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Testate amoebae as a potential tracer of organic matter dislodged from peat extraction areas

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In the boreal zone, surface waters mostly receive external organic matter (OM) from surrounding peatlands. The lake's biological communities may respond to changes in OM inputs caused by anthropogenic activities in the catchment. Testate amoebae (TA) possess an outer shell that preserves well in lake sediments and are commonly used in paleo-environmental studies. Additionally, they fall into the size range of particles transported from peatlands to lakes, making them potential particle tracers. Here, we compared TA communities of current and pre-peat extraction sediments from lakes receiving OM only from peat extraction areas (impact lakes) with lakes receiving OM from peatlands under other uses (control lakes). We found no differences between control and impact lakes, neither between current and pre-peat extraction. In conclusion, there is either no significant increase in the amount of organic matter discharge from peatland areas, or dislodged testate amoeba are retained in areas upstream of the lakes.

Introduction

In peatlands, net primary production exceeds the rate of decomposition, leading to accumulation of organic matter as peat. Hence, the role of peatlands is highly important in the carbon cycle, as net sinks of atmospheric CO_2 (Gorham 1995). Similarly to other ecosystems, peatlands have been subject to significant anthropogenic impacts for different purposes, such as drainage for forestry and agriculture (Coulson *et al*. 1990) and commercial peat harvesting (Kløve 1998, Pavey *et al*. 2007). In Finland, the original area covered by peatlands was ca. 104 000 km^2, equalling 30% of the total land area and of which, currently ca. 59 000 km^2 have been drained mostly for forestry and agriculture (Turunen 2008), and ca. 650 km^2 for active peat extraction (Väyrynen 2008). Peat erosion and its subsequent transport from the extraction field downstream to water systems is one of the main environmental problems of peat extraction (Kløve 1998). The eroded peat is transported to rivers and lakes where it accumulates causing eutrophication and increased sedimentation and oxygen consumption (Sallantaus 1984, Nieminen *et al*. 2010).

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In many boreal surface waters, the major source of organic matter are peatlands in their catchment areas (Clark *et al*. 2008). Increased leaching of organic matter due to anthropogenic activities influence nutrient cycling and photochemical processes (Stevenson 1994), mainly due to increases in colour, dissolved organic carbon concentrations (DOC), suspended solids (Kondelin 2006) and increased concentration of total nitrogen and phosphorus (Nieminen *et al*. 2017). In turn, these are expected to affect the biological communities of the downstream water bodies (e.g., Laine *et al*. 1995, Solomon *et al*. 2016). However, while environmental changes due to peat drainage for forestry and peat extraction in surface waters are well documented (e.g., Kauppila *et al*. 2016), their effects on the biota of recipient lakes are not.

Effective and accurate assessment of lake ecological status require long-term environmental data to define natural or reference conditions. Unfortunately, reliably measured data prior to anthropogenic impacts are rarely available (Smol 1992). Fortunately, in lakes, paleolimnology can be applied cost-effectively to assess, monitor and evaluate ecosystem changes caused by different phenomena such as climate change, sedimentation processes, deforestation, eutrophication, acidification and can include the analyses of changes in particle sizes, stable isotope signatures, minerals, metals, and diatom, invertebrate, plant and testate amoeba communities (e.g., Björck and Wohlfarth 2001, Abbott *et al*. 2000, Scholz *et al*. 2001, Meriläinen *et al*. 2003, Kilhman and Kauppila 2009, Wall *et al*. 2010, Kauppila *et al*. 2012). Additionally, to evaluate the transport of particles from peat extraction areas to lake sediments, particle tracking offers a practical solution to assess transport pathways of different sediments and to identify source-sink relations (Black *et al*. 2007).

Testate amoebae (TA) are a polyphyletic group of diverse, morphologically distinctive, ubiquitous protists characterized by outer shells (Tolonen 1986, Warner 1990) that preserve well in lakes sediments and peat deposits (Mitchell *et al*. 2007) even at low pH values that would affect the preservation of other fossil indicators such as molluscs, chironomids and ostracods (Dallimore *et al*. 2000, Beyens and Meisterfeld 2001). In addition, TA are highly abundant in wet environments and sensitive to environmental changes, i.e., they bear characteristics of ideal paleo-environmental indicators (Dallimore *et al*. 2000). TA communities from lentic ecosystems for example, have been used to study environmental changes caused by human activities such as land use changes and eutrophication (e.g., Boudreau *et al*. 2005, Reinhardt *et al*. 2005). Further, the material transported from peatlands downstream, typically includes organic particles smaller than 450 µm in size (Marttila and Kløve 2010), comprising the size range of TA and making them a potentially suitable particle tracer.

In this study, we evaluate the applicability of TA communities as potential indicators of organic matter leaching from peat extraction areas to lake sediments, and discuss whether there is an overall effect of sediment run-off from peat extraction areas on TA communities from receiving lakes compared with areas under other peatland use. For this, we compared surface lake sediments (current TA communities) with sediments before peat extraction started in Finland from unaffected lakes and lakes affected by peat extraction water discharge.

Material and methods

Study site and field sampling

A total of 36 lakes were selected in central Finland on the basis of possible impact from catchment peatland use: i) lakes with watersheds affected by peat extraction (impact); and ii) lakes with watersheds under different peatland use (control) (Fig. 1). In all catchment areas, other peatland uses such as agriculture and forestry were also prevalent. Lakes were paired (impact vs. control) according to their spatial proximity to the peat extraction sites. The physical, geological, and environmental settings in the catchments were identified, and in some cases, both impact and control lakes partly shared the same catchment while other pairs did not (Table 1). In cases where lake pairs shared the same catchment, the control lake received no influx from peat extraction areas and was situated upstream of the impact lake. From each lake, TA commu-

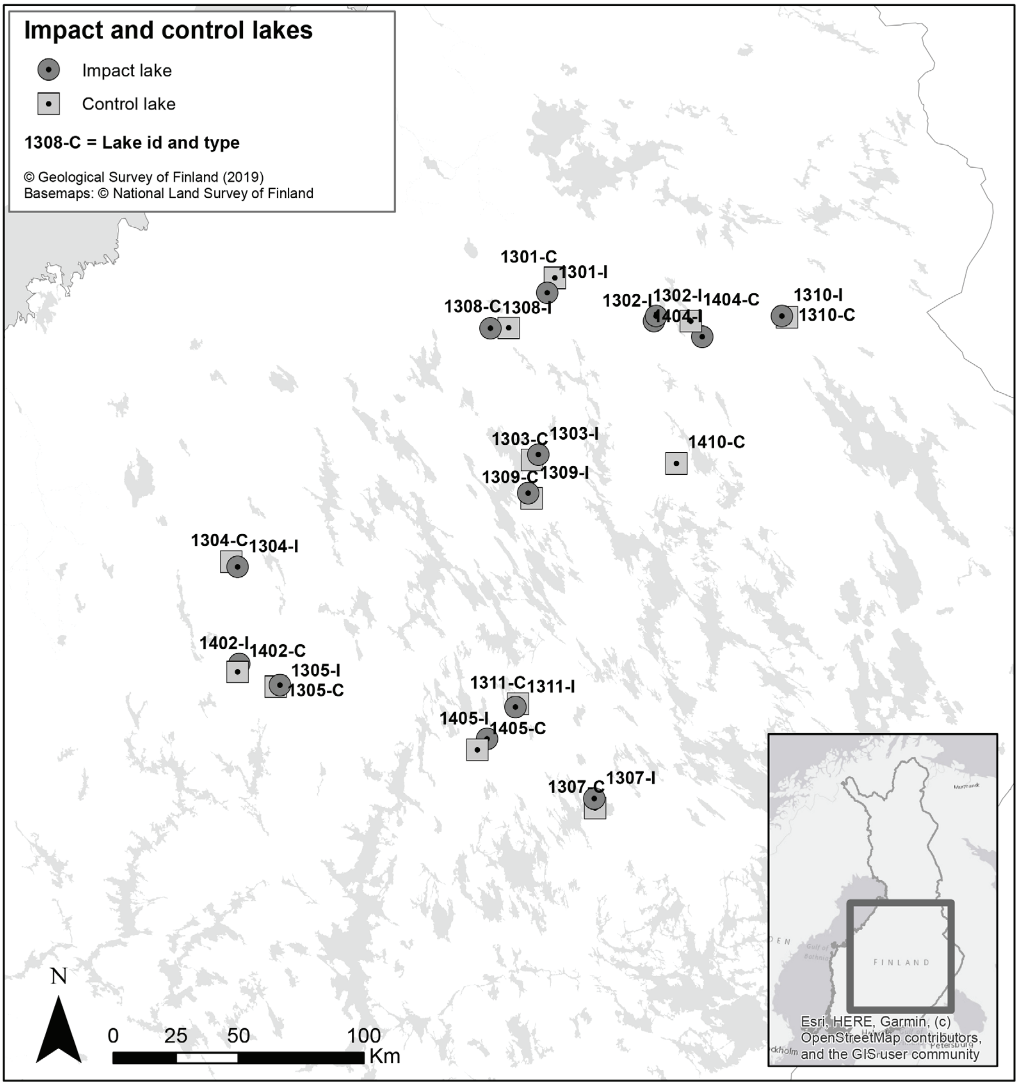

Fig. 1. Distribution area of the study sites in central Finland

nities were sampled from both sediments representing current and pre-peat extraction periods. Thus, the data set was divided into four groups: i) TA present day communities from control lakes; ii) TA present day communities from impact lakes; iii) TA communities predating peat extraction from control lakes; and iv) TA communities predating peat extraction from impact lakes. For brevity, groups will be referred to as: i) control surface; ii) impact surface; iii) control bottom; and iv) impact bottom, respectively.

Hydrological properties were used to classify the lakes into impact and control lakes (Appendix Table A1). We consider that a lake is possibly affected by peat production (impact) if it is located downstream of peat production areas or is directly receiving waters from the production area. The lake is considered unaffected by peat extraction (control) when there are no upstream peat production areas in the catchment.

The average lake area was 94.3 ha for impact (SD = 134.37) and 109.8 ha (SD = 164.1) for control lakes, whereas the average catchment area was 10 826.2 ha (SD = 8996.73) and 13 983.8 ha (SD = 9693.36), respectively. The average peatland area within impact lake

Table 1. Sediment depth–age levels, rate of deposition and depth of peak of AD 1986 Chernobyl ^{137}Cs from the sediment cores of the studied lakes.

Lake	Lake code*	Quality of ^{137}Cs analysis	Estimated depth of AD 1986 Chernobyl ^{137}Cs peak (cm)	Average rate of deposition between 1986–2013 (2014) (mm/yr)	Estimated # of years in the modern sample (0–1 cm)	Estimated decade for the sediment depth of 15 cm	Maximum ^{137}Cs concentration (Bq/kg)
Eitikka	1410	good	10	3.3	2	1970	483.69
Iso-Pajunen	1410	—	n/d	—	—		322.70
Peurajärvi	1310	poor	3	1	7	1860	926.88
Halmejärvi	1310	average	7	2.3	4	1950	1287.73
Haukilampi	1307	good	8	2.6	3	1960	7277.04
Hepojärvi	1307	good	3	1	7	1860	3389.62
Hietalampi	1402	good	5	1.7	5	1930	4004.22
Hirvijärvi	1402	poor	15	5	1	1985	1538.62
Ohenjärvi	1309	average	6	2	4	1950	1300.46
Ilkonlampi	1309	average	8	2.6	3	1960	1767.56
Iso Kiuloinen	1303	—	n/d	—	—		147.03
Sarvijärvi	1303	—	n/d	—	—		156.64
Kangaslampi	1401	good	4	1.3	6	1900	35531.21
Iso-Musta	1401	good	3	1	7	1860	8712.19
Kokko-Valkeinen	1304	average	3.5	1.2	6	1880	5921.71
Joutenjärvi	1304	average	4	1.3	6	1900	1478.78
Jyrkkä	1302	average	9	3	2	1970	1069.44
Päsmäri	1302	good	3	1	7	1860	1061.44
Jänkkärä	1311	average	3	1	7	1860	10381.34
Levänen	1311	good	6	2	4	1950	3872.93
Pitkänjärvi	1404	good	4	1.3	6	1900	873.82
Kotjonjärvi	1404	average	4.5	1.5	5	1920	1072.31
Lehmilampi	1305	good	5	1.7	5	1930	5363.70

n/d refers to those lakes where a ^{137}Cs Chernobyl peak could not be identified

* refers to the codes given in Fig. 1

Table 1. (*continued*)

Lake	Lake code*	Quality of ^{137}Cs analysis	Estimated depth of AD 1986 Chernobyl ^{137}Cs peak (cm)	Average rate of deposition between 1986–2013 (2014) (mm/yr)	Estimated # of years in the modern sample (0–1 cm)	Estimated decade for the sediment depth of 15 cm	Maximum ^{137}Cs concentration (Bq/kg)
Uitamonjärvi	1305	good	5	1.7	5	1930	3730.08
Marttisenjärvi	1301	average	5	1.7	5	1930	765.01
Salahminjärvi	1301	average	10	3.3	2	1970	489.89
Moskulanlampi	1403	average	11	3.7	2	1975	11086.68
Suojärvi	1403	poor	4	1.3	6	1900	2881.29
Valkeisjärvi	1308	—	n/d	—	—		754.08
Osmanginjärvi	1308	poor	3	1	7	1860	284.29
Vehkaputti	1407	average	5	1.7	5	1930	780.07
Pieni-Musta	1407	poor	5	1.7	5	1930	1895.74
Tiisjärvi	1409	poor	12	4	2	1980	2239.18
Saukkojärvi	1409	poor	5	1.7	5	1930	1427.64
Valkeislampi	1405	average	6	2	4	1950	9540.67
Ylemmäinen	1405	—	n/d	—	—		1002.03

n/d refers to those lakes where a ^{137}Cs Chernobyl peak could not be identified

* refers to the codes given in Fig. 1

catchments was 2492.6 ha (SD = 2606.98), and 3619 ha (SD = 3249.03) within the control lake catchments (Appendix Table A1). Only three catchments have a percentage of ditched area below 80%, and hence, none of the catchments can be considered to be near-pristine. However, in terms of catchment properties, all lakes represent typical lakes from peatland-dominated areas that have been affected by different anthropogenic land uses. The percentage of ditched area was higher for the impact sites (Paired *t*-test, $t = 2.27$, $p = 0.03$), while no differences were found between impact and control lakes for peatland percentage and percentage of production area ($t = 0.77$, $p = 0.44$ and $t = 0.76$, $p = 0.45$, respectively).

Lake echo-sounding studies were conducted during the springs and summers of 2013 and 2014 and the sediment sampling points were selected from the optimal sedimentation areas based on the echo-sounding study. Sediment samples were taken using a limnos sediment corer (for details see Kansanen *et al*. 1991). The first layer (current TA communities) and 15 cm depth samples (pre-peat extraction TA communities) of sediment were saved in plastic bags for TA extraction. The 15 cm depth sample was selected as the pre-peat extraction sample taking into account the annual sedimentation rate for a typical lake located in a peatland catchment. Thus, the 15 cm depth sample was expected to represent the sediments before the mid-1970s where industrial peat extraction began in Finland.

Sediment core dating

Caesium-137 is a radioactive isotope produced as a result of nuclear weapon testing and nuclear power plant accidents after the Second World War. Having no natural sources, ^{137}Cs contamination in environment stems from anthropogenic sources alone. The vertical distribution of caesium (^{137}Cs) provides an important dating method in establishing age–depth models of near-surface sediments (Appleby 2000). Based on annually laminated sediments (Ojala *et al*. 2012, Zolitschka *et al*. 2015), the 1986 peak from the Chernobyl accident is the dominant feature in sediments in southern and central Finland and central Sweden, completely masking the preceding features of the ^{137}Cs curve formed during the atmospheric nuclear weapons testing of the 50s and 60s (Klaminder *et al*. 2012, Ojala *et al*. 2016). Consequently, as our study area is mainly located within the Chernobyl fallout area, we interpreted observed peaks in ^{137}Cs in the recent sediments to be indicative of the year AD 1986.

All Caesium-137 analyses were performed using two different gamma spectrometers, an older EGandG Ortec ACE™-2K equipped with a four-inch NaI/TI detector and a new fully digital BrightSpec bMCA-USB pulse height analyser coupled to a well-type NaI(Tl) detector. Ojala *et al*. (2016) showed that despite detector-specific differences, both instruments provide similar results for the Chernobyl-age sediments, regardless of the sample pre-treatment or normalizing procedure. Here, ^{137}Cs concentrations of 36 study lakes were measured with a 1 cm resolution. The length of the sediment sections measured from each lake varied between 10 cm and 20 cm, and some of the measurements were repeated using parallel sediment sequences to assure the quality of caesium-137 determinations and sediment subsampling.

Testate amoebae sample processing

Sediments were preserved in plastic bags at 4°C and analysed during the following weeks after sampling. Sediments were gently mixed inside the bags to homogenize the sample. To obtain TA, 2 g of sediment were used. The subsamples were transferred to 250 ml flasks with 100 ml of distilled water and one tablet of *Lycopodium clavatum* (Batch 1031) standard preparation from Lund University (Sweden). After boiling, samples were gently sieved through a 300 μm mesh to remove coarse material (protocol slightly modified from Booth 2010). To retain TA, the mesh size commonly used in these types of studies ranges between 30 μm and 60 μm. However, in this study a 21 μm mesh was purposefully used to also include small TA. Retained material was centrifuged for 5 min at 3000 rpm. The supernatant was removed and samples were stored in Eppendorf tubes for further analysis.

Since TA concentrations were typically very low in this study, samples with more than 30 TA were included in analyses. All TA were identified using 400× magnification (Olympus BX41 microscope) and was estimated using the *Lycopodium* counts as an external marker (see Stockmarr 1971 for details). TA identification was based on test characteristics following several taxonomic keys (Kumar and Dalby 1998, Charman *et al*. 2000, Meisterfeld 2002, Mitchell 2002, Clarke 2003, Mitchell 2003).

Data analysis

Lake pairs with a total TA count lower than 30 in either lake, were removed from the analysis. Paired *t*-tests were used to check for differences in catchment and lake properties between lake groups (control and impact). To keep a balanced design, both sediment layers from the lake and its pair, were removed, leaving a total of 28 lakes (14 control lakes and 14 impact lakes). TA concentration, taxa richness, and Shannon's diversity index (Morris *et al*. 2014) were calculated for each sediment layer and averaged by group and tested using paired *t*-tests. To identify differences in the community structures, a Non-Metric Multidimensional Scaling (NMDS) based on Sørensen's (Bray-Curtis) distance was performed on TA communities. To determine if lake or catchment properties explained the current community structure and composition of control and impact lakes, environmental variables were fitted into an NMDS ordination in a constrained analysis. All calculations were done using R (ver. 3.0.2) and the vegan package for NMDS analysis (Oksanen *et al*. 2015).

Results

Sediment cores

The quality of the ^{137}Cs dating profiles varied between the sites with regards to the shape of the curve as well as the dating potential (Fig. 2). For that reason, sediment sequences were classified according to the quality of ^{137}Cs dating based on the appearance and distinctiveness of the Chernobyl-derived AD 1986 peak. They were classified as: (i) good; (ii) moderate; (iii) poor; and (iv) undatable. Below, we will describe the quality of each record and provide the dating basis for estimates of different sediment depth–age levels and the rate of deposition (Table 1).

Many lakes showed a distinct AD 1986 ^{137}Cs peak at the depth between 3 cm and 10 cm (Fig. 2) and were rated as "good", and i.e., they could be successfully dated with the profiles of the uppermost sediment sections. In some lakes however, the shape of the ^{137}Cs curve with a wide maximum peak of concentration spanning between ca. 4–12 cm being different compared to any of the other analysed lakes. We interpret that the lower section (9 cm) of these peaks represents AD 1986, because at this sediment depth, the ^{137}Cs concentration clearly rises above the background level. However, in these cases the shape of the curve is less clear due to a lower number of actual data points below the peak and thus the quality of dating was considered to only be of "moderate" accuracy. Some ^{137}Cs profiles were categorized to represent "poor" quality of dating because they were missing a clear peak values but show some indications of elevated concentration that were interpreted to represent AD 1986. There were also lakes that we considered entirely unsuitable for dating. Many of the unsuitable lakes had low concentration of caesium in the sediment, and were located in areas that are known to have received much less fallout from the Chernobyl accident than the other studied lakes.

Finally, in general, the estimated age of the sediment at 15 cm depth followed our expectations and predated the industrial extraction of peat for all but one impact lake i.e., Hirvijärvi. We therefore ran the analysis both including the lake pair 1402 and excluding it to assess its impact on the overall results. As the results did not change significantly, we present the results including the lake pair: Hirvijärvi-Hietalampi.

TA communities

A total of 54 TA taxa were identified in all the samples analysed. Overall, TA concentration was

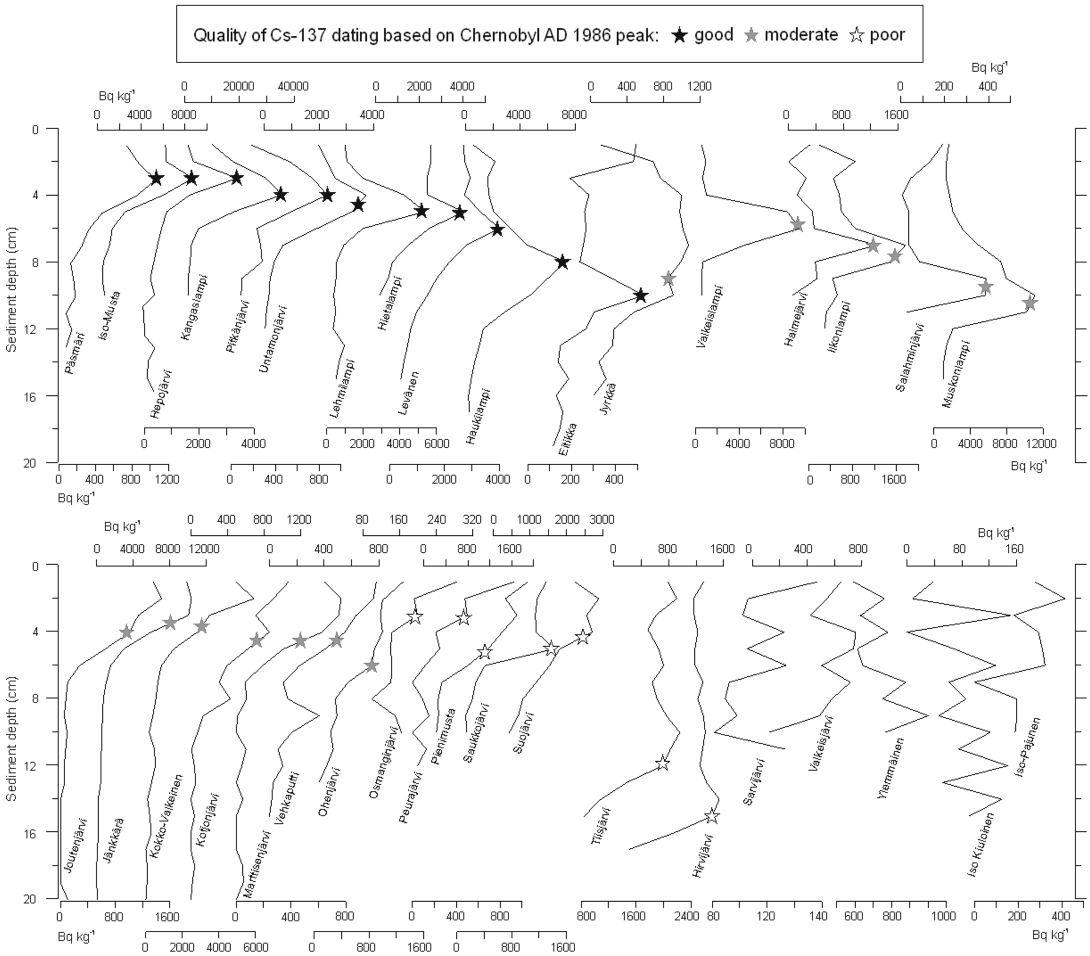

Fig. 2. Vertical distributions of caesium (^{137}Cs) in sediment cores of studied lakes.

slightly higher in the impact surface sediments, the highest taxa richness was found in the control bottom sediments and the highest diversity in the control surface sediments (Fig. 3). However, no significant differences were found when comparing concentration, richness and diversity between sample pairs: current communities from control lakes vs. impact lakes, current communities vs. pre-peat extraction in both control and impact lakes (paired t-test, $p > 0.1$).

On average, *Trinema lineare* was the most abundant taxa in all the sediment samples except for the impact bottom sediments where *Difflugia oblonga* "glans" showed the highest numbers, representing 11.8% of the community in that sample (Appendix Table A2). In general, TA from the genus *Difflugia* were abundantly found in all samples; the most common taxa being *Difflugia pulex*, *D. oblonga* "oblonga", *D. oblonga* "glans", *D. oblonga* "bryophyla", *D. urceolata* "urceolata", *D. urceolata* "elongata" and *D. protaeiformis* "acuminata". *Euglypha rotunda* and *Trinema-Corythion* were also two abundant groups among all samples. The different forms of the taxa listed in this study (Appendix Table A2), correspond to morphological varieties that have been separated into sub-groups by the taxonomic key for lake sediment TA designed by Kumar and Dalby (1998). However, despite the observed patterns, no association between specific taxa and sediment layers or catchment land use were identified.

NMDS ordinations repeated patterns observed for community indices (diversity and

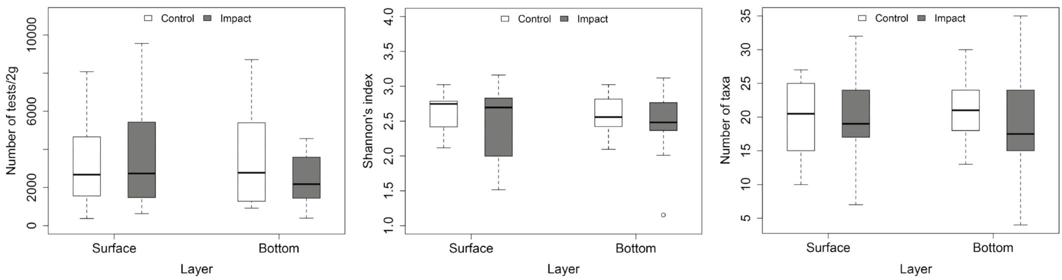

Fig. 3. Concentration (left), Shannon diversity (centre) and taxa richness (right) of testate amoebae communities in surface and bottom sediment layers of control and impact lakes. Concentrations of testate amoebae are per two grams of sample.

richness), and did not clearly group the data by sediment layer or by catchment land use (Fig. 4). Lake and catchment properties fitted into the NMDS ordination showed no association with the composition and structure of current TA communities (no significant relationship (R^2) at $p < 0.05$; Table 2).

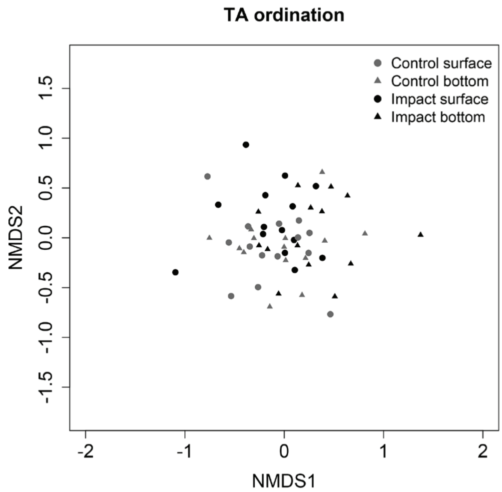

Fig. 4. NMDS ordination of testate amoeba communities from surface and bottom sediment layers of control and impact lakes.

Discussion

This study aimed to determine whether testate amoebae communities (TA) are a suitable indicator of organic matter leaching from peat extraction areas to lakes sediments. As TA are expected to be transported downstream, they were assumed to accumulate at higher rates in lakes receiving discharge from peat extraction areas (impact lakes) compared with lakes receiving discharge from peatlands drained for other purposes (control lakes). We found no statistically significant differences between TA communities from the control and impact lakes nor between the current TA communities of impact lakes (topmost sediment layer) and the pre-peat extraction communities of the same lakes.

In aquatic systems, the distance travelled by suspended particles is of high importance in many ecological processes. For example, disturbance caused by an increase in sediment load can affect dispersal, migration and population dynamics of organisms in streams (McNair and Newbold 2001). Thus, strong upstream human disturbance (e.g., peat extraction) should be observable in downstream reaches because dislodged material is travelling downstream in runoff (Kløve 1998). However, such an effect does not seem to be reflected in the TA communities found in the receiving lakes' sediments.

A number of physical factors and biotic interactions might explain the observed lack of differences in TA communities between the sediments from impacted and control lakes. For example, along the path from the upstream reaches to the lakes, there are a variety of very efficient particle retention structures, such as woody debris (Chergui *et al*. 1993), submerged macrophytes (Petticrew and Kalff 1992), cobbles (Chergui *et al*. 1993), riffles (Prochazka *et al*. 1991), and filter-feeding macroinvertebrates (Strayer *et al*. 1999, Malmqvist *et al*. 2001), that actively or passively retain TA before they reach the lakes.

In some of the sediment samples studied, very few TA were counted. However, the usual 150 tests counted for TA studies were found in more than half of the total samples. The low numbers of TA found in some lakes could be related to the sample preparation technique, as the amount of residuals following the treatment is relatively high — making TA difficult to pass through the 300 µm sieve. On the other hand, the low counts of TA might also be related to the effect of particle aggregation. When TA aggregate with other sediment particles, their mass increases and they sink before they reach the lakes (see also Black *et al*. 2007). Our results could also be explained by the distance from the peat extraction areas to the receiving lakes. Thus, lakes located closer to the peat extraction areas may receive more TA than those further away. However, it was not possible to obtain reliable calculations of peatland production area–lake distances in this study as such estimates would have had several potential sources of error: i) some lakes received discharge from multiple peat extraction areas; ii) the production rate of peat in extraction areas varied annually; and iii) the extraction areas of studied lakes were in various phases of peat production.

In flowing waters, seston (suspended matter) that is transported downstream, is often trapped by suspension feeders. While only part of the seston is directly assimilated by filter feeders, a big amount of the material is transformed into faecal pellets (e.g., Malmqvist *et al*. 2001). One example of such feeders are the blackfly larvae (Diptera: Simuliidae) that capture small seston and abundantly egest larger-sized faecal pellets (Hershey *et al*. 1996) due to their low efficiency to assimilate organic matter (Wotton 1978). This could potentially explain why dislodged TA from the peat extraction areas do not necessarily get transported all the way to the receiving lakes, but instead are trapped by temporally-abundant populations of filter-feeders (Malmqvist *et al*. 2001). However, if the amount of transported TA from peat extraction areas would be significantly higher when compared to other peatlands, we expect that this should still be observable in the receiving lake sediments due to the time lag between the establishment of such dense filter feeding communities and the onset of the high runoff periods from productions areas.

Another potential explanation for the lack of differences between TA communities in the current sediments from the control and impacted lakes is that control lakes are located in peatland catchments ditched for various other purposes than peat extraction, namely forestry and/or agriculture. In addition, peatlands drained for forestry require a periodic ditch network maintenance either by cleaning the existing ditches or by creating new ditches in between the previously existing ones, resulting in periodic increases in the suspended solids export (Aström *et al*. 2001, Nieminen *et al*. 2017, Koivusalo *et al*. 2008, Nieminen *et al*. 2017). In Finland, this drainage ditch maintenance practice has been considered to be very harmful for surface water quality (Finér *et al*. 2010). Given that ditching of forested peatlands results also in an increase in the export of suspended solids to the receiv-

Table 2. Relation of catchment and lake properties. NMDS ordination axis represents testate amoeba community structure and composition from control and impact lakes. The columns, R^2 and *p-value* refer to the correlation between the fitted values and the significance (at the 0.05 significance level), respectively.

Variable	NMDS1	NMDS2	R^2	*p-value*
Lake area (ha)	0.33	0.94	0.02	0.76
Catchment area (ha)	0.39	0.92	0.14	0.12
Peatland area (ha)	0.60	0.79	0.15	0.12
Ditched area (ha)	0.57	0.82	0.18	0.09
Production area (ha)	0.46	0.88	0.12	0.18
% Ditched area	0.03	0.99	0.03	0.68
% Peatland	0.98	0.15	0.09	0.30
% Production	0.63	0.77	0.02	0.75

ing water bodies, the observed TA communities may show response patterns not specific to peat extraction practices alone. However, given differences in the TA community composition between forested peatlands (drained for forestry purposes) and non-forested peatlands (usually drained for peat extraction; Daza Secco *et al*. 2016), we would have expected to observe structural and compositional differences in the communities found in impact vs control sediments. Kauppila *et al*. (2016) studied two of the lakes using the same pairing design used in our study (one impact and one control), and found that peat extraction did not increase the sediment thickness in the impact lake compared to its control. Similar studies evaluating the influence of the catchment area on the amount of organic matter discharge to the lakes (in the form of dissolved organic matter or total organic carbon) have suggested that the increase in organic matter is related to peatland proportion in the catchment (e.g., Mattsson *et al*. 2005) or the percentage of area covered by lakes (e.g., Arvola *et al*. 2016). Although the effect of catchment was not specifically examined in our study, peatland proportion of the catchments was not significantly different between the lake groups. Additionally, we found that none of the catchment or lake variables measured explained the structure and composition of current TA communities in the control and impact lakes.

The high abundance of *Trinema lineare*, *Trinema-Corythion* and *Difflugia pulex* in our samples differ from findings of earlier studies on TA from lake sediments (e.g., Boudreau *et al*. 2005, Burdíková *et al*. 2012). It should be noted, however, that we used a mesh size of 21 μm instead of sizes > 30 μm to retain tests. As the length of *T. lineare*, *Trinema-Corythion* and *D. pulex* is around 30 μm, they are not typically retained and commonly reported in lake studies despite being considered ubiquitous species found in most freshwater environments (e.g., Balík and Song 2000). Another abundant TA group found in all our samples were species of the genus *Difflugia* commonly found in lake sediments, both in current and fossil communities (e.g., Boudreau *et al*, 2005). These results do not highlight any change in the taxa composition of TA lake communities, as these taxa are not particularly associated to mosses or peatlands; but instead commonly found in lakes, and hence, do not indicate particle movement from the peatlands to the receiving lakes. Opportunistic taxa such as *D. protaeiformis* (Asioli *et al*. 1996, Reinhardt 1998), and particularly the “acuminate” variety, were abundant in all our sediment samples while *Centropyxis aculeata*, considered a typical benthic species (Balík and Song 2000), was present in most samples in low densities.

Results found by Patterson *et al*. (2013) to determine the effectiveness of TA as pH indicators found that some of the most representative taxa found across all sites belong to the genus *Difflugia* and *Centropyxis*. These findings are in concordance with our results suggesting that these taxa are commonly reported for boreal lakes. The high concentrations of *Difflugia spp* also suggests that all sediment lakes might contain high concentrations of organic molecules as these taxa are commonly related to habitats with plentiful sources of organic matter (Roe and Patterson 2014). Finally, similar studies have found sedimentary phosphorus as a main driver of TA community structure and composition in lakes (Patterson *et al*. 2012). Phosphorus concentration is one of the main factors determining the level of eutrophication, the lack of differences observed between TA communities which suggests that nutrient concentrations might not be significantly different between sediment groups.

Kihlman and Kauppila (2009, 2012) found a consistency between the geochemistry of lakes impacted by mining of metals and the TA species composition due to changes in environmental parameters, e.g., pH and metal concentrations. In the present study, water chemistry was not recorded. However, as no differences were found between the TA communities from the different sediments, and taking into account that TA respond fast to environmental changes, we suggest that the changes caused by peat extraction are not different enough from other activities on peatlands involving ditching to be reflected by TA communities in lake sediments.

Additionally, commonly known responses of lakes receiving discharge from disturbed peatlands are increases in suspended solids and nutrient concentrations (Marttila and Klöve 2010, Kauppila *et al*. 2016). Our results suggest that lake TA communities are not reflecting the potentially increased discharge of particles reaching impacted lakes. Although TA are likely to be dislodged due to peatland management practices, TA do not become deposited in downstream lakes, but instead are retained actively or passively upstream in ditches, brooks or rivers. Similarly, there may not be a significant difference in the rate of dislodgement of particles from peatlands drained for peat extraction when compared to that of peatlands used for forestry and agriculture. The lack of differences could be a result of water treatment procedures used in peat extraction areas that retain a substantial part of solids dislodged after ditching. In conclusion, despite the good applicability of TA communities as bioindicators of land use change in peatlands (Koenig *et al*. 2015, Daza Secco *et al*. 2016, Daza Secco *et al*. 2018) they may not be an adequate tool to trace changes caused by peat extraction compared to other peatland uses, or indicate that there are actually no significant differences between the effects of peat extraction compared to other land uses on lakes.

Finally, based on the study made in peatlands by Fialkiewicz-Koziel *et al*. (2015) who found that TA test composition seem to be affected by environmental disturbance such as extreme atmospheric pollution, we suggest that similar future studies investigating the potential of TA as indicators of changes in OM inputs into lakes should consider also TA functional traits and actual test composition as well.

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Appendix

Table A1. Geographic location (EUREF_FIN_TM35FIN WKID: 3067 Authority: EPSG), classification (Impact and Control paired according to their physical similarities and proximity) and summary of lake and catchment properties.

Lake	Status	Lake code (+)	Sampling year	Coordinates X	Coordinates Y	Lake area (ha)	Mean depth (m)	Catchment area (ha)	Peatland area (ha)	Ditched area (%)	Peatland (%)	Peat Prod. (%)
Eitikka	Control	1410	2014	546619.3564	7013137.174	97	2.1	8373	871	89.2	10.4	0.0
Iso-Pajunen	Impact	1410	2014	545089.8543	7012346.794	58	0.9	9144	1769	90.5	19.3	0.4
Peurajärvi	Control	1310	2013	590739.9878	7071338.286	19	6.2	26191	3678	86.6	14.0	0.0
Halmejärvi	Impact	1310	2013	588743.6577	7071904.004	24	5.0	13153	2739	92.2	20.8	1.2
Haukilampi	Control	1307	2013	514085.5042	6875321.243	16	*	37161	9904	90.2	26.7	4.6
Hepojärvi	Impact	1307	2013	513816.6089	6878865.772	77	*	37161	9904	90.2	26.7	4.6
Hietalampi	Control	1402	2014	371231.4997	6929710.647	18	*	9486	2227	95.3	23.5	6.6
Hirvijärvi	Impact	1402	2014	371888.3254	6932875.181	92	1.0	9486	2227	95.3	23.5	6.6
Ohenjärvi	Control	1309	2013	488721.9734	6999088.983	28	*	12014	2692	96.3	22.4	1.3
Ilkonlampi	Impact	1309	2013	487280.5553	7001033.587	17	*	2231	336	98.0	15.1	3.7
Iso Kiukoinen	Control	1303	2013	488888.5289	7014326.075	14	*	13417	2888	93.4	21.5	1.4
Sarvijärvi	Impact	1303	2013	491305.384	7016530.764	8	*	13417	2888	93.4	21.5	1.4
Kangaslampi	Control	1401	2014	396296.9946	6939831.421	10	5.2	7108	1598	92.5	22.5	2.0
Iso-Musta	Impact	1401	2014	395246.7104	6939722.847	22	3.5	7108	1598	92.5	22.5	2.0
Kokko -Valkeinen	Control	1304	2013	368752.985	6973747.704	26	*	17173	7335	76.4	42.7	5.2
Joutenjärvi	Impact	1304	2013	371199.642	6971581.118	42	0.9	15083	4985	76.6	33.0	6.2
Jyrkkä	Control	1302	2013	538481.0418	7071670.186	529	2.8	28103	6938	93.3	24.7	0.0
Päsmäri	Impact	1302	2013	537517.0826	7069640.798	283	1.9	28103	6938	93.3	24.7	0.0
Jänkkärä	Control	1311	2013	483264.5481	6917034.226	167	4.2	4314	748	94.9	17.3	2.9
Levänen	Impact	1311	2013	482174.5023	6915527.398	31	1.1	4314	748	94.9	17.3	2.9
Pitkänjärvi	Control	1404	2014	552439.3586	7070013.339	155	3.4	20226	6991	92.6	34.6	0.6
Kotjonjärvi	Impact	1404	2014	556836.7424	7063456.951	79	2.2	4252	1609	87.4	37.8	4.9
Lehmilampi	Control	1305	2013	386479.1559	6923821.825	18	2.9	3483	110	100.0	3.1	0.0
Uitamonjärvi	Impact	1305	2013	388093.4919	6924280.878	51	4.9	2213	327	100.0	14.8	0.0

+ refers to the codes given in Fig. 1

* refers to missing average depths

Table A1. (*continued*)

Lake	Status	Lake code (+)	Sampling year	Coordinates X	Coordinates Y	Lake area (ha)	Mean depth (m)	Catchment area (ha)	Peatland area (ha)	Ditched area (%)	Peatland (%)	Peat Prod. (%)
Uitamonjärvi	Impact	1305	2013	388093.4919	6924280.878	51	4.9	2213	327	100.0	14.8	0.0
Marttisenjärvi	Control	1301	2013	498022.5109	7087093.65	526	3.0	6513	1503	91.9	23.1	0.1
Salahminjärvi	Impact	1301	2013	494858.5307	7081086.513	520	7.3	5375	890	91.6	16.6	3.2
Moskulanlampi	Control	1403	2014	506221.3537	6955295.856	9	*	12621	1545	99.3	12.2	1.0
Suojärvi	Impact	1403	2014	510546.0734	6955459.298	26	*	12621	1545	99.3	12.2	1.0
Valkeisjärvi	Control	1308	2013	479522.2424	7067206.565	159	1.1	15672	7258	95.1	46.3	4.5
Osmanginjärvi	Impact	1308	2013	472329.621	7066898.153	286	1.2	7439	1105	97.0	14.9	0.8
Vuorilampi	Control	1407	2014	3419342.339	6916621.236	8	*	4437	284	97.4	6.4	1.0
Pieni-Musta	Impact	1407	2014	421668.7425	6913433.265	8	2.0	5510	290	92.7	5.3	1.0
Tiisjärvi	Control	1409	2014	306588.9376	6973309.942	170	*	22727	8422	85.6	37.1	2.1
Saukkojärvi	Impact	1409	2014	310302.0836	6952549.14	60	0.6	12532	4678	73.9	37.3	0.2
Valkeislampi	Control	1405	2014	467172.0509	6898602.014	8	*	2690	152	94.3	5.6	0.0
Ylemmäinen	Impact	1405	2014	470895.6845	6902925.298	14	*	5729	293	99.1	5.1	0.4

+ refers to the codes given in Fig. 1

* refers to missing average depths

Table A2. Mean abundance (%) of testate amoebae taxa in surface and bottom sediment layers of control and impact lakes. Standard deviations are given in parenthesis.

Taxa	Control (abundance %)		Impacted (abundance %)	
	Surface	Bottom	Surface	Bottom
Archerella flavum	0.00 (0.00)	0.30 (0.78)	0.00 (0.00)	0.89 (1.80)
Assulina muscorum	0.00 (0.00)	0.00 (0.00)	0.00 (0.00)	0.37 (0.91)
Arcella vulgaris	1.87 (2.41)	1.12 (1.54)	1.69 (2.27)	0.76 (1.61)
Arcella catinus	0.46 (1.72)	0.57 (1.71)	0.00 (0.00)	0.05 (0.19)
Centropyxis aculeata "aculeata"	1.29 (2.88)	1.08 (2.25)	0.08 (0.30)	0.60 (1.21)
C. aculeata "discoides"	0.57 (1.47)	2.40 (4.38)	0.69 (2.45)	0.51 (1.02)
Centropyxis constricta "constricta"	0.74 (1.78)	1.48 (3.51)	1.06 (1.77)	1.52 (2.36)
C. constricta "spinosa"	0.06 (0.22)	0.99 (2.26)	0.72 (1.22)	1.36 (4.42)
C. constricta "aerophila"	0.14 (0.54)	0.91 (2.25)	0.48 (0.84)	0.38 (0.97)
Cryptodifflugia sp.	0.58 (1.79)	0.00 (0.00)	0.07 (0.27)	0.27 (0.89)
Cucurbitella tricuspis	1.65 (3.76)	0.75 (1.26)	3.83 (9.51)	1.50 (3.66)
Cyphoderia ampulla	1.07 (1.24)	1.37 (2.66)	2.33 (3.07)	0.59 (1.40)
Cyclopyxis arcelloides	3.17 (3.19)	2.34 (2.62)	2.06 (3.69)	2.46 (4.55)
Difflugia pulex	4.81 (4.41)	10.16 (7.68)	2.24 (3.41)	6.09 (7.64)
Difflugia lucida	3.27 (4.52)	3.81 (4.32)	1.13 (2.92)	1.71 (2.98)
Difflugia globulosa	2.78 (3.60)	1.36 (2.26)	2.65 (3.01)	3.43 (4.51)
Difflugia corona	0.67 (1.32)	0.10 (0.26)	0.51 (1.19)	0.40 (1.36)
Difflugia bacillariarum	0.73 (1.82)	0.40 (0.79)	0.96 (3.09)	0.54 (1.82)
Difflugia tuberculata	0.04 (0.18)	0.05 (0.20)	0.25 (0.94)	0.00 (0.00)
Difflugia pyriformis	0.22 (0.83)	0.18 (0.69)	0.49 (1.84)	0.28 (1.06)
Difflugia oblonga "oblonga"	3.67 (3.04)	4.85 (3.12)	3.74 (3.61)	7.94 (5.38)
D. oblonga "glans"	5.96 (4.13)	4.90 (3.93)	7.17 (4.10)	11.83 (13.70)
D. oblonga "lanceolata"	1.74 (3.93)	0.99 (1.52)	2.87 (3.16)	1.79 (2.64)
D. oblonga "bryophila"	3.79 (5.23)	3.25 (5.22)	2.54 (3.36)	5.71 (8.02)
D. oblonga "tenuis"	0.75 (1.13)	1.35 (2.14)	1.60 (2.87)	2.57 (3.18)
D. oblonga "spinosa"	0.66 (1.54)	1.53 (5.12)	2.80 (7.20)	2.91 (9.90)

Table A2. (*continued*)

Taxa	Control (abundance %)		Impacted (abundance %)	
	Surface	Bottom	Surface	Bottom
D. oblonga "linearis"	0.33	0.45	0.21	0.25
	(0.67)	(1.13)	(0.50)	(0.64)
D. urceolata "urceolata"	6.07	3.23	7.85	4.38
	(7.12)	(4.56)	(11.53)	(4.26)
D. urceolata "elongata"	5.45	3.94	6.37	6.92
	(3.82)	(4.33)	(4.87)	(6.88)
D. protaeiformis "amphoralis"	1.32	0.46	0.60	0.65
	(1.94)	(0.52)	(1.36)	(1.12)
D. protaeiformis "acuminata"	6.72	2.07	2.89	4.01
	(0.71)	(2.71)	(3.21)	(5.44)
D. protaeiformis "claviformis"	1.44	1.75	2.96	3.25
	(1.97)	(1.98)	(2.97)	(4.78)
Euglypha tuberculata	2.10	1.48	1.04	1.79
	(3.36)	(1.79)	(2.00)	(2.89)
Euglypha rotunda	3.43	4.51	2.92	2.42
	(5.13)	(6.86)	(5.23)	(3.71)
Euglypha compressa	1.22	2.28	0.88	0.91
	(1.91)	(4.35)	(1.81)	(2.57)
Euglypha cristata	0.54	0.00	0.04	0.05
	(1.72)	(0.00)	(0.18)	(0.19)
Euglypha acanthophora	1.71	1.37	1.08	0.56
	(2.53)	(2.13)	(1.53)	(1.10)
Hyalosphenia papilio	0.00	0.23	0.11	0.29
	(0.00)	(0.48)	(0.42)	(0.77)
Heleopera sphagni	0.88	2.71	1.28	2.74
	(2.78)	(3.13)	(2.11)	(3.92)
Lagenodifflugia vas	1.09	1.20	0.38	0.43
	(1.47)	(2.85)	(1.13)	(0.86)
Lesquereusia spiralis	1.73	0.62	0.76	1.06
	(2.86)	(1.17)	(1.39)	(2.00)
Nebela bohemica	0.13	0.38	0.31	0.50
	(0.49)	(0.95)	(1.18)	(1.22)
Nebela parvula	0.00	0.00	0.04	0.09
	(0.00)	(0.00)	(0.17)	(0.25)
Nebela marginata	0.04	0.04	0.03	0.28
	(0.18)	(0.18)	(0.13)	(0.63)
Nebela sp.	0.14	0.18	0.20	0.20
	(0.39)	(0.40)	(0.55)	(0.56)
Pseudodifflugia fascicularis	1.45	0.78	0.39	0.36
	(3.02)	(1.64)	(1.33)	(1.37)
Paulinella chromatophora	2.89	0.81	1.47	0.58
	(6.12)	(1.93)	(2.21)	(1.41)
Pontigulasia compressa	0.24	2.21	1.67	1.02
	(0.91)	(7.48)	(4.63)	(2.44)
Quadrulela symmetrica	0.65	1.48	0.35	0.40
	(1.76)	(2.48)	(0.68)	(0.77)
Sphenoderia lenta	1.24	1.17	2.08	0.30
	(1.53)	(1.61)	(2.72)	(0.78)
Trigonopyxis arcula	0.11	0.00	0.03	0.06
	(0.30)	(0.00)	(0.14)	(0.23)
Tracheleuglypha dentata	1.91	3.46	3.46	3.91
	(2.03)	(3.66)	(3.61)	(3.86)

Table A2. (*continued*)

Taxa	Control (abundance %)		Impacted (abundance %)	
	Surface	Bottom	Surface	Bottom
Trinema lineare	10.00 (10.92)	10.97 (12.83)	13.66 (15.82)	3.39 (5.98)
Trinema-Corythion	6.26 (5.33)	5.76 (6.29)	4.72 (7.17)	2.47 (4.17)