suitable for determining the depth of the trophogenic zone. By inclusion of deep chlorophyll maxima in the sampling for TLI calculation, the TLI for some lakes would increase compared to using the Burns et al. (2000) protocol.

The issue of whether deep chlorophyll maxima should be accounted for in estimates of trophic state raises a fundamental question regarding whether the trophic state of a lake is best defined as a volumetric or areal assessment of a lake's trophic state variables (Carlson 1979; 1991). Burns et al. (2000) defined trophic state as the "life-supporting capacity per unit volume of a lake", but their TLI assumes that all phytoplankton are in the mixed layer. Thus, their TLI could be more accurately defined as the "life-supporting capacity per unit volume of the mixed layer of a lake".

Summer mixed layer thickness varies more than 7-fold among seasonally stratified New Zealand lakes (Davies-Colley 1988) and also varies greatly within lakes during the year. As the mixed layer thickness influences (i) how deep phytoplankton are mixed, (ii) the average light availability in the mixed layer and (iii) the amount of water that the phytoplankton in the mixed layer can grow in, the phytoplankton concentration in the mixed layer may not be a good indication of how much phytoplankton grows in a lake. Alternatively, this may be estimated by measuring the phytoplankton density per m^2 of lake area (e.g., Bayer et al. 2015; Carlson 1979). While a volumetric definition of trophic state focused on the mixed layer (i.e., TLI) is more likely to reflect human perception of phytoplankton biomass, an areal definition, which also includes deeper phytoplankton populations, is a better indication of the total phytoplankton biomass that a lake supports (Table 5).

Thus, if the TLI were to be updated, some consideration should be given to whether the trophic state of a lake could be better defined in areal, rather than in volumetric, units (Carlson 1979). Regression analyses using multi-lake datasets of Chl, expressed both in volumetric and areal units, vs nutrient load estimates to the water column could shed light on whether the definition of trophic state might be more meaningful if changed from volumetric units to areal units.

Table 5. Strengths and weaknesses of three ways of assessing trophic state, based on the layers sampled and on either the volumetric or areal expression of the data.

Objective of assessment	Volumetric, mixed layer	Volumetric, trophogenic zone	Areal, trophogenic zone
Human perception of trophic state (from surface)	Strongest correlation to casual visual perception	Includes deep phytoplankton layers which may not be casually perceived	Weakest correlation to casual visual perception
Assessment of whole lake phytoplankton biomass	Weakest correlation to whole lake biomass	Somewhat stronger correlation to whole lake biomass	Strongest correlation to whole lake biomass

Key points: *The Burns et al. (2000) TLI protocol ignores deep chlorophyll maxima, which occur in some lakes. To remedy this, if the TLI were to be updated, the selection of sampling depth should be made from Chl profiles, rather than from profiles of temperature and dissolved oxygen concentrations. Consideration should also be given to whether trophic state would be more usefully measured on a volumetric basis, or on an areal basis.*

6.2.6. Alternatives to Secchi depth as a component of TLI

The Burns et al. (2000) protocol stipulated SD as the measure visual clarity. This creates problems for the calculation of TLI if SD readings are not able to be made, as can occur in some shallow lakes where the Secchi disk may be visible while lying on macrophytes or on the lake bed.

Therefore, if the TLI were to be updated, an alternative measure of visual clarity should be considered (e.g., Davies-Colley et al. 2001). Alternatives could include turbidity, total suspended solids, black disk, or other optical measurements (see NEMS 2019).

Key point: *As a component of TLI, SD can be problematic to measure in some lakes. If the TLI were to be updated, an alternative method of measuring visual clarity should be used (at least in some lakes) for TLI assessments.*

6.2.7. Study of TLI3 vs TLI

In Section 5.1, the appropriateness of the common practice of substituting TLI3 for TLI was discussed. Verburg et al. (2010) justified using TLI3 as an index for reporting on trophic state because the correlation coefficient (R^2) between TLI3 and TLI in a subset of 70 lakes was 0.98. By substituting TLI3 for TLI, they were able to use data from an additional 42 lakes for national-scale reporting on trophic state (Fig. 3). This practice of substitution has been picked up by other researchers (e.g., Abell et al. 2020), but more statistical analysis should be undertaken to assess the importance of the contribution of SD to the TLI. For example, it was not clear whether the TLI data used in Verburg et al. (2010) were annual means or multi-year averages of annual means. In addition, their study only reported the correlation coefficient and P-value; the standard error of the prediction, residual mean square error or other estimators of potential bias should also be scrutinised. For example, Verburg (2012) showed that the generally strong correlation between Chl and SD amongst lakes was compromised in lakes with high natural sources of materials that absorb or scatter light (i.e., “optically challenged lakes”; Fig 4). Carlson (1991) explored these issues in some detail.

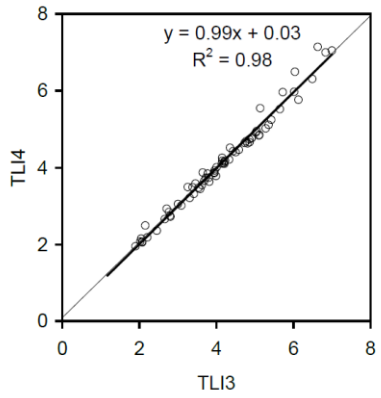

Figure 3. Correlation between TLI (TLI4) and TLI3. From Verburg et al. (2010).

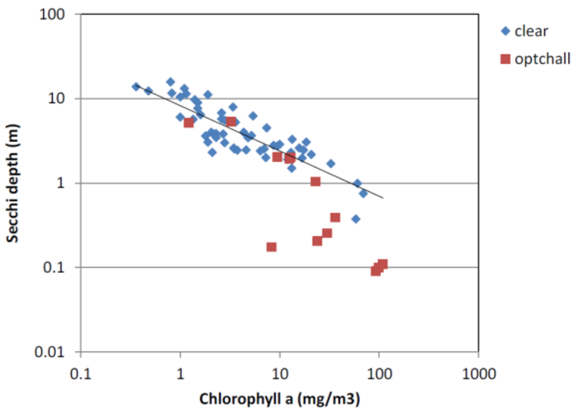

Figure 4. Regression of Secchi depth vs chlorophyll *a* for 54 lakes that do not contain elevated levels of non-algal substances that scatter or absorb light (“clear lakes”; blue diamonds). Lakes with naturally elevated levels of non-algal light-absorbing and light-scattering substances are also shown (“optically challenged”; red squares). From Verburg (2012).

Key point: *TLI3 (the TLI calculated without the SD sub-index) is sometimes used when researching or reporting lake trophic state. Before the substitution of TLI3 for TLI can be recommended, further statistical work should be done to determine the potential errors and biases that this substitution may result in.*

6.2.8. TLI and new data collection techniques

The TLI was developed at a time when grab sampling and individual probes and sensors, such as thermistors, were typically used to monitor lakes. Since the protocol was developed, new technologies for collecting data have become available and are now commonly used. For example, multi-probe data sondes (also known as a CTD) are now commonly used to simultaneously collect depth profiles of a several limnological parameters. This has sped up the collection of TLI-relevant data such as temperature, dissolved oxygen and Chl.

The now common use of optical *in vivo* fluorescence Chl sensors means that the depth of the trophogenic zone could replace the mixed layer depth in the calculation of a new type of TLI. If the TLI were to be updated, the sampling protocol should stipulate the use of *in vivo* Chl profiles for determining the sampling depths, so that deep chlorophyll maxima are also part of the assessment. Doing this may necessitate a recalibration of the TLI(TP), TLI(TN) and TLI(SD) sub-indices to a new TLI(Chl). However, Chl profiles obtained from the optical sensors should not be used as a substitute for solvent-extracted Chl measurements from grab samples in calculating the TLI(Chl).

Satellite and drone imagery is increasingly being used to assess optical properties of lake water (e.g., Lehmann et al. 2018). While this technology is not yet at the stage where it can be used to accurately estimate trophic state indicators, it can potentially be useful in assessing within-lake spatial variability of TSS and Chl. Accurate estimates of these attributes could be useful in determining how many sites may be required to assess the trophic state of large lakes and to determine where in the lakes the most representative sampling sites are located. Such information could be helpful in setting up a baseline monitoring programme, or in reviewing a routine monitoring programme, especially for large lakes.

Key point: *New sensor technologies are available which can provide much more detailed information on phytoplankton and total suspended sediment concentrations. Chl sensor profiling could improve the determination of sampling depths for TLI assessments, but optical Chla sensors should not replace solvent-extracted Chl measurements recommended for calculating the TLI by Burns et al. (2000).*

7. Recommendations and conclusions

This review highlights explores and highlights strengths and weaknesses of the TLI. Consideration of the information should inform future directions for the use of TLI in lake management. Figure 5 summarises a number of recommendations for three defensible ways forward regarding the future use of the TLI: (1) ceasing use of the TLI in favour of measuring its components separately, (2) continuing to use the TLI, but with improved consistency and adherence to the original Burns et al. (2000) protocol, and (3) updating the TLI to create a new, improved trophic state index.

The TLI has been a useful tool for lake management in that it imbodies trophic state characteristics of lakes that scientists, lake managers and the public intuitively understand and find useful. However, because it is a normative, multimetric index that not only attempts to quantify the state of phytoplankton density, but also the risk of proliferation associated with imbalances in the availability of key phytoplankton resources, it is an index that can be problematic to scientifically model and to interpret with respect to lake ecosystem functioning.

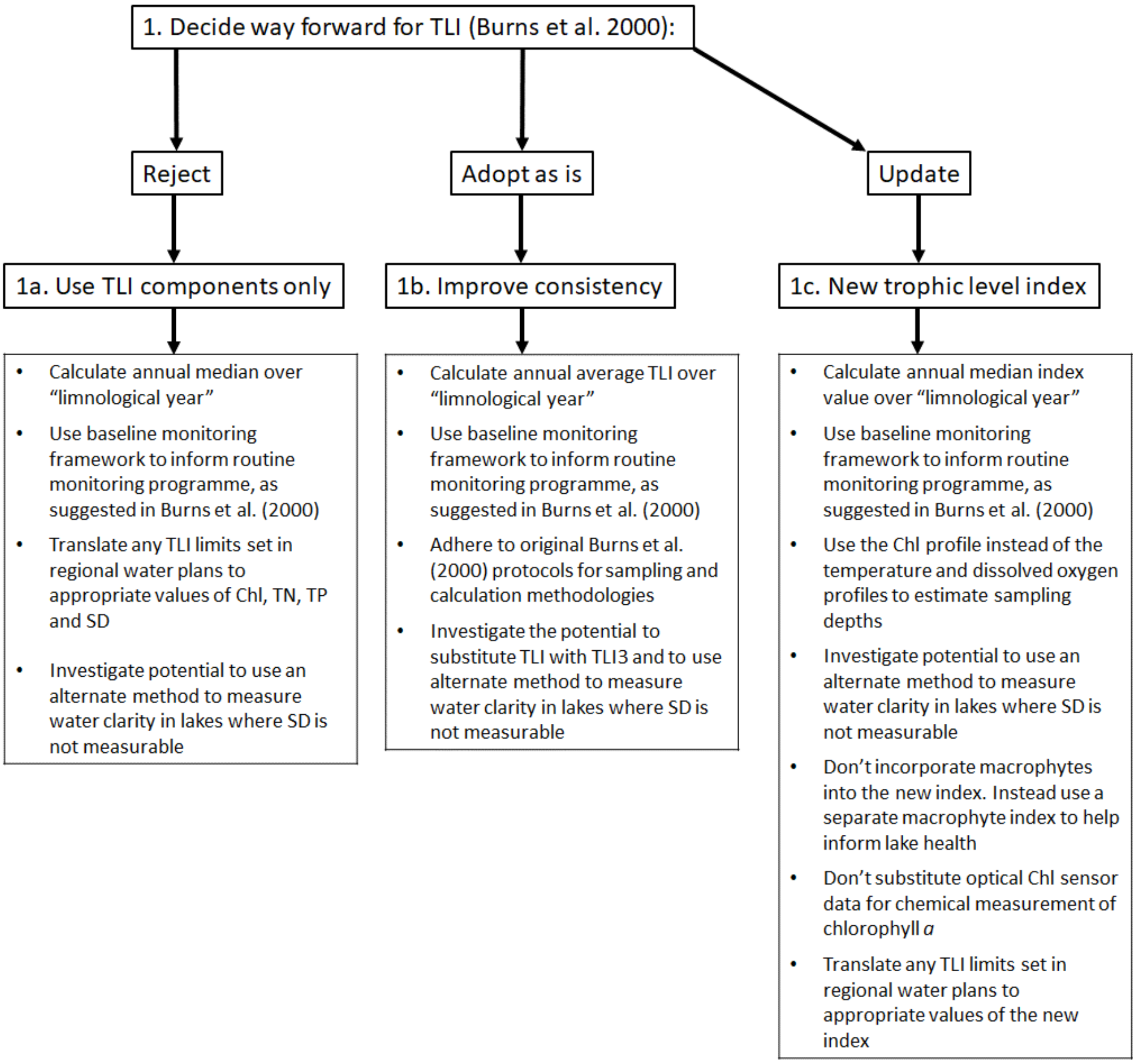

Figure 5. Recommendations for three possible ways forward for the TLI.

Perhaps lake management in New Zealand is at a TLI crossroad? This report attempts to stimulate thought and discussion regarding the TLI and to assist in setting policy directions regarding the important task of managing eutrophication in our lakes.

8. Acknowledgements

We thank the Regional Council Lakes Focus Group and particularly Deniz Özkundakci (Waikato Regional Council), Andy Hicks (Hawkes Bay Regional Council) for scoping and commissioning this report. Insightful comments on an earlier draft from Tina Bayer, James Dare, Andy Hicks, and Ebi Hussein improved the report.

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