Trophic state index validation based on the phytoplankton functional group approach in Amazon floodplain lakes

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ABSTRACT

Many trophic indices have been constructed for temperate aquatic environments, but few have proved reliable for tropical environments. Indices constructed on the basis of nutrients may not be effective for describing the trophic state because the fractions of nitrogen and phosphorus differ in their potential to predict the nutrient limitation in tropical aquatic environments. We developed an index based on the phytoplankton Reynolds functional groups for an Amazonian floodplain lake from samples collected during 2 contrasting hydrological periods (rising and flushing) and compared it with the index initially proposed by Carlson in 1977 and further adapted to tropical environments by Toledo in 1990. The functional group and Carlson indices matched only 37% and 56% of the sample units collected during the rising and flushing periods, respectively. Our study confirms the difficulty of assessing trophic states using only phosphorus and chlorophyll \( a \) in tropical floodplain lakes. In this environment (1) nitrogen may significantly limit phytoplankton growth and (2) complex phytoplankton–nutrient relationships occur during the hydrological cycle that cannot be accounted for through a simple phosphorus and chlorophyll \( a \) formula.

Introduction

Freshwater wetlands ensure fundamental services such as flood mitigation, groundwater recharge, water depuration, nutrient and sediment retention, and fish habitat availability (Costanza et al. 1997, Mitsch and Gosselink 2007) and present high levels of biodiversity (Dudgeon et al. 2006). Wetlands declined significantly worldwide in the last century (Gardner et al. 2015) and remain under increasing anthropic pressure (Vörösmarty et al. 2010). In the coming years, Amazonian floodplains will be increasingly threatened by development projects such as dams and river navigation networks, hydrocarbon exploration, and extension of the agricultural frontier (Ferreira et al. 2014, Forsberg et al. 2017, Anderson et al. 2018). These changes may significantly alter water quality, especially suspended solids and nutrient loads, in part by disrupting natural river regimes, which are the principal driver of floodplain characteristics (Junk et al. 2013, Forsberg et al. 2017). Known as “flood pulse,” the seasonal hydrological variation ensures water and nutrient renewal enhances matter exchange between aquatic and terrestrial ecosystems, and creates conducive conditions for high productivity and levels of biodiversity. The 4 common hydrological regimes are 2 extreme periods (low water and high water) and 2 intermediate periods (rising and flushing), during which the floodplain lake characteristics differ (Junk et al. 1989). During low-water periods, floodplain lakes present a small volume of highly mixed and turbid water and remain connected to the mainstream (Bourgoïn et al. 2007). Rising water levels are ensured by multiple sources of water (runoff from the local drainage basin, direct precipitation, groundwater, and mainstream). The amount of water varies interannually and differs according to floodplain morphology (Lesack and Melack 1995, Bonnet et al. 2008, 2017). Nutrient and water renewal and decay of submerged herbaceous vegetation promote a first peak of primary production...
during the rising period (Junk et al. 1989, Bonnet et al. 2016, Amaral et al. 2018). Higher depth, shorter water residence time, and dilution limit primary production during the high-water period, whereas a second peak of primary production generally occurs during the flushing period (Junk et al. 1989, Bonnet et al. 2016). Extensive cattle ranching and increasing human settlements in the floodplain (Peres et al. 2018) can contribute to nutrient enrichment or changes in their relative proportions and accelerate the eutrophication process (Kraus et al. 2019), which is a serious and increasing problem worldwide (Dodds and Smith 2016).

Primary productivity of phytoplankton can be limited by nutrient availability (Schindler 2012, Paerl et al. 2016). Recent studies in Amazonian lakes (Lopes et al. 2011, Bozelli et al. 2015) and in other Brazilian floodplain systems (Loverde-Oliveira et al. 2012, Padial et al. 2014, Huszar et al. 2015, Machado et al. 2015) have shown that nutrient availability is the most important driver of diversity and in structuring phytoplankton communities. Over the past several decades, many water quality parameter-based or phytoplankton-based indices have been constructed using datasets from temperate regions, yet few have proved reliable for tropical environments. Toledo Júnior (1990) adapted the Carlson (1977) water quality formula for tropical environments, but fractions of nitrogen (N) and phosphorus (P) differ in their potential to predict nutrient limitation in lakes. Total phosphorus (TP) and total soluble P are equally accurate indicators, whereas dissolved inorganic N has been shown to be a better predictor of phytoplankton biomass than total nitrogen (TN) or total dissolved N (Lewis 2011).

Ecologists have proposed approaches based on Reynolds functional groups (RFG) of phytoplankton assemblages based on latitude, trophic states, and morphometric characteristics (Reynolds et al. 2002) to identify simple patterns and predict biological dynamics in freshwaters (Kraus et al. 2018). Many works have updated the Reynolds functional groups. For example, Padišák et al. (2009) consolidated the Reynolds classification and included new groups, and Kruk et al. (2010) proposed a classification based on species morphology. Recently, the latter classification was shown to be most suitable for floodplain environment (Lobo et al. 2018). Padišák et al. (2006) developed the index Q, relating phytoplankton assemblage and water quality to lake ecological status. The Q index was successfully applied in various types of environments, for example in a deep Mediterranean reservoir (Becker et al. 2010), deep lakes in southern Alps (Salmaso et al. 2006), and in deep reservoirs in Brazil (Silva and Costa 2015). These studies showed that both ecological indices and the phytoplankton functional approach bring together multiple parameters or characteristics into an indicator capable of representing some type of environmental status. The development of a tool that integrates the 2 approaches could describe the spatial variations of the phytoplankton community in the environment and evaluate the state of this environment because most of the nutrients are typically locked up in phytoplankton cells (Kosten et al. 2012). Following this premise, and to contribute to the international effort for the preservation of these emblematic wetlands, we propose an index to monitor environmental status in floodplain lakes that could support the definition of adapted environmental management policies. In the present study, we developed an index based on the phytoplankton Reynolds functional group approach (denoted TSI_{PB}) to identify spatiotemporal patterns of system dynamics in Amazonian floodplains. We compared it with the well-known categories of trophic states based on nutrients and chlorophyll a (Chl-a). Because the floodplain characteristics vary widely between hydrological phases, the TSI_{PB} was built and validated from data collected during distinct periods of the hydrological cycle.

**Material and method**

**Study site**

We collected samples from the Curuáí floodplain located between 01°50′–02°15′S and 55°00′–56°05′W along the right margin of the Amazon River, south of Óbidos city in Pará state, Brazil (Fig. 1). The Curuáí floodplain is a large system composed of several temporally interconnected lakes. The maximum flood extent delimited by lands with altitude >11.5 m a.s.l. is ~2500 km², and it drains several small basins that extend 1500 km². The lake system is linked with the mainstem by several channels, but only the easternmost channel remains permanently connected (Bonnet et al. 2008). The floodplain is seasonally flooded by water from the Amazon mainstream and also from local drainage basins, seepage, and local precipitation. The high-water period occurs in June and low-water period in November. The large seasonal water level variation amplitude (average ~6 m) combined with flat relief induces a large variation in flood extent between low- and high-water periods. In the Curuáí floodplain, the seasonal water level variation results in a 4-fold variation (from ~500 to 2500 km²) of the open-water extent (Bonnet et al. 2008) along the water year. The river water, rich in inorganic suspended material and nutrients (Sioli 1984, Moquet et al. 2011, Park and Latrubesse 2015), contrasts with the water quality of the other water sources, which are poor in nutrients and rich in organic matter (Bonnet et al. 2016, 2017).
Several small communities with <100 families are spread across adjacent uplands and in the tidal regions. A few communities such as Curuá, Piraquara, and Tabatinga do Salé have a higher density of inhabitants. The population density in the region significantly increased in the last decade (IBGE 2011), augmenting urban cover in the floodplain and adjacent uplands (Peres et al. 2018).

**Water quality, biological sampling, and laboratory analysis**

The sampling station distribution was designed to detect heterogeneities among locations and to gather enough replicates for the subsequent analyses (Fig. 1). Stations were distributed among 7 lakes of the Curuá floodplain. During water sample collection we also reported water level variation in the river (Fig. 2). Sets of 26 and 25 samples were collected in March 2013 (rising period) and September 2013 (flushing period), respectively.

Samples for water quality analysis were collected at the subsurface with a Van Dorn sampler and stored in pre-cleaned plastic bottles with 10% HCl and rinsed with Milli-Q water. Three aliquots of 250 mL were immediately passed through Whatman GF/F glass microfiber filters (porosity 0.7 μm) using a low-pressure vacuum pump; the membranes were wrapped in aluminum foil and frozen for subsequent Chl-α analyses in the laboratory. In addition, an aliquot of 350 mL was passed through pre-dried and pre-weighed acetate cellulose membranes (pore size 0.45 μm) to obtain total suspended matter concentration. In the laboratory, Chl-α filters were extracted with buffered acetone (90% acetone + 10% saturated magnesium carbonate; Jespersen and Christoffersen 1987), and the extracts were held for 24 h in the refrigerator before colorimetric determination (APHA 2012). TP was measured on the filtered water by colorimetry (Mackereth et al. 1979), and TN was obtained with a non-dispersive infrared (NDIR) Shimadzu sensor coupled to a Shimadzu carbon analyzer (Shimadzu, Kyoto, Japan).

Quantitative phytoplankton samples were collected at the subsurface and placed directly into dark bottles and preserved with 1% Lugol’s solution. Samples were also concentrated through a plankton net with a mesh diameter of 20 μm to aid taxonomic identification when necessary. Phytoplankton was counted at 400× using an inverted microscope Carl Zeiss, Axioscop 25 model (Thornwood, NY, USA) according to Utermöhl (1958).
For numerous species, at least 100 individuals were counted according to Lund et al. (1958), and phytoplankton density was expressed as individual units (ind mL$^{-1}$). We used the Van Den Hoek et al. (1995) system to classify the phytoplankton. The algal biovolume was calculated by multiplying the abundance of each species by cell volume average (in mm$^3$ L$^{-1}$) according to Hillebrand et al. (1999) and Fonseca et al. (2014). The biovolume was converted into biomass assuming 1 mm$^3$ L$^{-1}$ = 1 mg L$^{-1}$ (Wetzel and Likens 2000).

**Trophic state indices based on water quality parameters and functional group approach**

We computed the well-known trophic state index based on water quality parameters (denoted TSI$_T$ hereafter), initially defined by Carlson (1977) and further adapted to tropical environments (Toledo Júnior 1990).

When the sediment load is high, the index calculated from Secchi depth should not be used (Lamparelli 2004). TSI$_T$ is thus obtained as the arithmetic average of TSI$_T$(Chl-$a$) and TSI$_T$(TP) given by equations 1 and 2:

\[
\text{TSI}_T(\text{Chl}-a) = 10 \left[ 6 - \left( \frac{2.04 - 0.695 \ln(\text{Chl}-a)}{\ln 2} \right) \right] \tag{1}
\]

\[
\text{TSI}_T(\text{TP}) = 106 - \frac{\ln(80.32 / \text{TP})}{\ln 2} \tag{2}
\]

where Chl-$a$ and TP are concentrations in $\mu$g L$^{-1}$. The limits of the trophic categories defined from TSI$_T$ values by Toledo Júnior (1990) are included in Supplemental Material 1.

The TSI$_{PB}$ index was computed from the RFG classification, further consolidated by Padisák et al. (2009) from each sample. The species with biomass contributing to at least 5% of the biomass in the total sample were selected to identify RFGs (or codon) following Kruk et al. (2002) methodology. Once codons were identified in each sampling unit, we defined trophic categories ($F_{cl}$) as follows: ultraoligotrophic (UO) $\leq$ 1.5; 1.5 $<$ oligotrophic (O) $\leq$ 2; 2 $<$ oligo-mesotrophic (OM) $\leq$ 2.5; 2.5 $<$ mesotrophic (M) $\leq$ 3; 3 $<$ meso-eutrophic (ME) $\leq$ 3.5; 3.5 $<$ eutrophic (E) $\leq$ 4; 4 $<$ eutro-hypertrophic (EH) $\leq$ 4.5; and 4.5 $<$ hypereutrophic (H) $\leq$ 5. Finally, to compute TSI$_{PB}$, the biomass percentage of each codon was multiplied by the trophic category it indicates. However, some codons were found to be independent from the trophic state, in which case the percentage of the remaining codons was adjusted so that the sum of their respective percentage remained 100% (equation 3):

\[
\text{TSI}_{PB} = \sum_{i=1}^{n} \frac{b_i}{B_T} F_{cl}^i, \tag{3}
\]

where $b_i$ is the biomass of $i$-th RFG, $B_T$ is the sum of biomass at the station, $F_{cl}^i$ is the trophic category indicated by the $i$-th functional group.

**Comparison of trophic state indices based on water quality parameters and functional group approach**

We compared the trophic categories indicated by each index by defining a concordance factor (denoted CF hereafter). The CF value was 1 if both indices gave the
same trophic category; 0.75 if categories differed by one state (for example if TSI_T indicated a eutrophic state and TSI_PB indicated a meso-eutrophic state); 0.5 if categories differed by 2 states (for example if TSI_T indicated a eutrophic state and TSI_PB indicated a mesotrophic state); and 0 otherwise.

Multilinear model analysis was undertaken to verify that TSI_PB correctly reflected the trophic state of the environment, even if it was not directly linked to the variation of a specific nutrient.

Results

Water quality-based trophic state index (TSI_T)

We report all water quality parameters (minimum, maximum, and median values) from this study (Table 1). During both data collection periods, the floodplain lakes were ~4 m depth and turbid, with a Secchi depth <0.5 m. TP was relatively high during the rising and flushing periods. The N to P mass ratio remained low, with a mean value ranging from 4.3 (standard deviation [SD] 2.7) to 6.6 (SD 5.7) during the rising and flushing periods, respectively. The Chl-a distribution was highly variable among stations, with values ~4 times higher during the flushing period (Table 1).

During the rising period, TSI_T values were classified as oligotrophic to hypereutrophic (Table 2). Two stations were classified as oligotrophic: S49 located in a small stream draining the local basin, and S2 located near a margin with shrub vegetation. One station, S10, was classified as hypereutrophic, influenced by a small stream draining communities and pastures. The remaining stations were classified as eutrophic (16 stations) or mesotrophic (7 stations).

During the flushing period, TSI_T values in the floodplain ranged from mesotrophic to eutrophic (Table 2). Only one station was categorized as oligotrophic (S31) and 2 stations were considered mesotrophic (S9 and S30). All the others were in the eutrophic category.

Functional groups and phytoplankton based-index (TSI_PB)

Phytoplankton diversity was high with 189 and 198 identified taxa during the rising and flushing periods, respectively (Lobo et al. 2018). The median biomass obtained from the sampling units was 1.0 (SD 0.7) and 4.0 (SD 5.3) mg L^{-1} during the rising and flushing periods, respectively (Supplemental Material 2).

During the rising period, we identified 18 RFGs listed in codons C, D, E, F, G, H1, J, Lo, M, MP, N, P, S1, Td, W1, W2, X2, and Y, divided into 44 taxa; the median biomass from sampling units was 1.2 (SD 0.6) mg L^{-1}. Species, codons, and their biomasses were reported as percentages of the sample total biomass (Supplemental Material 3). Among the identified codons, some can be found regardless of trophic state, and others, like X3 or P, are encountered only in specific trophic conditions. The latter were considered in computing TSI_PB using the F_d value attributed to the trophic categories (Table 3).

During the flushing period, we identified the codons F, G, H1, M, MP, N, P, W1, and W2, spread over 32 taxa. While observing a reduction of almost 50% in the RFG number, species from the cyanobacteria class were dominant, 96.7% of the species belonged to codons H1, M, and MP. Codon H1 was recorded in 100% of sample units and was the dominant group in most samples. The codons M and MP were recorded in 56% and 24% of the samples, respectively (Supplemental Material 3). H1 and P are representative of eutrophic state and M is representative of eutrophic-hypereutrophic state (EH). According to TSI_PB, all the stations were eutrophic (E) or in an intermediate status between eutrophic and hypertrophic states (E–H; Table 2).

Comparison between TSI_T and TSI_PB

During the rising period, the 2 indices TSI_T and TSI_PB perfectly matched in only 9 of 24 cases (37% of the

Table 1. Maximum (Max), minimum (Min), median (Med), and standard deviation (SD) of the water quality variables during rising and flushing periods.

| Variables     | Rising period (n = 26) | Flushing period (n = 25) |
|---------------|------------------------|--------------------------|
|               | Min | Max | Med | SD  | Min | Max | Med | SD  |
| Depth (m)     | 1.7 | 4.7 | 4.0 | 1.5 | 2.5 | 5.1 | 3.7 | 0.7 |
| Secchi (m)    | 0.4 | 1.7 | 0.4 | 0.3 | 0.3 | 1.5 | 0.6 | 0.3 |
| pH            | 6.6 | 7.7 | 7.2 | 0.3 | 6.1 | 9.42| 7.14| 0.9 |
| TSM (mg L^{-1}) | 10.6 | 47.6 | 35.6 | 10.5 | 1.5 | 93.5 | 30.8 | 22.8 |
| TP (μg L^{-1})  | 22.1 | 186.4 | 96.3 | 39.7 | 7.1 | 111.3 | 41.3 | 26.2 |
| TN (μg L^{-1})  | 225.4 | 629.6 | 363.2 | 90.7 | 187.1 | 570 | 276.9 | 104.3 |
| NH_4-N (μg L^{-1}) | — | 249.9 | 89.3 | 58.2 | 10 | 235.2 | 94.5 | 73.3 |
| Chl-a (μg L^{-1}) | 0.4 | 187.9 | 24.2 | 38.5 | 6 | 183 | 17.5 | 50.3 |

n = number of samples; TSM = total suspended matter; TP = total phosphorus; TN = total nitrogen; TIN = total inorganic nitrogen; Chl-a = chlorophyll a; NH_4-N = ammonium-nitrogen.
stations) and differed by more than one category in 29% of the stations (Table 2). In 11 cases, TSI_pb indicated a higher trophic state than TSI_T. Considering TSI_T based only on Chl-α or only on TP concentration did not improve index matching (Table 2). According to TSI_pb, the trophic state in of most of the sampling stations was eutrophic during the rising period. The scores were relatively better during the flushing period; both indices matched in 56% and differed by one category in 32% of the stations, respectively. During this period, Dolichospermum spp. (codon H1), found in eutrophic conditions, represented >80% of the biomass at most of the stations. In addition, Microcystis aeruginosa (codon M) was present at all stations from the Lago Grande and in some of the lakes, including Grande Poção and Piraquara. Euglena spp. (codon W1) was encountered at station S11, which presented the highest TSI_T of the floodplain during this period.

The TSI_T multilinear model was well fitted (adjR²) and significant (p < 0.05) only for Chl-α and P, demonstrating their direct relationship with these variables, as expected (Table 4). By contrast, the TSI_pb multilinear model exhibited no direct relationship with any of the nutrients. It was significantly associated with total suspended matter (Table 4), but the relationship remained weak and negative.

**Discussion**

**TSI_T and TSI_pb comparison**

Our results indicated a relatively poor agreement between the TSI_T index based on TP and Chl-α data and the TSI_pb index built from functional groups. We could not improve the indices match, even when considering TSI_T values based only on TP or only on Chl-α data. These results confirm the difficulty in assessing trophic conditions from only a small set of water quality parameters.
parameters, as noted in previous studies (Marchetto et al. 2009, Abell et al. 2010), and therefore joint analyses of biological compartments are advisable (Coelho et al. 2007). A complex interplay among hydrology, environment, and organisms through the food web network drives the phytoplankton community in terms of biomass and species in floodplain lakes, which is better reflected in the TSI(pb) index than in the TSI(t) index in the Amazonian floodplain lakes. The TSI(t) index implicitly assumes that TP is the most important driver of phytoplankton biomass and that its combination with Chl-α is enough to predict trophic conditions.

Regardless of the considered hydrological periods, the correlation between TP and Chl-α is generally poor in floodplain lakes; thus, inferring trophic state from these variables is complicated. In turbid waters, such as those encountered in this study, a large amount of the P is bound to suspended inorganic sediment and is only partly available for biological uptake (Engie and Sarnelle 1990). That TP does not correctly reflect the bioavailable P may partly explain why TSI(t)(P) is almost systematically greater than TSI(t)(Chl-α) during the rising period (Table 2). However, TSI(t)(P) is lower than TSI(t)(Chl-α) in almost all the sampling units during the flushing period, whereas turbidity is comparable between the 2 periods. As noted by Carignan and Vaithiyathan (1999), a small decrease in pH along the hydrological cycle may favor the release of a portion of the inorganic-bound P, making it available for biological uptake. Therefore, despite a smaller TP concentration during the flushing period, the bioavailable P could be higher. But according to Kraus et al. (2019), the proportion of bioavailable P (PO4) during the flushing period is even smaller than during the rising period. Our results during this period are therefore more likely explained by the fact that P is not the main limiting factor of phytoplankton growth during this period.

In Amazonian floodplain lakes, as in eutrophic environments, relatively low N:P may indicate that N is a limiting factor for phytoplankton growth, which would favor the emergence of atmospheric N2-fixing species (Visser et al. 2016). As shown in our study, during the flushing period most of the species belonged to the H1 functional group, with species such as Dolichospermum spp. known for their N2-fixation ability. Thus, indices constructed on the basis of nutrient values are intrinsically linked to 1 or 2 variable predictors and may not be effective in describing the prevailing ecological conditions with respect to the trophic state of floodplain lake environments. Because it is not tied to any specific predictors but instead reflects hydro-environmental-phytoplankton interactions, TSI(pb) more effectively reflects the trophic state of the environment.

We found a significant relationship between the TSI(pb) and total suspended matter concentration, but inferring causality is difficult. Controlled experiments and field data suggest that light availability might affect the competitive balance among a large group of shade-tolerant species of phytoplankton (Smith 1986, Scheffer et al. 1997). Furthermore, cyanobacteria proportion can be influenced by a high shade index (Kosten 2012). Even so, our approach seems to be effective. Even if based on functional groups and not on species, the TSI(pb) captures the differences in phytoplankton composition, principally in the cyanobacteria community. Distinct groups of cyanobacteria sensitive to light/shade differences are classified into different functional groups that indicate different types of trophic state. Moreover, the model showed that total suspended matter influence is weak and represents a small portion of TSI(pb), indicating that this influence can be due to the number of sites analyzed, and further applications can exhibit no influence.

### Trophic states in the floodplain lakes

TSI(pb) values indicated eutrophic conditions in most of the sampling stations during the rising and flushing periods and a transition toward higher trophic state between the rising to flushing period (Table 2). Affonso et al. (2011) previously reported an increase in trophic state along the hydrological year for the Curuai floodplain, but their TSI(t) computation led to higher trophic state
than in our study. The increase of the trophic state along the hydrological year was also suggested in Kraus et al. (2019) and reflects the successive influence of river incursion and biogeochemical processes in the floodplain along the hydrological year. During the rising period, the river incursion across the floodplain is the major process that brings nutrients and organic matter into the floodplain (Junk et al. 1989). As discussed, even if only part of these nutrients is available for biological uptake, the inputs satisfy the requirement to sustain a first peak of phytoplankton growth (Moreira-Turcq et al. 2013, Bonnet et al. 2016). Processes that occur in the floodplain participate in maintaining nutrients required for a second peak of phytoplankton during the flushing period (Kraus et al. 2019). Especially, nutrient transfer from the sediment bulk into the water column is ensured by herbaceous plants that pump nutrients from sediments and release them into the water column during their decay (Junk et al. 2010, Bonnet et al. 2016).

Conclusion

The TSI\textsubscript{T} computation based on the arithmetic average of Chl-a and TP data provided the best agreement with the TSI\textsubscript{P}, yet the agreement between both indices remained relatively weak, perfectly fitting in only 37% and 56% of the stations during the rising and flushing periods, respectively. The phytoplankton-based index more adequately described the environment because it captured the complex interplay between hydrological, environmental, and organismal components. Moreover, one source of discrepancy in floodplain lakes could be the phytoplankton growth limitation by N while TSI\textsubscript{T} only assumed P limitation. Both indices indicated an increase of the trophic state from the rising to flushing period, suggesting that processes occurring in the floodplains significantly participate in sustaining nutrient requirements for primary production. Our analysis also highlighted some locations with hypereutrophic state in the proximity of margins where organic matter and nutrient enrichment were expected from cattle ranching and domestic effluent. Given the high number and complexity of functional groups in the Curuaí floodplain, we were unable to identify ranges of TSI\textsubscript{T} values for specific functional groups, but our analysis showed that TSI\textsubscript{T} values >55 in the floodplain lake were mostly associated with specific codon H1 presence and, to a lesser extent, can also indicate the presence of the codons M and MP.

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Disclosure statement

No potential conflict of interest was reported by the author(s).

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