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A change in phytoplankton community index with water quality improvement in Tolo Harbour, Hong Kong

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Abstract: Water quality in Tolo Harbour and Channel (Tolo) has been improved since 1998 after the diversion of sewage effluent. However, it remains poorly understood how nutrient loading reduction has impacted the phytoplankton community. To evaluate this, we applied a Phytoplankton Community Index PI(mp) to the 23-year data (1991-2013) at inner (TM4) and outer (TM8) sites in Tolo, with the former being more eutrophic than the latter. The results show that 1) the phytoplankton community changed with time after sewage diversion; 2) “diatoms and dinoflagellates” were better indicators of nutrient impact than “autotrophic/mixotrophic and heterotrophic dinoflagellates”; 3) the rate of recovery differed between the two stations, but both reached a similar state at a similar time; 4) seasonality of the phytoplankton community showed greater disturbance in spring than in other seasons. Our findings indicate that the nutrient reduction in the Tolo resulted in a positive change in the phytoplankton community.

Key words: Nutrients; Tolo Harbor and Channel; Phytoplankton Community Change; phytoplankton community index.
1. Introduction

Marine ecosystems are sensitive to anthropogenic influences that potentially negatively impact their “health”. An undesirable disturbance was defined by Tett et al. (2007) as “a perturbation of a marine ecosystem that appreciably degrades the health or threatens the sustainable human use of that ecosystem”. Resilience, the ability to recover from such disturbance (Mageau et al. 1995) is, therefore, a key indicator of the success of any water quality management undertaken in mitigation of undesirable anthropogenic disturbance. The input of nutrients is one of the most common anthropogenic disturbances of marine waters. Several multimetric indices have been developed for evaluating ecosystem health through community structure based on species composition or functional groups such as benthic macroinvertebrates (e.g. Böhmer et al., 2004), periphyton (e.g. Hill et al., 2000), riparian plants (e.g. Ferreira et al., 2005). Since it is phytoplankton that initially react to this nutrient increase (Katsiapi et al., 2016; Marchetto et al., 2009), the assessment of phytoplankton resilience is ideal for water quality assessment.

Phytoplankton consist of taxonomically diverse life-forms with different functional diversity which exhibit marked seasonal variability. The idea of marine phytoplanktonic life-forms can be traced back to Margalef (1978) who distinguished organisms by their relative $r$-$K$ strategies, which are related to nutrient environments. The concept has been further developed by a range of studies, for example, by Smayda and Reynolds (2003). Given the ecological importance of phytoplankton taxonomic and functional diversity, the Phytoplankton Community Index PI(mp) was developed as an indicator by Tett et al. (2008). This index defines the system status based on multidimensional state variables that describe a particular state of a complex system. It provides a means of evaluating temporal and spatial changes in phytoplankton communities, hence providing a means to examine their resilience.

Situated in the north-eastern corner of Hong Kong, Tolo is a land-locked bay consisting of two parts: a shallow inlet and a narrow tidal channel to another semi-enclosed Mirs Bay (Liu et al., 2014). Because of its bottlenecked coastline configuration, tidal flushing is poor, with average tidal current velocities of 0.04-0.08
m s\(^{-1}\) and 0.08 m s\(^{-1}\) in the Harbour and Channel of Tolo, respectively (Xu et al., 2004a), giving long water residence time of 28 days on average, with 38 and 14.4 days in the dry and the wet seasons respectively (Lee et al., 2006). With the rapid development in agriculture, industrialization and urbanization, heavy nutrient loading has caused a serious deterioration in this water body, especially eutrophication during 1980s (Holmes, 1988). High phytoplankton biomass “red tides” frequently occurred in this bay (Anderson et al., 2012; Yin, 2003), harming ecosystems through accumulated biomass of algae, shading of other phytoplankton and sea grass beds, as well as causing faunal death via decay and oxygen depletion (Anderson et al., 2012; Villacorte et al., 2015).

In order to reduce nutrient loading and improve the water quality of the Harbour, the Hong Kong Government implemented the Tolo Harbour Action Plan (THAP) in 1998. A series of actions were taken: livestock waste control, sewage treatment modification, legislation enforcement and landfill restoration (Holmes 1988; Tse and Jiao, 2008). While these initial measures significantly reduced anthropogenic nutrient pollution, it was the full implementation of a large-scale sewage diversion scheme that reduced effluent loading from about 2 x 10\(^5\) m\(^3\) d\(^{-1}\) in 1996 to nearly zero in 1998 (Gowen et al. 2012). A long-time series of high temporal resolution phytoplankton data has been collected in Tolo Harbour since 1991, spanning the period of pre- and post-nutrient remediation. Tolo Harbour is therefore a unique case study to investigate the response of a phytoplankton community to nutrient remediation.

One of the major successes of the THAP was the subsequent reduction in the number of red tides (e.g. EPD 2016; Tse and Jiao 2008). However, the effect of the THAP on the ecosystem health is not clear (Davidson et al. 2012, 2014) as red tides in Tolo Harbour might also be caused by other factors (Lai and Yin 2014; Yin 2003). To evaluate the effectiveness of the THAP, it is also important to assess the role of nutrient reduction on the phytoplankton community in Tolo. This study therefore presents the application of the PI(mp) to Tolo to assess the long-term recovery of the phytoplankton as a whole, following nutrient loading management. The objectives are to examine the change in both multi-year and seasonal PI(mp) using the important functional groups.
of diatoms, both autotrophic/mixotrophic dinoflagellates (“/” here denotes “and” or “plus”) and heterotrophic dinoflagellates.

2. Materials and Methods

2.1 Data source and Study sites

Total nitrogen (TN, dissolved inorganic and organic nitrogen) data (1986-2013) and phytoplankton abundance data (1991-2013) were obtained from the Hong Kong Environmental Protection Department (EPD). The data were based on bi-weekly or monthly water samples at two sites: TM4 in inner Tolo Harbour, and TM8 in Tolo Channel which opens to Mirs Bay (Fig. 1).

Samples for TN were collected at 3 depths: surface (1 m below the surface), bottom (1 m above the sediment), middle (a depth between the two). TN values from the three layers were combined to give a water column average and then averaged again for all sampling within one year to give the yearly water column average. The methods for chemical analysis of TN are presented in EPD (2016). A TN trend is sufficient for the representation of the eutrophication status in our study area.

Samples for phytoplankton abundance data were collected 1 m below the surface with 3-litre Nansen type bottles. Aliquots of 200 ml were preserved with Lugol’s iodine (Lam and Ho 1989) and subsamples of 10 ml were used to perform phytoplankton species counting with the Utermöhl microscopic technique (Lund et al., 1958). No sampling replicate was applied. For better comparison, we used log10 to plot the abundance data, with the actual abundance data for different life forms being summarised in Supplementary Table 1.
2.2 Phytoplankton Community Index

The PI(mp) method used here was first developed by Tett et al. (2008) for investigation of changes in the community structure of phytoplankton using a state-space perspective, compared with a “good quality” reference state. In contrast to previous applications of the index which was related to systems that were moving away from good environmental status, in this study we take data collected between 2001 and 2013, which is considered to be a less anthropogenically impacted state as water quality has been improved (EPD 2016), as our “good quality” reference.

This method uses sets of two-dimensional pairwise life-form plots to compare reference and alternative conditions, as shown schematically in Fig. 2. This reference domain (grey shading in Fig. 2) is defined by the abundance of a pair of “life-forms” and is produced using the “convex hull” method (Weisstein, 2006). It represents the instantaneous state of the particular pair of phytoplankton life forms within the water-body. If the community of life forms resists small disturbance, it will revert to the set
of states that constitute the reference condition. However, a large disturbance will
transform this system and lead to a deviated set of states, which is generally considered
to be an undesirable disturbance. Between small and large disturbance, there may be a
spectrum of community states.

Fig. 2. Schematic representation of the “doughnut”-shaped PI(mp) state space envelope.
(Redrawn from Tett et al., 2008)

In order to evaluate changes in community, we can plot reference data and sample
data in the same state space and calculate the percentage of sample points that remain
inside the reference envelope, using equation 1 (Tett, 2006):

\[
\text{PI(mp)} = 1 - \frac{\text{sample points outside the reference envelope}}{\text{total sample points}}
\]  - (1)

Therefore, a PI(mp) of 1 indicates no difference between the sample and the “good
quality” reference conditions, and a value of 0 implies a complete change. A binomial
or chi-squared (if the number of points exceeds 200) test is used to estimate the
probability of a particular number of points falling outside the reference envelope. A
maximum 5% of sample points outside the reference envelope is considered acceptable for a definition of good quality.

In this study, the chosen life forms were diatoms and dinoflagellates as these are two major components of the microplankton community. Most red tide species are dinoflagellates, and hence the diatom/dinoflagellate ratio is often related to the probability of red tides and large changes in the ratio is commonly viewed as an undesirable disturbance (Davidson et al. 2012). Life-form pair combinations are included in the Marine Strategy Framework Directives (MSFD) as a means to provide an overview of the structure and function of the planktonic component of marine ecosystems. The ratio of heterotrophic to mixotrophic is one of the Marine Strategy Framework Directives (MSFD) biodiversity, food web, eutrophication and seabed integrity indicators. While this index was developed for European waters, it clearly has global applicability in terms of water quality and hence we also compared heterotrophic dinoflagellates with autotrophic/mixotrophic dinoflagellates to evaluate how different components of the dinoflagellate community responded to nutrient remediation. The species that these groups are composed of are summarized as Supplementary Table 2.

In contrast to previous applications of the PI(mp) (Tett et al., 2008; Whyte et al., 2016), as we expect the condition of the Tolo Harbour water body to be improved rather than deteriorated, we used recent phytoplankton data from 2011 to 2013 as the “good quality” reference conditions, with data for 1991–1993 (pre-THAP), 1997-1999 (coincident with THAP), 2003-2005 (post THAP) and 2008-2010 (far post THAP) as comparison conditions. This approach provides examples of different levels (or stages) of anthropogenic perturbation for comparison with the present-day reference condition (Table 1). We also calculated changes in the PI(mp) for each single year from 1991 to 2013 as a time series plot (Table 1).

Given that seasonal changes occur in community composition, we also chose to evaluate how the THAP impacted the phytoplankton community in different seasons. As there was less data for these sub-sections in a year, we chose longer time windows for comparison, with 2006-2013 as the reference period, and 1991-1998 as the anthropogenically impacted period. This allowed sufficient data for analysis, avoiding
error caused by small sample size.

Finally, we sought to evaluate the impact of spatial changes. The result of Xu et al. (2004b) showed that the water quality in the more highly flushed Channel Subzone was much better than that in the Harbour Subzone. Therefore, to determine the phytoplankton response in different locations within the harbour, we compared site TM4 (inner Tolo Harbour) with site TM8 (outer Tolo Channel), as shown in Table 1.

Table 1. Comparison of experimental design to calculate PI(mp) for TM4 and TM8 for two life forms in different time scales with their reference and comparison periods.

<table>
<thead>
<tr>
<th>Life Forms</th>
<th>Time scales of Change</th>
<th>Reference Period</th>
<th>Comparison Period</th>
</tr>
</thead>
<tbody>
<tr>
<td>Annual</td>
<td>1991 - 2013</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

3. Results

3.1 Nutrients

The TN concentration change from 1986 to 2013 was plotted in Fig. 3. We only present TN here because the South China Sea is mainly N limited (Chen et al., 2004; Ning et al., 2004). TN remained at a relatively high level at both sites from 1986 to 1998, but started to decrease rapidly upon the completion of THAP in 1998. It then maintained at a stable low concentration of about 0.2 mg/l which is approximately one-third to one-half of the enriched concentration. Comparing the two stations: TN
concentration at TM4 is consistently higher than that of TM8 because TM4 is located in the inner bay with more restricted water exchange. However, the difference in TN concentration between sites became smaller with time, especially following 1998, due to the diversion reduced sewage loading. More information on nutrient concentration changes through that period is presented by Lie et al. (2011). Below, we discuss how phytoplankton community structure response to these changes in nutrient concentrations.

![Graph showing TN concentration at TM4 and TM8](image)

Fig. 3. Yearly water column averages of total nitrogen (TN) at TM4 (white circle) and TM8 (black circle) in Tolo Harbour and Channel during 1986-2013 calculated from monthly samples. Data from three depths (surface, middle and bottom) were averaged, then averaged again against time annually to give the yearly water column average.

3.2 Life form one: diatoms and dinoflagellates

3.2.1 Temporal change

The PI(mp) diagrams comparing the different historical time periods with the
present-day reference condition (Fig. 4) confirm an initial slight shift towards
dinoflagellates, followed by a trend back to diatoms, with relative rise and decrease in
their abundance. The temporal changes in calculated PI(mp) values are summarized in
Fig. 5a. The PI(mp) value at TM4 indicates an obvious (undesirable) disturbance to the
“balance of organisms” in this area from 1991 to 2010 with no clear improvement with
time. After this period, a rapid change occurred with the condition of the community
being significantly improved. In contrast, the ecosystem at TM8 experienced a gradual
improvement from 1991 onwards, although it did not reach the 95% level until 2011,
the same year as TM4 (Fig. 5a).
Fig. 4. Two-dimensional pairwise life-form envelope plots of diatom and dinoflagellate abundances (in form of log10) in state space for the comparison periods: 1991–1993 (b, g), 1997-1999 (c, h), 2003-2005 (d, i), 2008-2010 (e, j) and for the “good quality” reference period 2011–2013 (a, f) in TM4 (left column) and TM8 (right column).
Fig. 5. (a) Annual PI(mp) value of diatoms vs dinoflagellates abundance, and (b) annual PI(mp) value of auto/mixotrophic dinoflagellates vs heterotrophic dinoflagellates abundance with trend lines (solid straight line for TM4, dashed straight line for TM8). Data are from 1991 to 2013 at TM4 and TM8 with the reference period of 2011–2013. Correlation coefficient, r, for trend lines: 0.470 for diatoms vs dinoflagellates at TM4, 0.829 for diatoms vs dinoflagellates at TM8, 0.521 for auto/mixotrophic dinoflagellates vs heterotrophic dinoflagellates at TM4, and 0.706 for auto/mixotrophic dinoflagellates vs heterotrophic dinoflagellates at TM8.
3.2.2 Seasonal comparison

In spring, a higher dinoflagellate abundance was evident in the pre-THAP condition (1991-1998) compared to the modern reference state (2006-2013) at both TM4 and TM8, but lower diatom abundance only at TM8 (Fig. 6, 7). Both stations experienced a relative shift towards dinoflagellates when being anthropogenically impacted. TM4 (PI(mp)=0.27) experienced a more significant disturbance than TM8 (PI(mp)=0.35), but both sites were drastically improved in spring during 2006-2013 (Fig. 8(a)). In summer, both sites exhibited lower diatom abundance during the pre-THAP period, while neither showed conspicuous change in dinoflagellate abundance with time (Fig. 6, 7). In terms of the PI(mp) value, TM4 (PI(mp)=0.65) witnessed a smaller disturbance than TM8 (PI(mp)=0.46), but the THAP again significantly changed the community at both sites (Fig. 8(a)). Similar to summer, in autumn, both sites showed lower diatom abundance during the pre-THAP period, with no obvious change in dinoflagellate abundance (Fig. 6, 7). TM4 (PI(mp)=0.50) was somewhat less disturbed than TM8 (PI(mp)=0.40) (Fig. 8(a)). In winter, pre-THAP, we observed higher dinoflagellate abundance but fewer diatoms at TM4 and TM8, and hence a relative shift toward dinoflagellates in both cases under anthropogenic influence (Fig. 6, 7). Both sites were markedly disturbed, with PI(mp) of 0.14 at TM8 and 0.21 at TM4 (Fig. 8(a)).
Fig. 6. Two-dimensional pairwise life-form envelope plots of diatoms and dinoflagellates abundances (in form of log10) in state space using the seasonal binned data of 1991–1998 in relation with the “good quality” reference period 2006–2013 at TM4 (a) spring, (b) summer, (c) autumn, (d) winter.
Fig. 7. Two-dimensional pairwise life-form envelope plots of diatoms and dinoflagellates abundances (in form of log10) in state space using the seasonal binned data of 1991–1998 in relation with the “good quality” reference period 2006–2013 at TM8 (a) spring, (b) summer, (c) autumn, (d) winter.
Fig. 8. Bar chart of PI(mp) values from the seasonal life-form envelope plots of (a) diatoms vs dinoflagellates, and (b) auto/mixotrophic dinoflagellates vs heterotrophic dinoflagellates at TM4 and TM8.

3.3 Life form two: autotrophic/mixotrophic dinoflagellates and heterotrophic dinoflagellates

3.3.1 Temporal change

We see from Fig. 9, that both sites experienced a temporary shift to autotrophic/mixotrophic dinoflagellates (1991-2005) and then a relative return towards heterotrophic dinoflagellates (2008-2010), with no clear trend in heterotrophic dinoflagellates abundance, but a primary increase followed by a decline with time in autotrophic/mixotrophic dinoflagellates abundance. The PI(mp) value indicates that the ecosystem condition at TM4 experienced a significant disturbance during 1991-2010, but finally showed an apparent improvement (Fig. 5(b)). The water body at TM8 showed a relatively steady improvement of ecosystem health over the years of 1991-2013, achieving good status in 2007 slightly prior to TM4.
Fig. 9. Two-dimensional pairwise life-form envelope plots of auto/mixotrophic dinoflagellates and heterotrophic dinoflagellates abundances (in form of log10) in state space for the comparison periods: 1991–1993 (b, g), 1997-1999 (c, h), 2003-2005 (d, i), 2008-2010 (e, j) and for the “good quality” reference period 2011–2013 (a, f) in TM4 (left column) and TM8 (right column).
3.3.2 Seasonal change

In spring and winter, we observed higher autotrophic/mixotrophic dinoflagellate abundance both at TM4 and TM8 during the pre-THAP period, with little change in heterotrophic dinoflagellate abundance in spring although some increase was evident during winter. This indicated a shift to autotrophic/mixotrophic dinoflagellates under anthropogenic impact (Fig. 10, 11). The PI(mp) value at TM4 (Spring PI(mp)=0.31, Winter PI(mp)=0.42) displayed a more obvious disturbance than TM8 (Spring PI(mp)=0.52, Winter PI(mp)=0.46) (Fig. 8(b)). We can hence conclude that the water body ecosystem status was improved in winter and spring from 1991 to 2013. In summer and autumn, there were no marked trends at either site. The water body ecosystem in summer (TM4 PI(mp)=0.52, TM8 PI(mp)=0.45) was more impacted than in autumn (TM4 PI(mp)=0.88, TM8 PI(mp)=0.57), with the disturbance being less at TM4 in both seasons (Fig. 8(b)).
Fig. 10. Two-dimensional pairwise life-form envelope plots of auto/mixotrophic dinoflagellates and heterotrophic dinoflagellates abundances (in form of log10) in state space using the seasonal binned data of 1991–1998 in relation with the “good quality” reference period 2006–2013 at TM4 (a) spring, (b) summer, (c) autumn, (d) winter.
Fig. 11. Two-dimensional pairwise life-form envelope plots of auto/mixotrophic dinoflagellates and heterotrophic dinoflagellates abundances (in form of log10) in state space using the seasonal binned data of 1991–1998 in relation with the “good quality” reference period 2006–2013 at TM8 (a) spring, (b) summer, (c) autumn, (d) winter.
4. Discussion

Documenting the recovery of estuarine ecosystems is important in understanding the effectiveness of management measures taken to deal with pollution from industrial and urban development (García-Barcina et al., 2006). For example, Smith et al. (1981) explored changes in water chemistry and biomass in Kaneohe Bay, Hawaii due to sewage input and diversion during 1976 and 1979. This work revealed a physical and ecological response of an ecosystem to sewage loading change, but over a timescale so short that some variables did not reach a new steady state. Similarly, Rask et al. (1999) studied the response of coastal waters in Funen, Denmark to lowered nutrient discharges, revealing lower phytoplankton production and a positive impact on oxygen condition. Temporal changes in water quality parameters can be used as indicators in evaluating the success of pollution controls (Doering, 1996). However, one cannot draw conclusions on whether an ecosystem has been restored by temporal trends of single water quality indicator, as natural variability in the environment has too great an influence on the ecosystem states (de Jonge et al., 2002). For example, research on Tampa Bay, Florida (Johansson and Lewis, 1992) found that seagrass bed and an attached macroalga vegetated a shallow area because of reduction in nutrient loading, but such parameters are not suitable indicators for Tolo Harbour since the fine sediments there cannot support them well (Trott, 1972).

Phytoplankton taxonomic and functional diversity provides a multi-dimensional parameter set which is indicative of the state or recovery of an ecosystem. As discussed by Garmendia et al. (2012), phytoplankton are often used as indicators of nutrient loading to marine water with their assessment being included in a range of legislations (e.g. The US Clean Water Act, the EU Marine Strategy Framework Directive (MSFD) and water framework directive (WFD), and the Oslo–Paris (OSPAR) and Helsinki (HELCOM) conventions that explicitly address eutrophication). Gowen et al. (2012) analysed a number of examples of high biomass harmful algal blooms, concluding that in some, but not all cases, that anthropogenic nutrient loading to coastal waters was the causative factor. However, their review identified no studies that had evaluated the temporal and spatial recovery of the phytoplankton in a previously impacted ecosystem.
based on a long duration.

The water quality prior to and during 1990s in Tolo was heavily eutrophicated and has been steadily improved since the THAP implementation in 1998 (EPD 2016; Tse and Jiao, 2008; Xu et al., 2004b). Hence, we defined the period 2011-2013 as the “good quality” reference condition and made backward comparison to show the changes in the ecosystem status, which differs from previous studies (Tett et al., 2008; Whyte et al., 2016).

Comparing the life forms diatoms vs dinoflagellates, an interim shift towards dinoflagellate dominance, often an indicator of eutrophication (Anderson et al. 2008) during high nutrient loading, followed by a return towards diatoms (Fig. 5) is in accordance with the observed increase and subsequent decrease in (dinoflagellate dominated) red tide frequency in Tolo Harbour.

Temporal change in the structure of the phytoplankton community took place over years and occurred at different rates for different life forms and at different sites, as is shown in the time series of PI(mp) values in Fig. 5, although nutrient loading to Tolo Harbour had decreased to low levels by 1998. An abrupt increase in the PI(mp) index was evident at TM4 in 2011 and that was in contrast with TM8 where the PI(mp) increased gradually with time. Harrison et al. (2012) and Davidson et al. (2014) compared red tide events in Tolo Harbour and another site in Hong Kong, Victoria Harbour, and emphasized the role of hydrodynamics in structuring the response of water bodies to nutrient enrichment, as Tolo Harbour is almost permanently stratified and hence more susceptible to red tide events than the weakly seasonally stratified Victoria harbour that experiences much shorter water residence time. Similarly, as TM8 is more seaward, and hence more rapidly flushed than the restricted inner TM4, PI(mp) at TM8 initially recovered more rapidly for both the diatom vs dinoflagellate and the autotrophic/mixotrophic vs heterotrophic life form comparisons than TM4. It is interesting that the rapid improvement at TM4 post 2010 resulted in both sites achieving the “good quality” condition at the same time although different trajectories were followed.

In the case of autotrophic/mixotrophic dinoflagellates vs heterotrophic
dinoflagellates, the PI(mp) index recovered more rapidly than the diatom vs dinoflagellate index (Fig. 5(b)). While this might be expected on the basis that diatoms and dinoflagellates are different life forms that tend to be temporally and/or spatially segregated (Smayda and Reynolds 2001), the results are consistent with close coupling between heterotrophic dinoflagellates and their autotrophic prey.

When considering the seasonality of diatoms vs dinoflagellates, the greatest impact indicated by the lowest PI(mp) values was evident in winter and spring at both TM4 and TM8. Winter and spring in Hong Kong are dominated by temperatures similar with those in temperate waters in summer and by high monsoonal winds predominantly from the NE which can push water inwards. As wind conditions favour long residence time in Tolo Harbour, temperature favours the dominance of dinoflagellates, which is what we can see in the seasonal PI(mp) plots (Fig. 6, 7), with the trend being more significant, particularly at the restricted TM4 site, during the period when nutrient loading was high.

For autotrophic/mixotrophic dinoflagellates vs heterotrophic dinoflagellates, there was a less clear seasonal pattern, with generally similar levels of disturbance in all seasons. We have observed a similar response of autotrophic and heterotrophic dinoflagellates to nutrient manipulation elsewhere (Davidson et al. 2007). Results from Tolo Harbour are therefore consistent with other observations. Although these groups exhibit fundamentally different life strategies in terms of their means of obtaining nutrition, the close coupling between heterotrophs and their autotrophic prey results in the two groups being similarly impacted by the same anthropogenic nutrient disturbance.

5. Conclusions

In summary, the PI(mp) index indicates that nutrient loading caused a significant disturbance in the ratio of both diatoms vs dinoflagellates and autotrophic/mixotrophic vs heterotrophic dinoflagellates, but this disturbance was greater for the former. Following the completion of the nutrient reduction management plan (THAP), the ecosystem health associated with the phytoplankton community in Tolo has been significantly improved, but the recovery was not instantaneous, taking approximately 13 years for diatoms vs dinoflagellates and 8 years for autotrophic/mixotrophic vs...
heterotrophic dinoflagellates. While the recovery in the autotrophic/mixotrophic vs heterotrophic dinoflagellates index occurred at a similar rate at both sites, the pattern was different for diatoms vs dinoflagellates with the less restricted TM8 site beginning to recover earlier, but doing so more gradually than TM4 that exhibited a more delayed step change. The impact of nutrient loading varied with seasons, with the greatest impact for diatoms vs dinoflagellates in winter and spring. Consistent with the effect of the THAP on red tide frequency that has been decreasing since 1998, our finding indicates that the planktonic ecosystem of Tolo has been significantly changed and improved, since the completion of nutrient loading management in 1998. This study demonstrates that the phytoplankton community index developed by Tett et al. (2008) supports the improvement evidence of water quality and can be a useful indicator to assess the ecosystem health status.

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