Reconstructing the long-term history of water quality and availability using fossil diatoms at an agricultural site in the Cape lowlands

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ABSTRACT

The Berg River is a pivotal source of fresh water for domestic, industrial and agricultural use as well as for in stream ecology, therefore knowledge of what impacts this rivers water quality and assessing whether ecological resilience has been surpassed are of the utmost importance. Since diatom assemblages are inextricably linked to the chemical, physical and biological characteristics of their environment, they were chosen for this study to investigate long-term changes in water quality and availability and suggest potential drivers of such changes at an agricultural site (Rhenostervlei Farm) adjacent the Berg River in the Cape lowlands through fossil diatom analysis of a sediment core (RV3). Diatoms were extracted along the length of the RV3 core, chemically and physically treated to remove unwanted material and then were mounted on a slide to be counted. Twenty abundant diatoms were chosen as environmental indicators for the analysis. Their abundances were plotted against depth and age and changes in water quality and availability were inferred based on their autecological characteristics. The most prominent shift in the indicator diatoms at Rhenostervlei Farm as shown by the stratigraphic diagrams, CONISS analysis and the PCA occurred at the onset of the 20th century CE and was characterised by a shift from a saline, dry and nutrient-poor environment (ca. 1790-1890 CE) to a more turbid, nutrient-rich environment with increased freshwater influence that was prone to periodic flooding (ca. 1890-2011 CE). Through the chronological analysis and the interpretation of the historical record (climate and land-use) as well as other environmental proxies (sediment accumulation rate and macro-charcoal), the causes of the detected change in water availability and quality at the floodplain site were likely related to land-use change in the form of agricultural intensification at Rhenostervlei Farm and potentially in the Berg River catchment as a whole. This could have involved burning, clearance of natural vegetation, soil disturbance and fertilizer use - all of which contributed to increased surface runoff, erosion and nutrient and sediment loading into the site. Furthermore, water extraction and diversion in the 1950s could explain the decreased flooding signal (i.e. lower abundance of *Aulacoseira granulata*). Although no evidence of a catastrophic regime shift was identified, if land-use practices continue to intensify (e.g. increased fertilizer use) and future climate change interacts and influences the agricultural alterations to hydrological systems, we may expect increased vulnerability to global change and unexpected ecological outcomes such as regime shifts. In order to improve the interpretation of fossil diatom records in terms of changing water quality and availability, a study with multiple proxies should be undertaken to help infer environmental conditions in a complex environment that has many potential drivers, such as the Berg River.

1. INTRODUCTION

The challenge of determining the influence of climate and land-use change on ecosystem functioning is of crucial importance for biodiversity conservation, current and future land-use management as well as the sustainable use of ecosystem services (Carpenter et al., 2009; De Groot et al., 2010). Environmental changes, whether due to natural processes or anthropogenic forcings may pre-date historical information and measured data, hence alternative methods, such as palaeoecology, are needed to understand the natural variability of a system and assess whether its ecological resilience has been surpassed (Gillson, 2015; Forbes et al., 2018).

Palaeoecology, which involves the study of past ecosystems, is a useful tool, as it provides a long-term perspective which can be used to reconstruct environmental history over decadal-millennial timescales. These techniques allow for the observation of ecological systems before major environmental change, thus providing a reference state or range of variability against which to define restoration or management targets (Willis and Birks, 2006; Jackson and Hobbs, 2009; Forbes et al., 2018). Furthermore, a palaeoecological perspective provides an opportunity to identify drivers of environmental change by tracking the interacting effects of climate change and land-use in a system (Gillson, 2015).

When attempting to reconstruct past environments from stratigraphical records, a fundamental principle is applied; fossil organisms accumulated in the sediment have the same biological and environmental requirements as their modern day counterparts (Lawrence, 1971). Palaeoecology utilizes the relationship between organisms and their environments (Rull, 2010) to provide proxies which act as indirect environmental indicators (Gillson, 2015; Davidson et al., 2018). One such proxy that has been extensively used as biological indicators in palaeoecological studies are fossil diatoms preserved in the sedimentary record (e.g. Birks et al., 1990; Dearing et al., 2012; Brugam and Munoz, 2018). By transferring modern diatom ecological information to its fossilised occurrence within the palaeoecological-record, one can create a snapshot of the environment at the time of deposition (Lawrence, 1971).

Diatom assemblages are inextricably linked to the chemical, physical and biological characteristics of their environment. Therefore, by analysing preserved diatom assemblages in sediment cores, which may reflect hundreds to thousands of years of sediment accumulation (Julius and Theriot, 2010), it is possible to reconstruct how environmental conditions have changed over varying time-scales (Harding and Taylor, 2011). Changes in environmental conditions such as water quality and availability are predominantly driven by climate and land-use, thus fossil diatom analysis can have broader implications for piecing together how climate and land-use have changed over time as well as how they have impacted the aquatic ecosystems the diatom inhabits.

Climate is a key driver of water quality and availability through its influence on various hydrological processes such as precipitation, soil moisture changes, runoff, changes in the distribution of river flows and groundwater recharge (Kundzewicz et al., 2008). Therefore, the diatom-inferred changes in water quality and availability may act as climatic signals. For example, if diatoms in the sediment record show increased nutrients and salinity in a lake system, this can be an indirect measure of climate variability as lake water generally becomes more concentrated when regional climates become drier (Anderson, 1995). However, this climatic signal present in the sedimentary record can often be altered or obscured by anthropogenic activities (Wolin and Duthie, 1999).

Over the past few centuries human activities have exerted a fundamental influence on aquatic ecosystems and freshwater quality (Smith et al., 1999; Dudgeon et al., 2006; Hall and Smol, 2010). These human induced changes to hydrological systems can directly affect ecosystems and possibly lead to shifts in the biotic community (Snoeijs and Weckström, 2010); all of which can be captured by fossil diatoms in the sedimentary record (Battarbee, 1999). Therefore, palaeoecological studies using fossil diatom analysis can provide valuable insight into links between historic changes in water quality and land-use (Stevenson and Pan, 1999). Diatoms can be extremely sensitive to environmental change and therefore are useful biological indicators for assessing anthropogenic impacts on a system and evaluating whether steps should be taken to restore the system to a desired state of ecosystem function (Hall and Smol, 2010; Julius and Theriot, 2010).

1.1 Diatom morphology, habitat and their usefulness as palaeoecological proxies

Diatoms are a group of microscopic, unicellular algae of the phylum Heterokontophyta. All diatoms are characterized by having a siliceous cell wall known as a frustule which is composed of two valves and several girdle bands – a feature making them unique amongst algae (Taylor et al., 2007). Diatoms are divided into three groups, namely: i) Coscinodiscophyceae - commonly called centrales or centrics, ii) Fragilariophyceae, and iii) Bacillariophyceae – both commonly called pennates or pennales. Fragilariophyceae and Bacillariophyceae are differentiated by the absence or presence of a raphe, respectively, while centric and pennate diatoms are distinguished by the shape. Centrics are circular, where the frustule develops radially around a point while pennate frustules form bilaterally, elongated along a plane (Julius and Theriot, 2010).

Diatoms commonly have a cosmopolitan distribution and are found in a wide range of habitats where water is at least occasionally present (Smol and Stoermer, 2010). These habitats range from open ocean, estuaries and shallow coastal environments (see Fritz et al., 2010) to lakes and rivers (see Reavie and Edlund, 2010; Wolin and Stone, 2010) as well as damp subaerial habitats, including soils and bryophytes (see Gaiser and Ruhland, 2010; Johansen, 2010). Due to their sheer abundance and diversity, diatoms are an important and often dominant component of microalgal assemblages (Julius and Theriot, 2010).

Diatoms have a variety of characteristics that make them ideal proxies for environmental and climatic reconstructions (Julius and Theriot, 2010). Diatoms are powerful indicators because have a short life span with fast growth and immigration rates, which enables a rapid response to environmental changes and early colonization of habitats after a disturbance (Hall and Smol, 2010). They are ubiquitous, taxonomically distinct with unique morphologies allowing for species level identification and often preserve well in the sedimentary records due to their siliceous frustule (Stoermer and Smol, 1999).

Diatoms exhibit a variety of life strategies and different species have specific optima to given environmental variables, such as habitat and water chemistry requirements (Round et al., 1990). Diatoms are highly sensitive to many environmental variables, with their community as a whole responding directly to changes in light, moisture conditions, temperature, current velocity, turbidity, pH, oxygen, salinity, nutrient concentrations as well as fluctuations in water level (van Dam et al., 1994; Julius and Theriot, 2010). Essentially, diatom communities largely reflect the chemical, physical and biological variables present in the habitat (Fritz et al., 1999).

1.2 Inferring environmental change using diatoms

The analysis of the past and present assemblage composition and knowledge of the relevant autecological characteristics of the taxa present, aids in inferring environmental changes, including acidification, eutrophication and climatic changes (van Dam et al., 1994; Julius and Theriot, 2010). Autecological characteristics, which refers to a species' ecological preferences, sensitivities or tolerances, can be used to link diatoms to their preferred environment (Stevenson and Pan, 1999) Some of these include diatom life form, salinity preference and nutrients preference (Vos and Wolf, 1993; van Dam et al., 1994).

1.2.1 Life Form

Diatoms can be divided into two broad life form categories, namely: planktonic, which are free floating living in the open water; and benthic which are littoral forms, which live on and attached to various substrates in near-shore habitats (Wolin and Duthie, 1999). There are also tycoplanktonic forms, which are usually associated with the benthos or the near-shore community but can easily be transported into the water column (Vos and Wolf, 1993; Wolin and Duthie, 1999). As well as dominating in open waters, the planktonic community may also dominate in turbid waters where the reduced light penetration hinders benthic growth (Wolin and Stone, 2010). It has been suggested that planktonic diatoms in sediments are by definition allochthonous, having been transported (e.g. by tidal currents or rivers) from elsewhere and as result provide information on larger spatial scales (Reavie and Edlund, 2010). In contrast, benthic diatoms are often considered to be autochthonous, i.e. locally derived species, which give researchers the advantage of knowing that the diatom species probably lived at or near the sampled location. As a result, benthic diatoms are ideal indicator organisms as they can be confidently calibrated to the water quality and other environmental conditions present at their respective sampled locations (Reavie and Edlund, 2010).

Benthic diatoms can be subdivided into the following types based on their specific substrates they are associated with: epilithic diatoms (rocks/stones), epipsammic diatoms (sand), epipelic diatoms (fine sediments in the littoral zone) and epiphytic diatoms (macroalgae or water plants) (Vos and De Wolf, 1993; Hall and Smol, 2010). Shifts or dominance in one of these benthic groups can be indicative of changes in the environment or hydrological regime (Reavie and Edlund, 2010). For example, increases or decreases in epiphytic diatoms can indirectly track the expansion or retraction of macrophytes, which are known to be associated with soil formation and stabilization of the near-shore environment (Wolin, 1996). Therefore, a shift toward epiphytic taxa may indicate a higher abundance of submerged macrophytes in a low energy environment (Vos and De Wolf, 1993; Reavie and Edlund, 2010).

Aerophilic diatoms are another form of benthic taxa that are adapted to irregular floods and areas that are subject to periodic drying which include wet to semi-wet soils and mosses (Vos and De Wolf, 1993; Spaulding et al., 2010). The dominance of aerophilic species (e.g. *Pinnularia borealis*, *Hantzschia amphioxys* and *Diadesmis confervacea*) in the fossil records can be interpreted in numerous ways; the most common of which is that they represent shallow waters and possibly drought events (Gaiser and Ruhland, 2010). Therefore, the presence of aerophilic species can aid in inferring past climates and how they’ve changed.

In addition to climate change, aerophilous species can also help in elucidating how humans have impacted systems. For example, in a previous study conducted by Gell et al. (2007), the presence of aerophilous taxa, *Hantzschia amphioxys* and *Diadesmis confervacea*, in a sediment-based archive from a floodplain lake along the lower River Murray in Australia was used to indicate what environmental conditions in Loch Luna were like before European impact. In this study, predominance of aerophilous taxa prior ca. 1850 CE suggested that pre-European Loch Luna was a shallow water mudflat and suggested that the shift to deeper water conditions dominated by planktonic diatoms was as a result of anthropogenic impacts. As well as being an indicator of shallow water conditions, peaks in both aerophilic species and sedimentation rates may be a reflection of flooding events, resulting in increased erosion and surface runoff (van der Putten et al., 2008).

1.2.2 Salinity

As well as life form, diatoms can also be grouped based on their specific salinity habitat preference; which in order of decreasing salinity includes: marine, brackish, freshwater/brackish, brackish/freshwater and freshwater habitats (Vos and Wolf, 1993; van Dam et al., 1994). Salinity is generally a reflection of the interaction of source waters and inputs into an aquatic environment (Fritz et al., 2010). In freshwater systems such as inland rivers, salinity changes may be a product of varying influences of freshwater such as groundwater flow and precipitation and runoff. Salinity is largely controlled by climate in that variation in precipitation and temperature will affect the precipitation to evaporation ratio (*P-E*) with resultant consequences for the concentration or dilution of dissolved salts (Gasse et al., 1995; Haynes et al., 2007). Higher *P-E* (more freshwater input) would decrease salinity, while lower *P-E* (higher temperature or dry conditions with more evaporation) would increase salinity.

Diatom salinity preference has been utilised by many researchers to reconstruct climatic variability over time. For example, the precipitation variability in the winter rainfall zone of South Africa over the last 1400 years was determined using diatom-inferred salinity (Stager et al., 2012). Stager et al. (2012) interpreted elevated percentages of mostly planktonic, dilute-water diatoms (such as *Aulacoseira granulata*, *Aulacoseira ambigua*, *Nitzschia lacuum* and *Synedra cf. delicatissima*) as representative of increased runoff and river inputs into the lake under relatively wetter climatic conditions (i.e. an increased *P-E*), while high percentages of fresh-brackish indicator species (such as *Cyclotella meneghiniana*) were taken to represent more brackish conditions and moderately reduced *P-E*.

As well as diatoms in sediment records being used to reconstruct salinity changes under climate variation (varying influences of precipitation and evaporation), they have also been used to reconstruct human influences on hydrological systems. For example, Taffs (2001) used the diatom assemblage from three cores from key wetland areas of the Upper South East, Australia, to construct an inferred salinity curve and compared it to known changes in human activity which allowed for the reconstruction of salinity trends for the wetlands, over a period of European agricultural activities. The study by Taffs (2001) found that, while groundwater salinity was increasing, the wetland areas had experienced a freshening of surface water due to past human activities which through flow of surface water, a result of constructed drainage systems flushing salts from the wetlands.

1.2.3 Nutrients

The concentration and supply rates of nutrients can play a pivotal part in shaping diatom communities. Phosphorus, for example, is known to be one of the most commonly limiting factors in productivity in freshwater systems (Hall and Smol, 2010). The composition and supply of nutrients is determined by the watershed characteristics and by the amount of precipitation, respectively (Underwood and Kromkamp, 1999). Each diatom species has an optimum and a tolerance for nutrients and can generally be categorized in one of the four main groups, in order of increasing nutrient requirements: oligotrophic, mesotrophic, eutrophic and hypertrophic. Knowledge of a taxa's nutrient range may assist in the reconstruction of environmental conditions, especially where nutrient availability has played a central role in diatom community development and change (Hall and Smol, 2010).

Kirsten and Meadows (2016) utilised diatom nutrient preference to reconstruct past environmental change in Princessvlei, a eutrophic, freshwater coastal lake in the winter rainfall zone of South Africa. The 2600-year-old diatom record from this study revealed an oscillation between two ecologically stable states in Princessvlei: a low nutrients state characterised by clear water, oligotrophic, benthic species (e.g. *Planothidium biporomum* and *Achnanthes subaffinis*) and a turbid high nutrient state dominated by eutrophic, planktonic species (e.g. *Melosira nummuloides* and *Aulacoseira ambigua*). A comparison to known climate variability during the 2600 year period was then used to determine possible drivers of the ecological state shifts, which suggested the relative dominance of catchment precipitation or groundwater influx as a driver (Kirsten and Meadows, 2016).

Diatom nutrient preference has also been widely used to assess increased nutrient and particulate loading into rivers and lakes as a result of anthropogenic activities (Hall and Smol, 2010; Reavie and Edlund, 2010). Anthropogenic nutrient inputs largely come from domestic and industrial waste disposal, agricultural activities and soil erosion (Hall and Smol, 2010). Landscape modifications such as the removal of natural vegetation through deforestation or for farming can result in increased nutrient inputs and higher water levels as the lack of vegetation results in decreased evapotranspiration. Decreased evapotranspiration leads to increased surface runoff and soil erosion into water bodies (Wolin and Stone, 2010). Nutrients such as phosphorus and nitrogen in fertilizers can be transported downhill in runoff, often leading to aquatic eutrophication (Carpenter and Kinne, 2003; Gordon et al., 2008; Dodds and Smith, 2016).

To understand the history and impacts of human activities on Lake Superior over the last two centuries, Chraïbi et al. (2014) conducted a palaeolimnological study using diatom fossil assemblages, sedimentation rates, organic and trace metal accumulation rates, and GIS-reconstructed human land-use stressors. This study revealed a reorganization of the diatom community in the late 19th century which corresponded to the time of European settlement and significant agricultural development. This shift in the diatom assemblage was characterized by an increase in the species *Aulacoseira islandica* which is known to respond to phosphorus enrichment and likely occurred in response to nutrient enrichment by human activities such as deforestation, mining, agriculture, and raw sewage disposal (Chraïbi et al., 2014).

1.3 Using diatoms to assess environmental change in rivers and streams

Diatom-based palaeoecological studies which reconstruct environmental change in aquatic environments have thus far been predominantly focused on lake ecosystems (Reavie and Edlund, 2010). The successful application of palaeoecological studies in these depositional environments is largely due to the stable nature of lake sediments and their general resistance to erosion by physical processes making them suitable for acquiring stratigraphically constrained and dateable sediment cores (Battarbee, 1999; Reavie and Edlund, 2010).

In contrast to lakes, obtaining continuous sedimentary records from rivers and sites near rivers can be problematic due to dynamic natures of river ecosystems in which riverine material is constantly or frequently being reworked (Reavie and Edlund, 2010). As a result, the application of palaeoecological techniques to fluvial environments has largely been avoided (Reid and Ogden, 2006). However, the value of an extended temporal perspective is perhaps even greater for river ecosystems as these systems are characterised by high climatic and hydrologic variability at inter-annual and decadal scales (Reid and Ogden, 2006).

In South Africa, the potential of diatoms as indicators of present-day water quality was realised many years ago (e.g. Cholnoky, 1968; Schoeman, 1976; Bate et al., 2002). However, the use of fossil diatoms-based palaeoecology studies to reconstruct past water quality and availability are rather limited due to the low abundance of suitable sites and the discontinuous accumulation of aquatic sediments over time (Scott and Lee-Thorp, 2004). Of the studies that have been completed in South Africa, the majority are focused on lake or wetland ecosystems (see Braamhoek wetland in Finné et al., 2010; Lake Sibaya, Verlorenvlei in Stager et al., 2012, 2013), leaving river systems relatively untouched. However, with careful site selection the collection of material from more stable environments such as river floodplains, river ecosystems hold great potential for environmental reconstructions using fossil diatoms in sediments (Reavie and Edlund, 2010) in the South African context.

In the Western Cape of South Africa, the Berg River is a pivotal source of fresh water for domestic, industrial and agricultural use as well as for in stream ecology (River Health Programme, 2004). Healthy rivers in good ecological conditions can provide goods and services which contribute to human welfare and economic growth. Therefore, knowledge of the impacts on the Berg River's water quality and assessing whether ecological resilience has been surpassed are of the utmost importance.

In order to assess the ecological functioning of a river system and determine whether intervention is needed, palaeoecology can be used to detect trends and establish the historical range of variability (Reid et al., 2006). The palaeoecological approach is vital as it provides much needed long-term data to accounts for the high climatic and hydrologic variability at inter-annual and decadal scales that river systems experience. This approach can also lead to a better understanding of the processes driving change and thus can assist in identifying appropriate management actions to restore and enhance resilience of threatened environments (Reid et al., 2006).

1.4 Aims and objectives

The aim of the present study is to apply a palaeoecological-approach to develop a high resolution record of changing water quality and availability over time (centennial timescale) through fossil diatom analysis of a sediment core from a lowland agricultural site adjacent the Berg River in the Cape lowlands. As well as establishing trends and historical range of variability in the ecosystem, this project also aims to suggest possible drivers of change by comparing diatom abundances to historical records of climate and land-use and other proxies with the goal of informing land-use management practices.

The objectives of the present study include: (1) To reconstruct water quality and availability using diatoms as a proxy; (2) to suggest drivers of change by relating the changes in the diatom assemblages to records of climatic and land-use changes over the past ca. 220 years; and (3) to discuss the potential management implications of these findings.

2. METHODS

2.1 Site description

Rhenostervlei Farm is situated along the lower to middle reaches of the Berg River, just upstream from the Klein Berg tributary, within the Cape lowlands of the Western Cape, South Africa (Figure 1). The study site, located at 33°12'54.60" S and 18°56'57.60" E, is situated on a floodplain adjacent to the river on Rhenostervlei Farm. Three other sites were cored in the area, however, this site had the best palaeo-proxy preservation and was the least disturbed.

The farm is situated in an area that originally supported West-Coast Renosterveld vegetation. This area has two identifiable vegetation types classified by Mucina and Rutherford (2006), namely: Swartland Shale Renosterveld and Atlantis Sand Fynbos, which exist as an ecotone. Swartland Shale Renosterveld is characterised by medium dense cupressoid-leaved low shrubland with an open grassy understory and is dominated by renosterbos; while Atlantis Sand Fynbos has dense, moderately tall ericoid shrubland with occasional tall sclerophyllous shrubs and is dominated by restioid and proteoid fynbos (Mucina and Rutherford, 2006).

The geology of the lower to middle reaches of the Berg River is comprised of Table Mountain Group (quartzitic sandstone), Malmesbury Group (shale), Klipheuwel Group and Sandy sediments (River Health Programme, 2004). This area is situated within the Cape Floristic Region (CFR), and originally supported Fynbos (Sandstone, Shale, Alluvium) and Shale Renosterveld (River Health Programme, 2004). This area falls within the winter rainfall zone (WRZ) which is characterized by a temperate Mediterranean-type climate that experiences summer drought and maximum rainfall during the winter season (Tyson and Preston-Whyte, 2000; Chase and Meadows, 2007).

In the CFR, Renosterveld vegetation has been primarily affected by land-use, with 91-97% being transformed for mainly agricultural purposes (Von Hase et al., 2003; Rouget et al., 2003). This vulnerability of lowland Renosterveld to agricultural transformation is due to the higher nutrient status of lowland shales as well as its accessibility compared to other vegetation types at higher elevations (e.g. montane Fynbos).

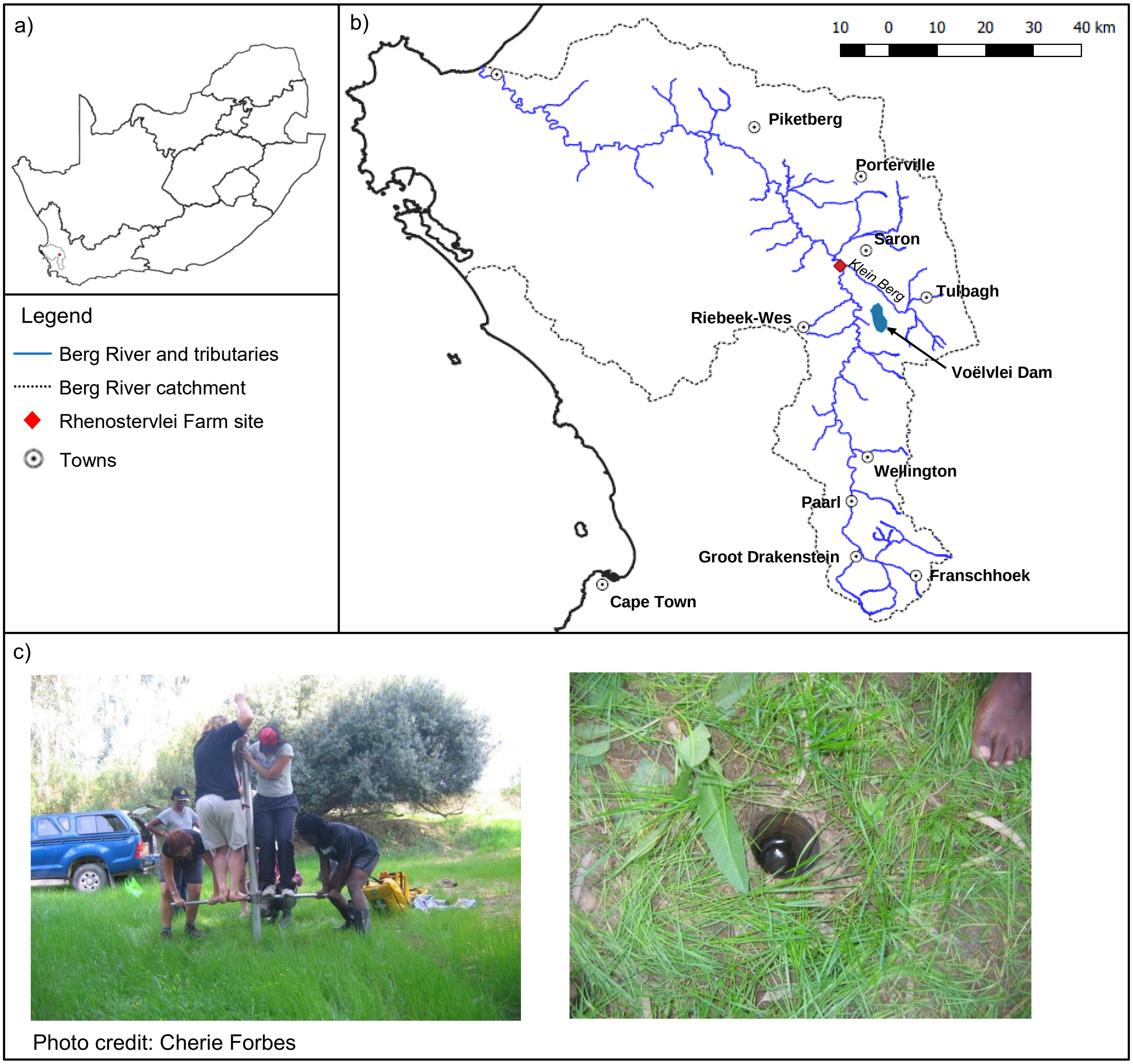

Photo credit: Cherie Forbes

Figure 1: *Map of study site including: a) the Berg River catchment within the Western Cape of South Africa; b) the Berg River and its tributaries with surrounding towns, the Voëlvlei Dam and the Klein Berg tributary - the red diamond indicates the location where the core was retrieved on Rhenostervlei Farm; and c) photographs of core retrieval in 2011.*

The Berg River flows for 300 km from the Groot Drakenstein Mountains near Franschhoek, northwards toward the west coast where it enters the sea at Velddrif (Figure 1). The Berg River drains a catchment area of about 9 000 km^2 and shows river flow peaks during the winter season, from June to August. Throughout the catchment, evaporation exceeds precipitation and the river water budget is dominated by runoff (De Villiers, 2007). Stream flow is highly seasonal, with low flows (upper river = 0.2 m^3/s and lower river = 2.0 m^3/s) in summer (November–February); and higher flows (upper river = 4 m^3/s and lower river = 15 m^3/s) in winter (May–August). Flooding occurs as regular winter events, in which flow rates may increase to 150 m^3/s in the upper river and 600 m^3/s in the lower river (G. Howard, Ninham Shand, pers. comm. Reported by Snaddon, 2009).

The Berg River catchment has a long history of river manipulations and agricultural activities. In 1699, farmers began to graze stock in the Swartland to the north of Cape Town, as far as Riebeek-Kasteel (River Health Programme, 2004). Later, during the 19th century, the towns of Hopefield, Moorreesburg, Darling, Porterville and Piketberg were established and grain farmers joined the stock farmers (River Health Programme, 2004).

Over the last century, the Berg River has experienced major changes (De Villiers, 2007; Kading et al., 2009). Various stretches of the river have undergone a decrease in flow. For example, in 1966, the river mouth closed completely during summer, and had to be dredged to allow local boaters to use the harbor (Slinger and Taljaard, 1994). The decrease in flow may be an indication of the high extent of recent water withdrawal within the catchment for agricultural activity: more than 50% of the catchment area supports agriculture (de Villiers, 2007). Various dams have been constructed along the Berg River since 1947 to provide drinking and irrigation water, and also have contributed to reduced stream flow (Kading et al., 2009). The Klein Berg tributary was historically perennial, but now has no flow in its lower reaches during summer as a result of over abstraction for agriculture and the diversion of water to Voëlvlei Dam, which was commissioned in 1952 (River Health Programme, 2004).

Since the 1960s the Berg River has been displaying a trend of increasing salt levels (DWAF, 1993), particularly along the middle to lower reaches. This could be attributed to the naturally saline geology of the river (De Clercq et al., 2010), but also may represent dryland salinization in which past clearing of natural vegetation (Renosterveld) for cultivated lands and pastures, alters the water balance , mobilizing salts stored in the regolith (Pannell and Ewing, 2004). This is evident throughout the wheat lands of the Swartland region in the Western Cape.

The nutrient status of the Berg River has also deteriorated due to a 10-fold increase in inorganic nitrogen and phosphorus content over the past 20 years due to agricultural runoff and sewage effluent, further exacerbated by reduced runoff (De Villiers, 2007). The timeline for climatic and land-use changes for Rhenostervlei Farm and surrounding areas is displayed in Table 1.

Table 1: *Timeline outlining climatic and land-use change during the study period (ca. 220 years) at Rhenostervlei Farm and the Berg River catchment. Inserts highlighted in bold are specific to Rhenostervlei Farm.*

Climate	Land-use	Date	Description	Reference
20th century warming	**Commercial crop farming (wheat, oats and Cape Pondweed) and rotational grazing**	**2007**	**Flooding on farm**	**Gildenhuys pers. comm. 2019**
		1990	**Further land modification at site (cleared with tractor)**	**Gildenhuys pers. comm. 2019**
		1984	**Farmer Altus Gildenhuys took over farm**	**Gildenhuys pers. comm. 2019**
		1977	**Flooding on farm**	**Gildenhuys pers. comm. 2019**
		1969	Voëlvlei Dam wall raised – more water extracted from Klein Berg River	River Health Programme, 2004
		1960	**Stopped using donkeys – introduced farming machinery**	**Gildenhuys pers. comm. 2019**
		1952	Voëlvlei Dam built	River Health Programme, 2004
		1906	**Grandfather of current farmer Altus Gildenhuys bought farm**	**Gildenhuys pers. comm. 2019**
LIA (1400-1800)	Early agriculture	19th century	Grain and stock farming	River Health Programme, 2004
	Early agriculture	18th century	Stock farming in Swartland	River Health Programme, 2004

2.2 Sediment core collection and description

One sediment core from a floodplain adjacent the Berg River on Rhenostervlei Farm was collected in 2011. An aluminium pipe of 1.6 mm thickness, 7.62 cm in diameter and 3 m in length was used to core the site. Using a handle attached to the aluminium pipe, the core was pushed down into the sediment while minimising disturbance (Smith, 1987; Miller et al., 1991). Before removing the core, a rubber bung was inserted into the top of the pipe creating a vacuum to keep sediments in place. This facilitated the retrieval of a 120 cm core labelled RV3. The total length of core penetration into sediment was 195 cm, with sediment retrieval at 120 cm indicating a compaction of 38%. Once the core was retrieved, the excess piping was removed with a saw and the pipe ends were filled with plastic bags and sealed with tape. A surface sample was also collected by scooping up some sediment from the top 2-3 cm layer close to the coring site.

The core was then transported to the University of Cape Town where it was split in half lengthwise using an electric jig saw. Both halves were then wrapped in cling wrap and black plastic bags and were stored in a 4°C fridge to inhibit any microbial growth. One half served as a reserve while the other was subsampled for diatom analysis, lead-210 (^{210}Pb) and Accelerator Mass Spectrometry (AMS) radiocarbon dating. The core was then described based on grain-size distribution and organic content using the Troels-Smith scheme for stratigraphic notation to indicate changes in lithology (Troels-Smith, 1955). The sediment was also described based on colour changes using the Munsell (1954) colour chart.

2.3 Diatom analysis

Sub-samples of 1 cm^3 of sediment were collected from the sediment core for diatom analysis using a plastic medical syringe with the tip removed (Forbes, 2014). Sub-samples were taken at 2-8 cm intervals along the length of the core. Each sample was then transferred to a 50 ml conical screw cap tube. To isolate the diatoms, the sub-samples required treatment to remove extraneous materials such as salt, organic matter and minerogenic matter. Diatom extraction was achieved by following the systematic process (adapted from Battarbee, 1986 and further modified by Kirsten, 2014) outlined in the schematic in Figure 2.

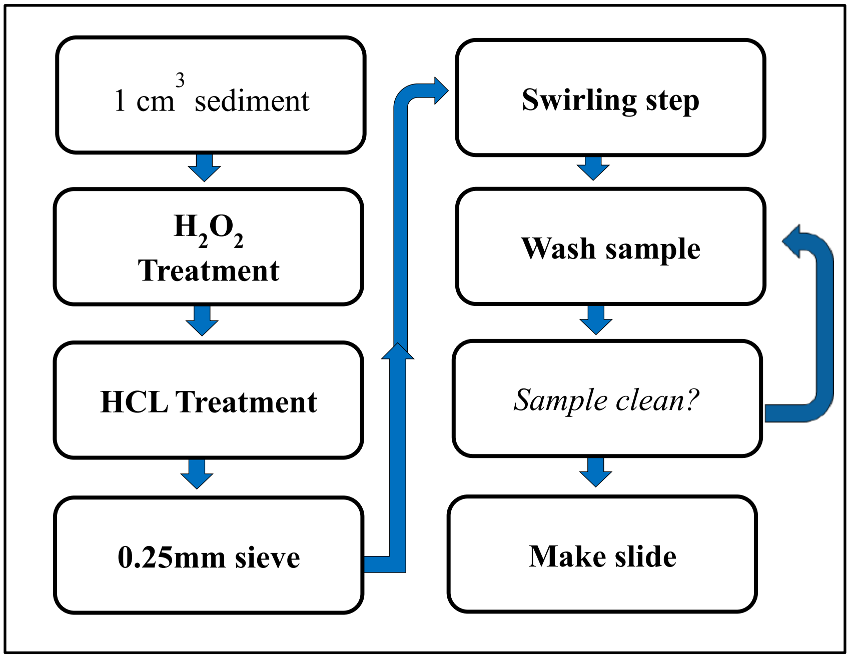

Figure 2: *Summary of laboratory procedure of sediments for diatom analysis (adapted from Battarbee, 1986 and further modified by Kirsten, 2014).*

To break down organic material within the sediment, the samples were treated with about 20 ml of 30% H_2O_2 while being heated gently in an 80 ^{0}C water bath. H_2O_2 was added in 5 ml aliquots and ethanol was used to stop samples from bubbling over. Samples were swirled intermittently over the course of the reaction which was continued for at least 3 hours to ensure all organic matter had been removed. Once the reaction was complete (samples stopped bubbling) distilled water was added to each sample to the 45 ml mark and was left overnight. The supernatant was then removed while taking care to not lose any sediment at the bottom of the test tube.

The samples were then treated with 10% HCl to remove all carbonates from the sediment. This was achieved by adding 10 ml HCl into each sample in an 80 ^{0}C water bath and swirling the test tubes intermittently. Another 10 ml was added if the carbonates were not fully digested. The reaction was considered to be complete when the liquid within the test tubes turned a luminous yellow colour and was close to transparent. Distilled water was added to each sample to the 45 ml mark and was left overnight. The supernatant was then decanted and the sample was sieved through a 0.25 mm mesh to remove coarse material, including plants, organic matter and sediments.

Samples were decanted into 100 ml beakers using as little distilled water as possible to clean out the test tube. Samples were then swirled for a few seconds before being decanted back into the test tube while leaving the coarse mineral matter in the beaker. Additional distilled water was added to the beaker and the swirling and decanting steps were repeated until the test tube was filled to 45 ml. To remove the finer clay particles, the samples were washed by subsequent refilling and decanting. Once samples were filled to the 45 ml mark, they were settled for 8 hours and then the supernatant was decanted with the fine clay particles in suspension being removed with it. This step was repeated until the supernatant appeared to be clear. Around 0.2 ml of sample was mounted on a slide using Pleurax (R.I. = 1.73) which allows for optimal visualisation of the diatom valves under the microscope.

After mounting the slide, diatoms were counted under a light microscope at X1000 magnification. For each level, a minimum of 300 diatom valves were counted. Fragments were only counted if they included the valve centre or another single characteristic feature as to avoid double counting. Identification of diatom species was based on the Taylor et al., (2007) diatom guide for South Africa as well as catalogues from internet resources (https://diatoms.org/; http://www.algaebase.org/). Initially, a sub-sample of 13 levels ranging over the full length of the sediment core was counted with identification of all diatom species present. Thereafter, only the most abundant diatom species (i.e. indicator species with at least 4% abundance in a level) were chosen for counting while the rarely occurring species with low abundance were grouped under a category called ‘Other’.

2.4 Chronology

The chronology for the RV3 core was determined using two dating techniques. Firstly, AMS radiocarbon dating was used to date one sample from the base of the core (depth = 108 cm). This sample was submitted to Beta Analytic Inc. Laboratory in Florida at the North American Facility Headquarters for radiocarbon dating. Since the core contained sediments from the last 100 years, it was then sampled for lead-210 (^{210}Pb) dating (Appleby et al., 1979). Thirty-three samples from 0-100 cm for lead-210 were analysed at the Core Scientific International in Canada. In total, 34 samples were sent for dating.

2.4.1 Accelerator mass spectrometry radiocarbon dating

The radiocarbon approach for dating utilises the rate of decay of the radioactive or unstable carbon isotope (^{14}C) in organic material (Walker, 2005). The rate of radioactive decay of the unstable isotope is constant; therefore by measuring the amount of ^{14}C remaining in a sample of fossil material and comparing to modern ^{14}C material, an age can be determined (Walker, 2005). The half-life of the radioactive isotope is ~5730 years, giving an upper dateable age limit of approximately 45 000 years (Walker, 2005).

2.4.2 Lead-210 dating

The lead-210 (^{210}Pb) age determination is based on the unstable isotope ^{210}Pb which forms one of the daughter nuclides in a series of nuclides in the Uranium-series decay chain involved in the decay of inert Radon gas (^{222}Rn). ^{210}Pb decays over a period of about 150 years to form the stable isotope ^{206}Pb. As this half-life is relatively short, this technique is used to date younger sediments that are under 150 years old (Bennion and Appleby, 1999). Assuming a constant atmospheric flux of ^{210}Pb, the time since the lead was deposited in a column of sediment related to depth can be ascertained by measuring the ratio of unstable ^{210}Pb to the stable ^{206}Pb. An important step in this dating technique is to determine supported ^{210}Pb (derived from the decay of uranium or its daughters) and subtract it from unsupported ^{210}Pb (produced in the atmosphere). The constant rate of supply (CRS) model (Appleby and Oldfield, 1978), which assumes that unsupported ^{210}Pb is being supplied to the sediments at a constant rate, was used to calculate the inputs of supported ^{210}Pb and unsupported ^{210}Pb, thus ensuring for accurate sediment age determination.

Ages resulting from the above mentioned dating methods were then combined to construct an age-depth model using the package called Bacon (Blaauw and Christen, 2013) in the statistical software program R (R Core Team, 2018). This approach to age-depth modelling uses Bayesian statistics to reconstruct Bayesian accumulation histories for deposits, by combining radiocarbon and other dates with prior information. The lead-210 ages were used with an added 1 year error without calibration while the radiocarbon age was calibrated with the with the SHCal13 calibration dataset for Southern Hemisphere terrestrial samples (Hogg et al., 2013). This calibration dataset is a combined set of dendrochronologically-dated records from sites in the Southern Hemisphere and can be confidently applied to ^{14}C measurements to account for the temporal difference between hemispheres (Hogg et al., 2013).

2.5 Statistical software and analysis

The diatom assemblage was analysed and displayed using the TILIA and TILIAGRAPH computer package programme (Grimm, 1997). TILIA and TILIAGRAPH present stratigraphic data against a primary y-axis of depth or age. Additionally, TILIA can perform a zonation function by stratigraphically constrained cluster analysis by incremental sum of squares (CONISS). CONISS analyses the species relative abundances throughout the core and determines whether adjacent levels are either similar or dissimilar thus creating different zones as indicated by a dendrogram. The significant zones revealed by this model can be used to pinpoint the timing of reorganization by the diatom assemblage in response to changing environmental conditions.

Additionally, a principal component analysis (PCA) was performed on the relative abundance data and the zones defined by CONISS, using the Vegan package (Oksanen et al., 2012) in R (R Core Team, 2018) PCA is used to identify the key species influencing patterns within the dataset and determining other underlying aspects of the data. In order to test the significance, a screen plot was plotted using the broken-stick model (MacArthur, 1957). The data was presented as a distance biplot based on the PCA scores.

3. RESULTS

3.1 Sediment description

The lithology for the RV3 core is presented in Figure 3a. The core consisted of different shades of brown with some areas of mottling (see Munsell colours in Figure 3a). The base of the core from 120-110 cm had some large pieces of ligneous material and was dominated by light brown coarse to fine sand with some gravel, and hence was not chosen for diatom analysis. After a thin layer of silt and fine sand with ligneous material and herbaceous roots, 109-105.5 cm consisted of a high amount of ligneous material and more fine sand with some silt. The next section of the core from 105.5-72 cm was dominated by clay of different shades of brown as a base colour mottled with red streaks and had a fair amount of charred ligneous material. The layer between 72-69 cm had a high amount of mottling and appeared as a red band which was mostly made up of silt with some fine sand. A layer of clay from 69-61cm which contained small amounts of silt and ligneous material was followed by a layer of mostly fine sand and some silt from 61-56 cm. The 56-48.5 cm section was predominantly clay with red mottling and the next section from 48.5-33.5 cm was mostly silt with some fine sand and was mottled with a red and yellow colour and had some charred ligneous material. From 33.5-6 cm, a layer of mostly fine sand and some silt occurred which had some yellow mottling. The surface layer from 6-0 cm consisted of fine sand and silt had pieces of charred ligneous material.

3.2 Chronology

AMS radiocarbon dating showed an age of 1790 CE at a depth of 108 cm (Table 2b). Measurable quantities of ^{210}Pb were found in the RV3 core between 0-90 cm (Table 2a). The radiocarbon and ^{210}Pb dates produced an age-depth model (Figure 3c) and suggest a consistent sediment accumulation (0.5 cm per year) from 108 cm to 92 cm, a slower sedimentation (0.28 cm per year) from 92 cm to 68 cm, followed by a higher sediment accumulation (0.80 cm per year) from 68 cm to 0 cm (Figure 3b). The sedimentation appeared to be continuous with no reversals.

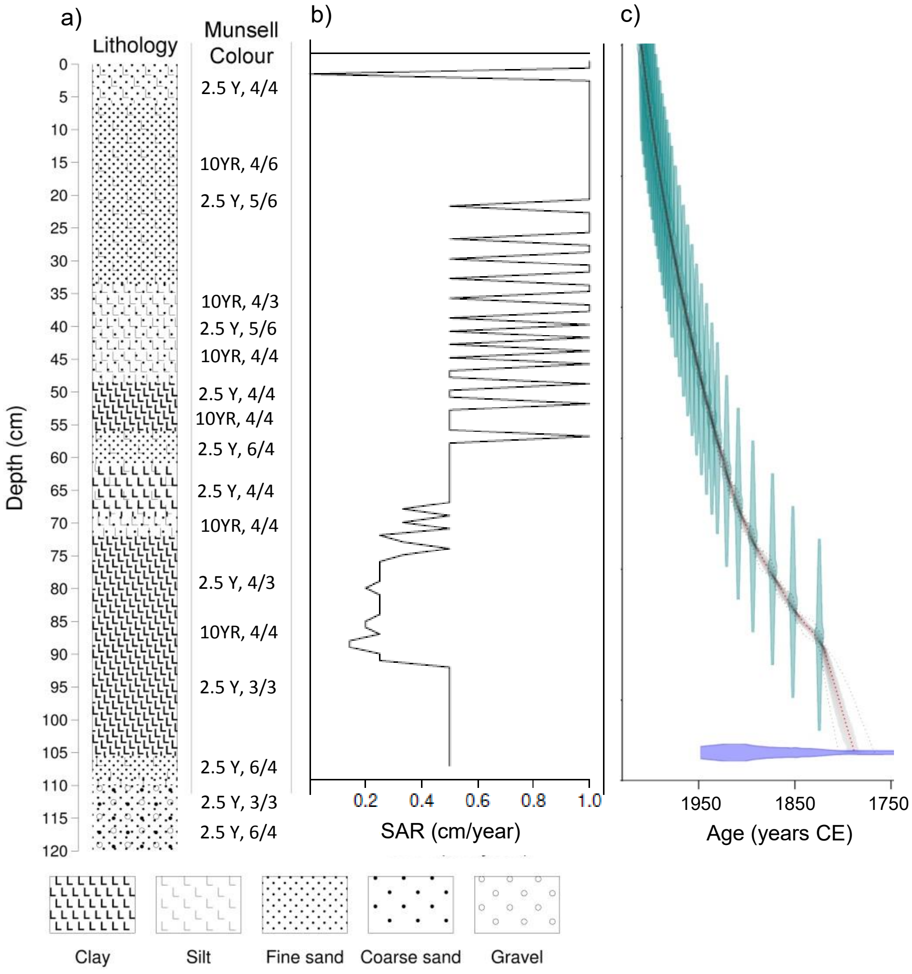

Figure 3: *Description of RV3 core, including:* ***a)*** *lithology based on Troels-Smith notation and the Munsell colour chart,* ***b)*** *sediment accumulation rate (SAR) cm/year against depth and* ***c)*** *age-depth model for samples from 0 to 108 cm with the date that the core was extracted (2011 CE), 31* ^{210}*Pb dates and one radiocarbon date.*

Table 2: *Results obtained from* ***a)*** ^{210}*Pb dating for 33 samples from depths 0 to 90 cm and* ***b)*** *AMS radiocarbon dating for one samples from the RV3 core.*

a)

Sample depth (cm)	Age at Bottom of Extrapolated Section in yr BP (constant rate of supply CRS model estimate)	CRS Sediment Accumulation Rate (g/cm^2/yr)	^{210}Pb Total Activity (DPM/g)	^{210}Pb Unsupported Activity (DPM/g)	Error ^{210}Pb +/- 1 S. D. (DPM/g)	Calendar years (CE)
0-2	1.60	0.271	13.772	12.652	0.47	2009
2-4	3.29	0.266	13.562	12.442	0.47	2008
4-6	5.11	0.260	13.530	12.410	0.47	2006
6-8	6.95	0.258	12.608	11.488	0.45	2004
8-10	8.93	0.254	12.507	11.387	0.45	2002
10-12	10.90	0.253	11.593	10.473	0.43	2000
12-14	12.88	0.252	10.886	9.766	0.42	1998
14-16	14.93	0.251	10.383	9.263	0.41	1996
16-18	17.07	0.250	10.005	8.885	0.4	1994
18-20	19.26	0.249	9.421	8.301	0.39	1992
20-22	21.57	0.247	9.170	8.050	0.38	1989
22-24	23.93	0.245	8.628	7.508	0.37	1987
24-26	26.34	0.244	8.048	6.928	0.36	1985
26-28	28.84	0.243	7.694	6.574	0.35	1982
28-30	31.44	0.241	7.325	6.205	0.34	1980
30-33	35.34	0.239	6.685	5.565	0.33	1976
33-36	39.39	0.237	6.164	5.044	0.32	1972
36-39	43.58	0.235	5.634	4.514	0.3	1967
39-42	48.00	0.232	5.238	4.118	0.29	1963
42-45	52.53	0.230	4.745	3.625	0.28	1958
45-48	57.26	0.227	4.385	3.265	0.27	1954
48-51	62.31	0.223	4.086	2.966	0.26	1949
51-54	67.64	0.220	3.757	2.637	0.25	1943
54-57	73.16	0.216	3.384	2.264	0.24	1938
57-60	78.89	0.213	3.076	1.956	0.22	1932
60-65	88.49	0.208	2.636	1.516	0.21	1923
65-70	100.52	0.200	2.453	1.333	0.2	1910
70-75	115.77	0.189	2.216	1.096	0.19	1895
75-80	136.20	0.173	1.959	0.839	0.18	1875
80-85	157.39	0.161	1.566	0.446	0.16	1854
85-90	184.92	0.147	1.391	0.271	0.15	1826

b)

Sample depth (cm)	Radiocarbon date (yr BP)	Calibrated date (AD)	Probability (Standard deviation and %)	Chosen age (CE)
108	118 ± 30	1806-1951	2 SD (95%)	1790

3.3 Diatom results

The initial 13 levels which ranged over the full length of the core revealed total species richness of 180 diatom species. Most levels were highly diverse and included many species with low occurrences. Therefore, 20 indicator diatom species with the highest abundance (at least 4% relative abundance in a level) were selected for counting while the rarely occurring species with low abundance were from this point grouped into a rare species category called ‘Other’. These species made up approximately 61% of the total diatom counted. The full scientific name and example images of the 20 selected diatom species are found in the Appendix, Table 1. The top 20 diatom species and their abundances (%) against depth and age can be viewed in Figure 4 and 5. A total of 37 samples were processed of which 35 samples were countable (reached the minimum count of 300 diatom valves). The two levels that were not included in the analysis, depths of 100 cm and 96 cm (Figure 5 and 6), had low preservation potential and the minimum count was not reached. The most abundant species was *Aulacoseira granulata* which had a total abundance of 12.6% of all diatoms counted. The other most abundant species in ascending order were: *Navicula rostellata* (4.7%), *Tryblionella levidensis* (3.7%), *Cyclotella meneghiniana*, *Pinnularia borealis* (3.6%) and *Gomphonema parvulum* (3.5%).

Based on CONISS (Grimm, 1987), the RV3 core assemblage can be divided into two major units: Zone 1 (approximately 108-78 cm, 1790-1890 CE); and Zone 2 (approximately 78-0 cm, 1890-2011 CE), which is further subdivided into Zone 2a (approximately 78-54 cm, 1890-1945 CE) and Zone 2b (approximately 54-0 cm, 1945-2011 CE) (Figure 4 and 5). The changing diatom abundance in each zone and the ecology of some key species are described below.

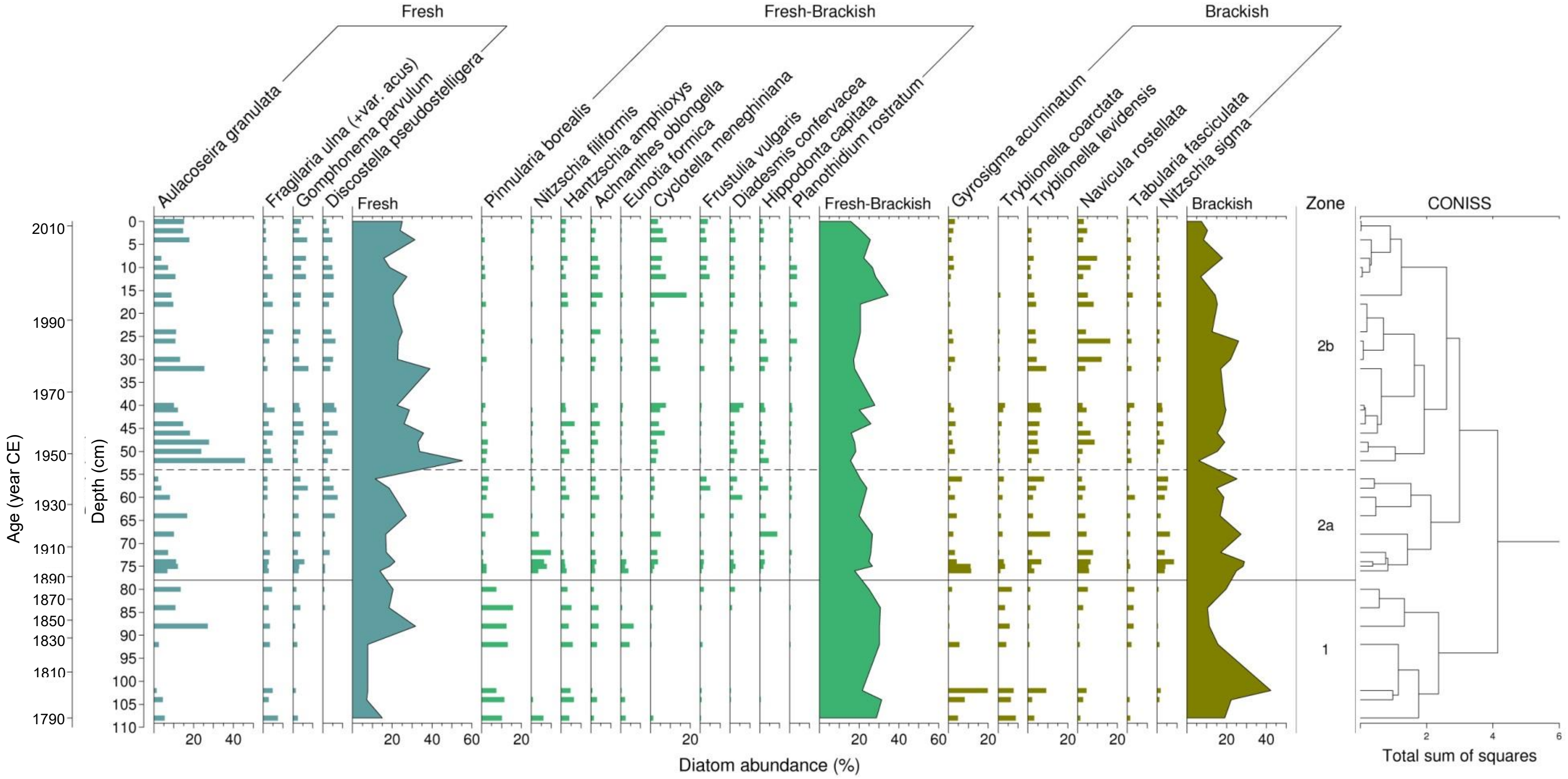

Figure 4: *The RV3 core percentage diagram for selected indicator diatom species against depth (cm) and age (year CE). The diatom species are grouped based on salinity preferences and the total species in each salinity preference as shown. Three zones (1, 2a and 2b) were identified based on CONISS.*

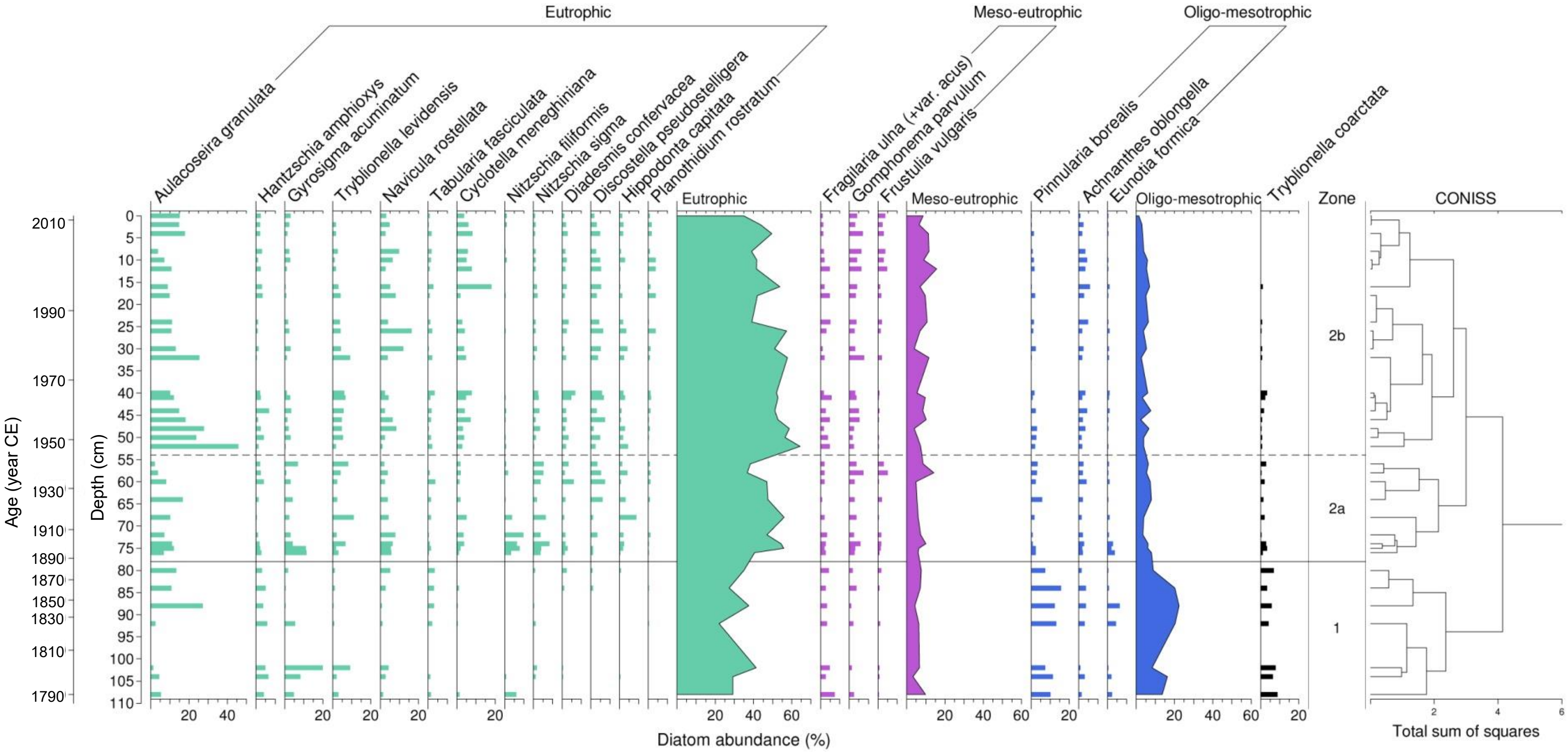

Figure 5: *The RV3 core percentage diagram for selected indicator diatom species against depth (cm) and age (year CE). The diatom species are grouped based on nutrient preferences and the total species in each nutrient preference as shown. Three zones (1, 2a and 2b) were identified based on CONISS*

Zone 1 (7 subsample depths; 108-78 cm)

Zone 1 included 7 subsample depths and dated from ca. 1790-1890 CE (108-78 cm). This zone displayed a varied presence of the freshwater and mostly planktonic species, *Aulacoseira granulata.* This species abundance appeared to be particularly low (1.3-5.3% range) in depths 108-92 cm (c.a 1790-1820 CE) but then increased (10.7-27%) between depths 88-80 cm with a high abundance at depth 88 cm (ca. 1850 CE). *Pinnularia borealis*, an aerophilous taxon, remained relatively constant in Zone 1 (7.3-15.7%) reaching a peak of 15.7% at depth 84 cm (c.a 1860 CE). The less abundant fresh-brackish and aerophilous species, *Hantzschia amphioxys,* also had a consistent presence within this zone but at a lower percentage (3-6.3%). *Gyrosigma acuminatum* had a variable presence in Zone 1 (0.3-19.7%) but showed a higher abundance from 108-102 cm (c.a 1790-1800 CE) and peaked at 102 cm (c.a 1800 CE). Other brackish benthic species, *Tryblionella coarctata* and *Tryblionella levidensis,* also appeared to be more abundant between 108-102 cm. *Tryblionella coarctata* remained relatively constant through-out this zone (3.3-8.7%) and ranged from 6.3-8.7% between 108-102 cm. *Tryblionella levidensis* was less abundant in Zone 1 but did peak at 9% at 102 cm (c.a 1800 CE). The oligotrophic species, *Eunotia formica* although not consistent had a slight presence in this zone (0.3-6.3%).

Species ecology:

Aulacoseira granulata is a freshwater diatom which is primarily planktonic but also can be found in epipelic and epilithic habitats and seems to prefer turbulent environments (Vos and De Wolf, 1993; Sonneman et al., 2000). This species is common in eutrophic lakes and rivers (Taylor et al., 2007). *Aulacoseira granulata* is found in eutrophic waters and is particularly common in rivers and shallow floodplain lakes with high water turbulence (Kilham and Kilham, 1975) and also has been shown to be associated with increased turbidity (Costa-Böddeker et al., 2012). Although its principal habitat is the euphotic zone, this species is often characterised as tychoplanktonic as it sometimes sinks out of the water column during periods of weakened seasonal mixing where it survives in a dormant state until it’s re-suspended when turbulent conditions return (Kilham and Kilham, 1975; Stager et al., 2013). *Pinnularia borealis* is cosmopolitan aerophilic species, found on rocks, soils and moss in mesotrophic rivers and lakes (Vos and De Wolf, 1993; Taylor et al., 2007). *Hantzschia amphioxys* is also aerophilous, favouring periodically dry habitats, including soils and rock crevices and is found in eutrophic waters (Vos and De Wolf, 1993; Taylor et al., 2007). This species is widespread in a range of rivers, but is probably washed in from soils (Taylor et al., 2007). Both *Pinnularia borealis* and *Hantzschia amphioxys* are considered to be oligohalobous indifferent meaning they are indifferent to salinity – can live in brackish and fresh waters (Vos and De Wolf, 1993). *Eunotia formica* is a fresh-brackish species found in standing or slow flowing oligotrophic waters and sometimes on wet surfaces (van Dam et al., 1994; Taylor et al., 2007). The brackish species *Gyrosigma acuminatum* and *Tryblionella levidensis* can be found in nutrient-enriched environments and are epipelic; often living on finer sediments within the littoral zone (Vos and De Wolf, 1993). *Tryblionella coarctata* is a cosmopolitan benthic species found in brackish alkaliphilic waters (Taylor et al., 2007). This species nutrient preference is unknown.

Zone 2 (28 subsample depths; 78-0 cm)

- *Zone 2a (9 subsamples; 78-54 cm)*

The subdivision Zone 2a consisted of 9 subsample depths, dating from ca. 1890-1945 CE (78-54 cm). In this zone, *Aulacoseira granulata* initially remained at a relatively high abundance (6.7-16.7%) between depths 78-64 cm (ca. 1890-1925 CE) and peaked at depth 64 cm (ca. 1925 CE), after which its abundance progressively decreased to 2% at depth 56 cm (ca. 1940 CE). The freshwater species *Discostella pseudostelligera* (0-7.3%) and *Gomphonema parvulum* (2.3-7.3%) appeared more prominently in Zone 2a compared to Zone 1. *Discostella pseudostelligera*, another planktonic species, reflected the same progressive decrease as *Aulacoseira granulata* from 60 cm ending at 3.3% at depth 56 cm (ca. 1940 CE). *Cyclotella meneghiniana*, the planktonic species which prefers a more fresh-brackish environment, also was more prominent in Zone 2a although it remained at low abundance (1-5%). This trend was observed for other fresh-brackish and nutrient tolerant species such as epiphytic *Planothidium rostratum* (0-1%), *Diadesmis confervacea* (1-6%), and *Hippodonta capitata* (0.3-8.7%). The latter species had its highest peak throughout the core in Zone 2a peaking at 68 cm (ca. 1920 CE). The brackish nutrient tolerant species, *Nitzschia sigma* (1.7-8.3%), *Navicula rostellata* (2-7.7%) and *Tryblionella levidensis* (1.7-11%) were also more abundant in Zone 2a. *Nitzschia sigma* and *Navicula rostellata* stayed more or less constant while *Tryblionella levidensis,* had a more variable abundance, and peaking at 11% at 68 cm (ca. 1920 CE). The benthic brackish species, *Gyrosigma acuminatum* (2.3-11.3%) appears to be more consistently present in Zone 2a however it never reached as high a peak as it did in Zone 1 (19.7%). *Nitzschia filiformis* showed a discrete presence in Zone 2a concentrated between 76-68 cm (ca. 1895-1920 CE) ranging from 3.3-9.7%. The abundance of *Pinnularia borealis* (0.7-5.7%) displayed a distinct decrease compared to that of Zone 1 (7.3-15.7%). A more subtle decrease was noted for the abundance of aerophilous *Hantzschia amphioxys* (0.3-4%) compared to that of Zone 1 (3-6.3%). *Tryblionella coarctata* (1-3.3%) appears at a lower abundance than Zone 1.

- *Zone 2b (1945-2011 AD; 54-0 cm)*

The subdivision Zone 2b consists of 19 subsample depths, dating from ca. 1943-2011 CE (54-0 cm). *Aulacoseira granulata* appears to be most prominent in Zone 2b (3.7 - 45.7%), displaying a drastic peak in abundance just after the onset of subdivision Zone 2b at 45.7% at depth 52 cm (ca. 1945 CE) followed by a reasonably steady decrease as seen from depths 50-40 cm (ca. 1950-1965 AD). This species then showed a moderate peak of 25.3% at depth 32 cm (ca. 1975 CE), after which it remains more or less consistent throughout the zone. *Cyclotella meneghiniana* also appeared to be more prominent in Zone 2b (1.7 - 18%), reaching its highest peak of 18% at depth 16 cm (ca. 1995 CE). The brackish eutrophic species, *Navicula rostellata,* had a variable abundance (2.3-16.3%) but also appeared to be more prevalent when compared to Zone 2a with a peak of 16.3% at depth 26 cm (ca. 1985 CE). The other freshwater species, *Discostella pseudostelligera* (1.3-7.3%) and *Gomphonema parvulum* (1.7-7.7%), were more or less consistently present throughout Zone 2b, remaining at similar abundances as in Zone 2a. *Tryblionella levidensis* (0-9%) remained at similar abundances in Zone 2b but did appear to gradually decrease after 32 cm (ca. 1975 CE). The abundance of *Gyrosigma acuminatum* (0.3-3.3%) appeared to decrease in this Zone compared to Zone 2a while *Tryblionella coarctata* had a very low abundance and actually disappeared from the assemblage after 16 cm (ca. 1995 CE). *Nitzschia sigma* (0.3-3%), presented a general decrease in abundance compared to Zone 2a. *Pinnularia borealis* (0-3%) had consistently low abundances in Zone 2b and seemed to be at slightly lower levels compared to Zone 2a.

Species ecology:

Cyclotella meneghiniana is planktonic and considered to be oligohalobous halophilous meaning it is a salt-tolerant freshwater species (Vos and De Wolf, 1993). *Discostella pseudostelligera* is primarily planktonic but also can be benthic, tychoplanktonic, epilithic, or epipelic (Reynolds et al., 2002). This species typically occurs in shallow, nutrient-enriched, well-ventilated, and often turbid freshwater aquatic habitats (Reynolds et al., 2002). *Gomphonema parvulum* is a freshwater species and has been classified as a benthic epiphyte; living attached to macroalgae or water plants (Vos and De Wolf, 1993). *Gomphonema parvulum* has a broad tolerance, particularly for nutrients, and is considered to be tolerant of extremely polluted conditions (Sonneman et al., 2000; Taylor et al., 2007). *Hippodonta capitata* is a benthic species which can be epipelic or epilithic (Sonneman et al., 2000) which is often found in fresh-brackish, eutrophic waters and is tolerant of critical pollution levels (Taylor et al., 2007). *Diadesmis confervacea* is cosmopolitan species found in a range of waters, including eutrophic, electrolyte rich and extremely polluted waters (Taylor et al., 2007). *Nitzschia sigma* is a brackish epipelic species found in eutrophic, electrolyte-rich inland waters (Vos and De Wolf, 1993; Taylor et al., 2007). *Navicula rostellata* is a cosmopolitan eutrophic epipelon species that is known to be tolerant of critical levels of pollution

(Vos and De Wolf, 1993; Taylor et al., 2007). *Nitzschia filiformis* found in waters of moderate to high electrolyte content also extending into brackish waters and is able to tolerant strongly polluted conditions, but not of critical levels of pollution (Taylor et al., 2007).

When the percentage abundance of *Aulacoseira granulata* was plotted against rainfall data from Rhenostervlei Farm, there appeared to be a general correlation between the two variables, though this wasn't significant ($R^2 = 0.24$, p-value > 0.5) (Figure 6).

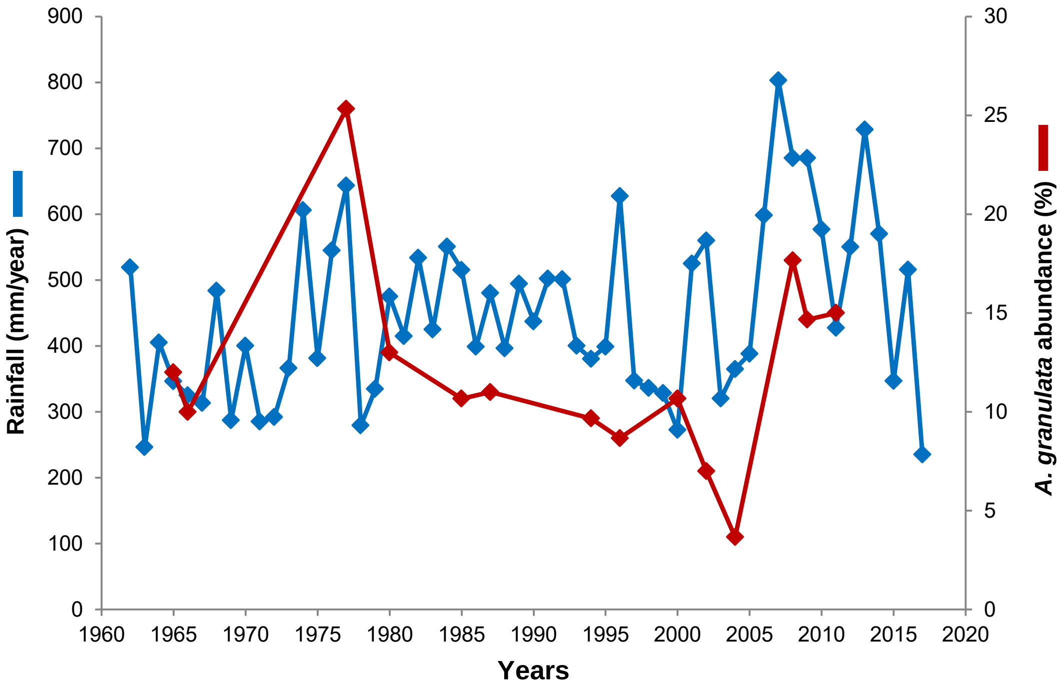

Figure 6: *Rainfall (mm/year) data collected from Rhenostervlei Farm from 1962 – 2017 plotted against the abundance (%) Aulacoseira granulata in the RV3 core.*

3.4 Principle Component Analysis (PCA) results

Two principal component axes were determined to be significant, contributing 41.99% to the overall explained variance. Component one, which explains 26.76% of the variance, has large positive loadings on the brackish benthic species, *Tryblionella coarctata* and aerophilic species, *Pinnularia borealis* and *Hantzschia amphioxys*. This component had large negative loadings on more freshwater and planktonic diatom species, *Discostella pseudostelligera*, *Gomphonema parvulum* and *Cyclotella meneghiniana*. The positive loading on brackish and dry tolerant species and negative loading on freshwater and planktonic species indicate that the primary control on the diatom community and the first component of the PCA is related to be the availability of moisture and freshwater influence at the site.

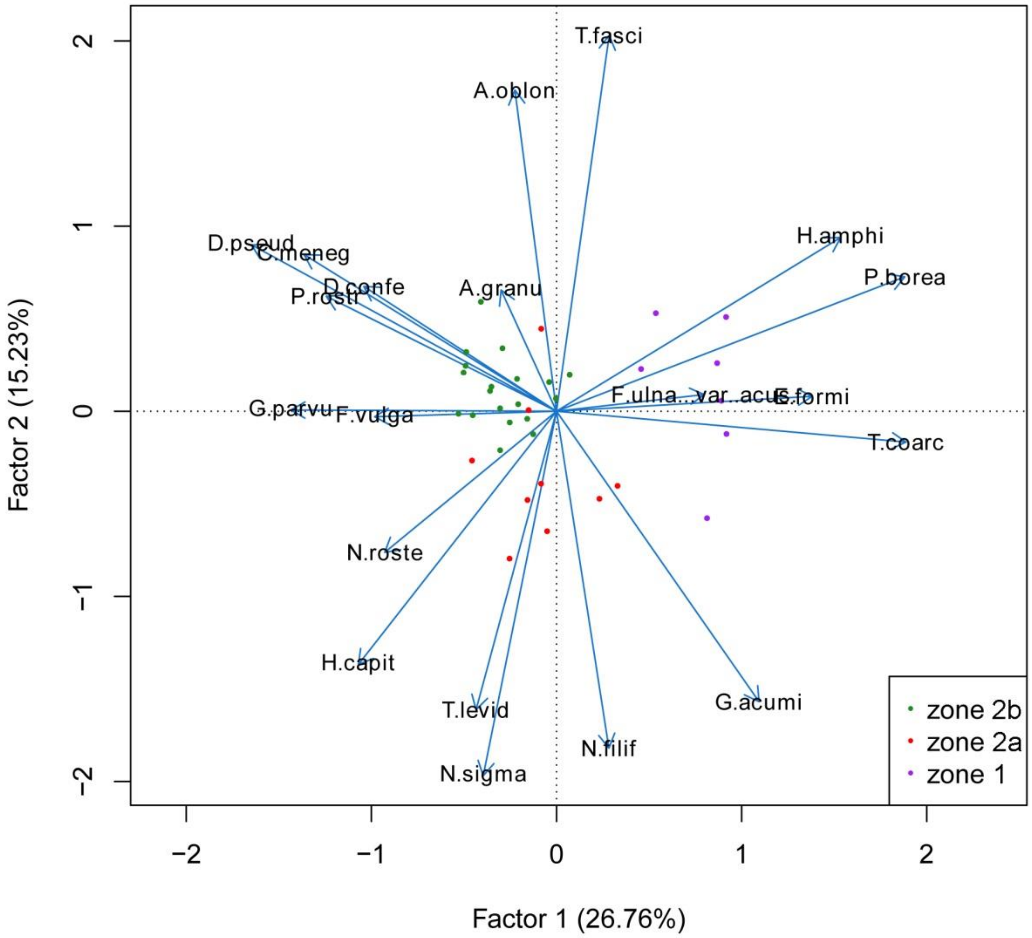

Figure 7: *Principal component analysis representing the relationship between the 20 indicator diatom species and samples of the RV3 core. Sample points are colour coded based on zones identified by CONISS in TILIA.*

Component two, which explains 15.23% of the variance, had large positive loadings on *Tabularia fasciculata* and *Achnanthes oblongella. Tabularia fasciculata* is a cosmopolitan species with a broad ecological amplitude, however this taxon does appear to favour moderately to high electrolyte concentrations (Taylor et al., 2007). This species is often recorded in both epipelic and epilithic assemblages (Sonneman et al., 2000). *Achnanthes oblongella* is often occurs in small, circumneutral, oligotrophic, electrolyte poor streams (Taylor et al., 2007) and is commonly recorded in epilithic (grown on rock surface) assemblages (Sonneman et al., 2000). This species prefers nutrient poor environments

(Battegazzore et al., 2004), and is typically found in the upper reaches of an estuary or in streams where flash flooding is a common occurrence (Hodgson et al., 1996; O'Driscoll et al., 2012) The region appears to experience more high-intensity rain events, with species indicative of flash flooding, for example, *Achnanthes oblongella* (Hodgson et al., 1996; O'Driscoll et al., 2012). This component had large negative loadings on *Nitzschia sigma, Nitzschia filiformis, Tryblionella levidensis* and *Gyrosigma acuminatum.* These species are brackish, eutrophic and benthic either on epipelon or epilithon (Vos and De Wolf, 1993). There doesn't appear to be a clear driving mechanism for component two. It could potentially indicate a nutrients related mechanism controlling the diatoms abundance due the oligotrophic preference of *Achnanthes oblongella* and the eutrophic preference for the species with large negative loads. This mechanism is not clear and is interpretation is particularily hindered due to the limited autecological information available for *Tabularia fasciculata.*

4. DISCUSSION

4.1 Validity of diatoms as environmental indicators at Rhenostervlei Farm

Diatoms were consistently present throughout the record from ca. 1790-2011 CE, except for two levels corresponding to ca. 1810 CE and ca. 1820 CE, where preservation was poor. The chosen 20 species for analysis constituted the majority (61%) of the total diatom abundance and thus were considered as adequate indicators for this environment. This suite of indicator species encountered, including *Aulacoseira granulata, Navicula rostellata, Tryblionella levidensis, Cyclotella meneghiniana, Pinnularia borealis* and *Gomphonema parvulum* reflect a range of environmental conditions from freshwater to brackish, low moisture (aerophilic species) to high moisture (planktonic species) availability and nutrient-poor (oligotrophic species) to nutrient-rich (eutrophic species). Thus, the study successfully establishes the utility of diatoms as biological indicators for reconstructing change in a range of environmental variables for the study site, Rhenostervlei Farm.

4.2 Diatom-based reconstruction of water quality and availability over ca. 220 years

The study site is a damp low-lying area adjacent to the Berg River, which is formed mainly of river sediments and is subject to flooding when water breaches the river banks. Sources of freshwater to the site may occur through groundwater flow, seepage, precipitation with resultant surface runoff or river flow, which sustains the moist environment. Throughout the Berg River catchment, evaporation exceeds precipitation and the river water budget is dominated by runoff (De Villiers, 2007).

The benthic species encountered in the study (e.g. *Pinnularia borealis* and *Gomphonema parvulum*) were probably locally derived species, representing the autochthonous (local), baseline community which lived at or near the sampled location, thus making them ideal indicator organisms for water quality and other environmental conditions present at Rhenostervlei Farm. The planktonic species in the study (*Aulacoseira granulata*, *Cyclotella meneghiniana* and *Discostella pseudostelligera*) however, were likely introduced to the site by flooding or higher water levels. Furthermore, these species could have been deposited after being carried in the river from further afield (allochthonous). Therefore, the presence of planktonic species was considered as possibly providing information on a larger spatial scale of the Berg River with at least some of the ecological interpretation from these species representing conditions up-gradient from the sample location (Reavie and Edlund, 2010).

4.2.1 Salinity and freshwater influence reconstructions

During ca. 1790-1890 CE the dominance of aerophilous taxa such as *Pinnularia borealis* and *Hantzschia amphioxys* (see Figure 4, Zone 1), was interpreted as the site remaining relatively moist with minimal freshwater supply or having shallow standing waters that were prone to occasional desiccation (van Dam et al., 1994; Taylor et al., 2007). The limitation of freshwater inputs and the presence of standing waters during this period was also supported by the relatively high abundance of brackish species (such as *Gyrosigma attenuatum, Tryblionella levidensis* and *Tryblionella coarctata*), indicating more saline conditions at the site. The presence of brackish species may be a consequence of standing puddles being more concentrated and saline which may occur due to evaporative loss during warmer periods, a lack of diluting effect of freshwater inputs either from rainfall or river overflow (decrease in the *P-E* ratio).

A period of particularly high salinity in the lower portion of the core between ca. 1790-1800 CE was reflected in the notably high peak in the halophilous taxon, *Gyrosigma attenuatum* and the higher abundances of brackish species, *Tryblionella levidensis* and *Tryblionella coarctata*. In a previous diatom-based environmental reconstruction study, Gell et al. (2007) also interpreted high abundance of *Gyrosigma attenuatum* as indicative of more saline conditions in Loveday Wetland, South Australia.

In the present study, the period of high salinity was also supported by the low abundances of freshwater and planktonic or benthic species including the mostly planktonic *Discostella pseudostelligera* and *Aulacoseira granulata* and the benthic *Gomphonema parvulum.* The slight presence of *Gomphonema parvulum* and *Aulacoseira granulata* during this period of apparent saline conditions may be a result of their oligohalobous-indifferent nature meaning that even though their preference is predominantly freshwater, they can still tolerate higher salinities (Vos and De Wolf, 1993). The relatively low abundance of freshwater and planktonic species during this time period ca. 1790-1800 CE may indicate that the site was receiving low freshwater inputs from rainfall and/or river overflow. There may have been some river over flows after ca. 1840 CE as indicated by the rise in *Aulacoseira granulata* which could reduce the salinity of the standing puddles and explain the decrease in brackish species during this time.

The diatoms in this study indicated that the environment at the floodplain site during ca. 1790-1890 CE was dominated by damp marsh conditions with possibly muddy sediments and moss cover with limited freshwater pulses. Water at the site was likely in the form of standing puddles which were prone to evaporation and ultimately, desiccation. These concentrated shallow isolated ephemeral pools of water without freshwater inputs could support the brackish diatoms. River overflows possibly started at this site after ca. 1840 CE and could explain the decreased dominance in brackish species corresponding to that period.

Since the ca. 1890s CE (Zone 2) the diatom assemblage at the site was characterised by an increased appearance of freshwater species including the epiphytic species *Gomphonema parvulum* and planktonic species *Discostella pseudostelligera*, as well as freshwater/brackish, planktonic species, *Cyclotella meneghiniana*. This indicates a higher freshwater influence and possibly wetter conditions, which may have involved a shift from a damp marsh condition with isolated short-lived pools to the presence of longer persisting larger interconnected pools of water – a change that may have resulted in the decrease of dry tolerant aerophilic species which mainly occur on wet and moist or temporarily dry places (van Dam et al., 1994). Since the fresh-brackish and brackish species still persisted, we can assume that these pools continued to experience successive evaporation but perhaps at a slower rate – therefore supporting either fresh-brackish or brackish depending on the amount of evaporation. The potential mechanism of availability of moisture and freshwater influence as a primary control of diatom abundance at the site is supported by the PCA results (see Figure 7) which showed a shift from brackish and dry tolerant species to freshwater and planktonic species.

Another freshwater species which was interpreted as indicating freshwater influence was *Aulacoseira granulata*. This species is often characterised as tychoplanktonic as it sometimes sinks out of the water column where it survives in a dormant state until it's re-suspended when turbulent conditions return (Kilham and Kilham, 1975; Stager et al., 2013). This species displayed a highly variable presence and since its occurrence may be linked to turbulent conditions, its presence was interpreted as indicating periodic flooding events in which high water levels deposit this species at the site adjacent the river. The Berg River is characterised by low flows in summer and higher flows in winter in which flooding occurs as regular events (G. Howard, Ninham Shand, pers. comm. Reported by Snaddon, 2009). Therefore, this species high abundance from the onset of Zone 2b between ca. 1945-1950 CE, could indicate a period of high water levels and flooding. After this peak it appears to show a general decreasing trend between ca. 1950-2005 CE besides a peak around ca. 1975 CE. This could possibly indicate a period of decreasing water levels with less river overflows.

4.2.2 Nutrients and turbidity reconstructions

In addition to the diatom-inferred change in salinity and freshwater influence, the diatom assemblage at the site also potentially indicated a change in the nutrient status starting at ca. 1890 CE. The dominance of high nutrient tolerant benthic (e.g. *Navicula rostellata*, *Nitzschia sigma*, *Diadesmis confervacea* and *Hippodonta capitata*) and planktonic (*Aulacoseira granulata*, *Cyclotella meneghiniana* and *Discostella pseudostelligera*) species from ca. 1890-2011 CE (see Figure 5, Zone 2) may be interpreted as a shift to more eutrophic conditions at the site and within the river at Rhenostervlei Farm. As well as indicating freshwater influence and nutrient enrichment, the higher abundance of planktonic species, *Discostella pseudostelligera* may also indicate turbid conditions in the river. There is also possible evidence of turbidity, river turbulence and nutrient enrichment being higher after ca. 1940 CE in subsection Zone 2b (ca. 1945-2011 CE) as indicated by peaks in eutrophic and turbidity indicator, *Aulacoseira granulata.* This zone also displayed peaks in other eutrophic species including planktonic *Cyclotella meneghiniana* and the benthic *Navicula rostellata* indicating nutrient enrichment in the river and floodplain habitat, respectively.

4.3 Drivers of change in water availability and quality

The presence, composition and abundance of diatom assemblages are directly affected by various low-level factors such as resources (e.g. light, nitrogen, phosphorus), stressors (e.g. pH, temperature, toxic substances) hydraulic conditions, and biotic interactions (Larned, 2010). However, the impacts of these low-level factors are predominately controlled by higher-level factors such climate and land-use (Stevenson, 1997).The study period of ca. 220 years covers significant known changes in climate (Altwegg et al., 2014) and land-use (River Health Program, 2004), which would have influenced water quality and availability at Rhenostervlei Farm and the Berg River catchment as a whole.

4.3.1 Climate

Climatic events captured in the study period (ca. 220 years) include the end of the Little Ice Age (LIA) which occurred from ca. 1400-1800 CE (Nicholson et al., 2013) as well as the 20th century warming (Cronin et al., 2003; Haensler et al., 2010). Previous palaeoclimatic studies in the southern African region suggested a cold, wet LIA (e.g. Lee-Thorp et al., 2001; Holmgren et al., 2003; Stager et al., 2012). For the WRZ of South Africa, Stager et al. (2012) inferred a wet LIA from the diatom record from Lake Verlorenvlei with precipitation maxima occurring around ca. 1350, 1420, 1480, 1620, 1750, 1860 and 1930 CE.

In contrast, the sediment record from Rhenostervlei Farm indicated that the time period corresponding with the LIA, ca. 1790-1800 CE, was dominated by dry tolerant and salinity tolerant species - indicating minimal freshwater influence. This signals a dry environment or standing puddles which were vulnerable to evaporative loss either due to reduced rainfall or increased temperatures. Since high temperatures in the LIA would have been unlikely, the dry, saline signal detected by the diatoms around ca. 1790-1800 CE may indicate a drier climate or could possibly suggest that climate (specifically increased rainfall of the LIA) has less of an influence on the diatom assemblage during this period. This could be due to the small spatial scale of this specific site and therefore land-use change may play a bigger role than climate in determining changes in local hydrology.

Historical records from the CFR have shown a warming trend over the 20th century (Altwegg et al., 2014), which is particularly prevalent in the latter part of the century (Cronin et al., 2003; Haensler et al., 2010). This trend was seen in the WRZ in which temperature were shown to increase from 1960-2003 using timelines from six weather stations (Kruger and Shongwe, 2004). Under 20th century warming, increasing temperatures could have increased evaporation and decreased water balance (decreased *P-E* ratio) with a potential increased salinity signal being observed in the latter section of the diatom record. However, the diatom record of this study did not indicate a clear trend of increasing brackish species over the 20th century. The instrumental record of rainfall from 1962-2017 CE from Rhenostervlei Farm (Figure 6) did not show an increasing trend and therefore could not offset the enhanced evaporative demand that would be associated with increasing temperature. The lack of salinity signal in diatoms may be due to extensive rainfall variability over-ridding this signal or land-use in the form of vegetation removal which would increase runoff leading to more freshwater influence at the site.

Aulacoseira granulata was used as a potential indicator of climatic variation in the study. This species was likely introduced to the site by river overflows. Since precipitation (frequency and intensity) is a main factor in determining the river discharge, i.e. the volume of water flowing through a river channel (Vandenberghe, 2002), the abundance of *Aulacoseira granulata* could be linked to flooding events driven by the prevailing precipitation. This apparent relationship between this mostly planktonic diatom and precipitation has been utilised in various studies (Stager and Mayewski, 1997; Stager et al., 2012, 2013). For example, Stager et al. (2012) interpreted elevated percentages of *Aulacoseira granulata* to represent periods of increased runoff and river inputs to the Lake Verlorenvlei under relatively wetter climatic conditions in the WRZ South Africa over the last 1400 years.

When comparing the percentage abundance of *Aulacoseira granulata* to rainfall data collected since the 1960s on Rhenostervlei Farm, a similar trend was distinguished in which peaks, troughs and level trends in rainfall were generally followed by that of *Aulacoseira granulata* abundance (Figure 6). For example, during a known flood event due to high rainfall on Rhenostervlei Farm in 1977 (Gildenhuys pers. comm. 2019), a reflected increase in *Aulacoseira granulata* abundance in the sediment record was observed. However, a significant correlation could not be found, indicating that other environmental variable could be controlling *Aulacoseira granulata* abundance.

4.3.2 Land-use

Anthropogenic activities can often alter or obscure the climatic signal present in the sedimentary record (Wolin and Duthie, 1999). Similarly, this study's relatively short time-scale and its position within an area which is heavily impacted by anthropogenic stressors (i.e. agricultural development), may have concealed the long-term climatic signal, with changes in water quality and availability as indicated by diatom abundance predominantly reflecting the land-use impacts felt at the site. The land-use driver of water quality and availability could involve anthropogenic landscape modification and changes associated with agricultural intensification.

The Berg River catchment has a long history of agricultural activities (River Health Programme, 2004). Although detailed information about the long-term history of land-use at Rhenostervlei Farm is limited prior to the 20th century, historical records of the Swartland region indicate that in the 1700s CE farmers started stock grazing and later during the 1800s, started farming grain (River Health Programme, 2004). We can then assume that there were agricultural activities on Rhenostervlei Farm during ca. 1790-1890 CE (within Zone 1). Therefore, the period of high salinity between ca. 1790-1800 CE may be explained, by the site being less prone to flooding (as indicated by the low abundance of *Aulacoseira granulata*) due to land-use management such as river bank stabilization, preventing overflow or this could reflect the presence of riparian vegetation which stabilised the river banks.

The most prominent change in the indicator diatoms at Rhenostervlei Farm occurs at the onset of the 20th century CE, characterised by a shift from a saline, dry and nutrient-poor environment (ca. 1790-1890 CE) to a more turbid, nutrient-rich environment with increased freshwater influence that was prone to periodic flooding (ca. 1890-2011 CE). As climate has been shown to have had little influence in this system, a likely cause of this shift may be land-use change on Rhenoterstvlei Farm and potentially the Berg River catchment as a whole.

On a regional scale, agricultural land-use activities intensified from the 20th century onwards in the western Cape (Hoffman, 1997) and on a local scale, the transition from Zone 1 to Zone 2 (ca. 1890 CE) roughly coincides with the farm being obtained by a new owner (1906 CE) - thus the shift observed in diatom assemblage may reflect the agricultural land-use changes that ensued and possibly indicates that agricultural activities intensified. Land-use related factors that may have been involved in driving water quality and availability could include burning, clearance of natural vegetation, soil disturbance, fertilizer use and water extraction and diversion.

Evidence which supports land-use driven change during the time period of Zone 2 (ca. 1890-2011 CE) at Rhenostervlei Farm is indicated by macro-charcoal - a proxy for local fires - and the sediment accumulation rate. Charcoal particles have been widely used to infer past burning regimes in palaeoecological records (e.g. Gell et al., 1993; Millspaugh and Whitlock, 1995; Forbes et al., 2018). Preliminary charcoal results from a study that is currently underway using the same sediment core from Rhenostervlei Farm, revealed a drastic peak in macro-charcoal in the early ca. 1900s (Forbes pers. comm. 2019). This could indicate the burning and clearing of natural vegetation or old crops by the new farm owner in order to prime the land for crops/grazing. Nutrients, such as phosphorus and nitrogen, are often mobilized by fire, which can lead to increased loading into river ecosystems (Sherson et al., 2015). These fire-driven nutrient pulses can increase nutrient concentrations many fold and are usually associated with floods (Bixby et al., 2015). The increased nutrient inputs into the river system at Rhenostervlei Farm may explain the increase in eutrophic indicators species since the 1900s CE.

Slightly after the peak in macro-charcoal, sediment accumulation, as inferred from the chronology from radiocarbon and ^{210}Pb dates, increased, starting ca. 1920 CE and continued to be consistently high through-out the time period of Zone 2 (see Figure 3b). Sedimentation, which is the process of soil being eroded, transported and deposited as layers of solid particles in water bodies, can be driven by climate change (increased rainfall/flooding) but can also be linked to anthropogenic disturbance and landscape modification (e.g. Xu, 2003; Stinchcomb et al., 2011). Since the instrumental record didn’t indicate increasing rainfall, it was assumed that land-use, particularity agricultural activities, was driving this increase in sedimentation. Agricultural activities such as the clearance of vegetative cover and increased soil disturbance, may drive sedimentation through increased surface runoff and soil erosion (Návar and Synnott, 2000; Stinchcomb et al., 2011). This has many implications for water quality and could explain the increased freshwater influence, turbidity and nutrient state starting at the beginning of the ca. 1900s CE.

The increased freshwater influence and flooding signal from ca. 1890-2011 CE may be a result of natural vegetation being cleared from Rhenostervlei Farm in the early 1900s (potentially from fire as indicated by the increase macro-charcoal abundance) to be replaced by crops and grazing land. By removing vegetation cover, evapotranspiration and infiltrability of the soil would decrease, causing more water to enter the river ecosystem as runoff. This effect of vegetation clearance in the Berg River catchment is supported by a study that compared the runoff and evaporation difference between indigenous renosterveld vegetation at Voëlvlei Nature Reserve and an adjacent wheat field in 2007 (De Clercq et al., 2010). Compared to Renosterveld, the wheat lands had lower evaporation, limited infiltration and greater surface runoff that was faster during large rain events (De Clercq et al., 2010). The site on which Rhenostervlei Farm is situated was once dominated by Renosterveld and Fynbos vegetation, and we would therefore expect that the transformation to crops which include wheat lands and grazing lands would cause the above-mentioned impacts.

The impact of land-use causing increased freshwater influence and higher water levels was shown in a diatom-based Holocene record from a coastal environment of the Tuckean Swamp in eastern Australia (Taffs et al., 2008). In this study, the increasing dominance of *Cyclotella meneghiniana* correlated well with the introduction of stock grazing and cropping which involved native vegetation clearance in the catchment area. This was interpreted as an increase in freshwater influence in the swamp's environment due to the associated increased runoff from these activities (Taffs et al., 2008). Similarly, the increased runoff at Rhenostervlei Farm after ca. 1900 CE could have resulted in more standing waters at the site; and/or more water entering the river causing high water levels and more river overflow into the site – possibly causing the increased appearance of freshwater or planktonic diatoms (*Gomphonema parvulum*, *Discostella pseudostelligera* and *Cyclotella meneghiniana*) and decrease of dry tolerant aerophilic species (*Pinnularia borealis*).

As well as seeing a shift in freshwater influence, there also appeared to be an increased signal for high nutrient, turbid and turbulent conditions in ca. 1890-2011 CE which may have been driven by agricultural activities. Through vegetative cover removal and increased soil disturbance, surface runoff and soil erosion could increase leading to higher nutrient loads entering the system. This could have been further emphasised by fertilizers-use on Rhenostervlei Farm - a practice that is particularly common on rain fed farms in this area (Gildenhuys pers. comm. 2019) - as nutrients (phosphorus and nitrogen) from fertilizers can move downhill in runoff, often leading to freshwater and estuarine eutrophication (Carpenter and Kinne, 2003). In addition to the Rhenostervlei Farm site becoming more eutrophic as indicated by locally derived benthic species (such as *Navicula rostellata, Nitzschia sigma, Diadesmis confervacea* and *Hippodonta capitata*), there also could be a signal for the nutrient-enrichment of Berg River on a more regional scale as indicated by the planktonic species (*Aulacoseira granulata*, *Cyclotella meneghiniana* and *Discostella pseudostelligera*) which could have been deposited at the site from areas up river.

Over the last century, the Berg River has experienced major changes nutrient status (De Villiers, 2007). Likely anthropogenic nutrient inputs included agricultural runoff and increased wastewater discharge and runoff from informal settlements along the Berg River (de Villiers, 2007). In a study of the Berg River from 1985-2005, nutrient monitoring stations along the river have indicated a clear increase of close to 10-fold in downstream inorganic nitrogen ($NO3^{-} + NO2^{-}$) and phosphorus (dissolved total phosphorous and PO_4^{3-}) concentrations, with these parameters peaking along the middle section of the river around Hermon (de Villiers, 2007). Even though the sources of nutrient input are concentrated in the middle section of the river (between Paarl and Hermon) - the most heavily cultivated and most populated area along the river - there was still a temporal trend of increasing nutrient concentrations in the lower-middle reaches of the river which includes the Rhenostervlei study site. Therefore, the higher dominance of eutrophic planktonic species *Aulacoseira granulata*, *Cyclotella meneghiniana* and *Discostella pseudostelligera* in the latter section of Zone 2 (Zone 2b ca. 1945-2011 CE) may reflect the increase in the nutrient status from up river as well, possibly due to agricultural intensification as well as increased wastewater discharge and runoff from informal settlements along the Berg River.

The presence of *Aulacoseira granulata* and *Discostella pseudostelligera* within the time period ca. 1890-2011 CE, were also thought to indicate more turbid conditions and may be indicative of further agricultural intensification. This signal appeared to be especially high between ca. 1945-2011 CE with particularly high peaks seen in *Aulacoseira granulata* between ca. 1945-1950 CE (see Figure 5, Zone 2b). This increase in turbidity towards the mid-1900s CE may be due to increased agricultural activities which cause surface runoff in partnership with soil erosion, enhancing the sediment loads entering the river causing more turbulent flows and excess levels of turbidity. Cultivated slopes are known to be the major source of sediment when compared to vegetated areas (Xu, 2003). The lower to middle section of the Berg River (Gouda, Malmesbury and Piketberg) is prone to turbid condition due to the abundance of wheat lands within this area which carry far more sediment in runoff compared to other agricultural lands such as fruit farms and vineyards (Harrison and Elsworth, 1958).

After the high peak of *Aulacoseira granulata* between ca. 1945-1950 CE, its abundance appears to progressively decrease to the ca. 2005 CE, suggesting lower water levels and decreased flooding and river over flows (Figure 4, Zone 2b). This could possibly be a result of increased water extraction in the area or specifically from the Klein Berg tributary which joins to the Berg River in close proximity to the

Rhenostervlei Farm site. Voëlvlei Dam was commissioned in 1952 and was constructed by enlarging the capacity of a small natural lake and connecting it via a canal to the Klein Berg River near Gouda (Harrison and Elsworth, 1958). The dam provided water to towns in the surrounding areas (e.g. Riebeek-Wes, Malmesbury and Moorreesburg) and farms along the supply route (River Health Program, 2004). The Klein Berg River used to supply a maximum of 1.3 million m^3 per day to Voëlvlei Dam however, in 1969 the dam wall was raised and more water was abstracted (maximum of 1.7 million m^3 per day) due to Cape Town's increasing water demand (River Health Program, 2004). Since the Klein Berg River feeds directly into the area of where the site on Rhenostervlei Farm is situated, the increased extraction from this tributary may have decreased the flow feeding in to the Berg River, causing water levels at the site to decrease, causing the occurrence and intensity of flooding and river overflows to decrease with a resultant decrease in *Aulacoseira granulata* abundance.

Despite the importance of anthropogenic drivers in this system, significant feedbacks and interactions may still exist between climate and land-use. A possible interaction between climate and land-use may have been at play with regard to the abundance of *Aulacoseira granulata* in the sediment record. This species possibly increased in the system due to land-use such as reduced natural vegetation cover, more surface runoff which caused higher turbidity and water levels and flooding which deposited the diatom at the site. In a sense, land-use made the system more prone to flooding (due to vegetative cover removal which caused increased surface runoff into the river) but ultimately, the amount of flooding will depend on the amount of precipitation.

4.4 Management implications

The Berg River is a pivotal source of fresh water for domestic, industrial and agricultural use as well as for in-stream ecology and also provides goods and services which contribute to human welfare and economic growth (River Health Programme, 2004). Therefore, knowledge of the impacts on river water quality and assessing whether ecological resilience has been surpassed, are of the utmost importance. In order to assess the ecological functioning of a river system and determine whether intervention is needed, palaeoecology can be used to detect trends and establish the historical range of variability (Reid et al., 2006). The palaeoecological approach can also lead to a better understanding of the processes driving change and therefore can be used to define appropriate management actions (Reid et al., 2006).

In this study, a high resolution palaeoecological approach was successfully used to establish trends and variability in water quality and availability at an agricultural site on the Berg River over a ca. 220 year period using fossil diatoms. The site adjacent to the Berg River on Rhenostervlei Farm was found to have experienced increased freshwater inputs from river overflows and also increased nutrient and sediment loads moving into the ca. 1900s CE.

As well as establishing changes, this project was also able to suggest possible drivers of change by comparing diatom abundances to historical land-use and climatic records and other proxies such as macro-charcoal and the sediment accumulation rate. The findings suggest that the changes in diatom assemblage were likely due to land-use change in the form of agricultural intensification which involved the removal of natural vegetation, soil disturbance, fertilizer use – all of which contributed to increased surface runoff, erosion and nutrient and sediment loading into the site. Although changes in water quality were observed, there was no clear evidence of a drastic environmental shift in which certain diatom indicators completely dominated while others were lost i.e. the complete loss of oligotrophic species to be replaced by high-nutrient species. However, under a scenario of increasing agriculture intensification, regime shifts may occur. A rapidly growing body of evidence suggests that agricultural modification of the quantity and quality of hydrological flows can increase the risk of catastrophic ecological regime shifts (Scheffer et al., 2001; Carpenter and Kinne, 2003. Rietkerk et al., 2004; Gordon et al., 2008). One such regime shift that has been observed in aquatic ecosystems and also has been studied in diatom-based environmental reconstructions, is freshwater eutrophication (Carpenter, 2005; Gordon et al., 2008; Hall and Smol, 2010).

Despite the area under maize, wheat and dairy farming in South Africa having decreased significantly over the last 20 years (Agricultural Statistics, 2008), production appears to be constant revealing a trend of agricultural intensification (WWF, 2010). One characteristic of this agricultural intensification may involve the increased use of fertilizer (WWF, 2010). Therefore, if agriculture continues to intensify in the Berg River catchment, we may see a catastrophic regime shift to eutrophic conditions.

Land-use appeared to be a prominent driver of change as the variation in the instrumental rainfall record failed to explain high amount of variation seen in water quality. Furthermore, the predicted impacts of 20th century warming (decreased P-E with resultant increased salinity) showed no clear correspondence with the change in water quality. This knowledge may be empowering for land managers as implementing solutions to minimise the negative impacts of land-use on water quality are within their capacity - whereas the impacts of global climate change are less so. However, this capacity to manage for land-use impacts on water quality may be limited under future climate change. Climate predictions for the CFR generally suggest increasing temperatures and aridity as well as changes in rainfall patterns such as decreased winter rainfall and increased drought periods (Altwegg et al., 2014). Furthermore, the hydrological consequences of climate change may interact with land-use, moving ecosystems either closer to or further away from some critical thresholds; thereby influencing the systems vulnerability to other agriculturally induced changes in hydrology (Gordon et al., 2008). Alterations in water quality and availability through agricultural intensification are likely to increase the risk of unexpected regime shifts unless management practices are changed (Gordon et al., 2008).

4.5 Limitations and further investigation needed

Although this study revealed a possible change in water quality and availability (increased freshwater influence due to river over flows at the site, nutrient enrichment and turbid conditions) in the 20th century there was still much variability in the diatom abundances which this study could not explain. This may be due to the complex nature of riverine ecosystems which often can be subject to high inter and intra-annual variability (Reid et al., 2006). Furthermore, diatoms assemblages have a diverse array of factors controlling species abundance and composition, including disturbances (short-term), stressors (long-term), resources (e.g. light, nitrogen, phosphorous), hydraulic conditions and biotic interactions (Stevenson, 1997; Larned, 2010).

Another limitation is that the diatom record doesn’t reflect seasonality as each level is an amalgamation of environmental signals summed over both seasons within a year or over multiple season cycles (Kirsten and Meadows, 2016). Therefore, the different signals that could indicate a wet winter with high flow and flooding or a particularly dry summer may be overlain causing some uncertainty in interpretations.

To improve the interpretation of diatom fossil assemblages in order to infer environmental change on Rhenostervlei Farm and the Berg River catchment, a study with multiple proxies should be undertaken. The inclusion of multiple proxies, such as fossil pollen (vegetation), coprophilous fungal spores (herbivory) and Loss of Ignition (LOI) could improve the ability to identify possible drivers causing variation in environmental conditions and could also aid in piecing together the complex interactions these drivers may undertake in the system. Valuable information may also be gained from extending the study to more sites at different parts of the catchment on different types of land-use.

4.6 Conclusions

The fossil diatom record from a floodplain adjacent the Berg River on Rhenostervlei Farm suggests various changes in water quality and availability over the ca. 220 year period captured in this study. Most notably, there was a shift from a saline, dry and nutrient-poor environment (ca. 1790-1890 CE) to a more turbid, nutrient-rich environment with increased freshwater influence and periodic flooding (ca. 1890-2011 CE). Through the chronological analysis and the interpretation of the historical record as well as other environmental proxies (sediment accumulation rate and macro-charcoal), the causes of the detected change in water quality and availability at the floodplain site are likely related to land-use change in the form of agricultural intensification at Rhenostervlei Farm and potentially in the catchment as a whole. The higher sedimentation rate and macro-charcoal peak during ca. 1890-2011 CE may be linked to increasing agricultural activities on Rhenostervlei Farm such as burning, clearance of natural vegetation, soil disturbance, fertilizer use and causing increased runoff and nutrient and sediment loading which would provide an explanation for the increased presence of freshwater, planktonic and eutrophic diatom species. Within this time period, diatom inferred eutrophic and turbid conditions seem to be particularly high between CE ca. 1945-1950 CE (see Figure 5, Zone 2b) and may be indicative of further agricultural intensification. The decreased appearance of *Aulacoseira granulata*, as a high water turbulence and flooding indicator, after this peak may be due to decreased water levels due to water extraction and diversion from the sites main tributary Klein Berg. Although no evidence of a catastrophic regime shift was identified, if land-use practices continue to intensify (e.g. increased fertilizer use) and future climate change interacts and influences the agricultural alterations to hydrological systems, we may expect increased vulnerability to global change and unexpected ecological outcomes such as regime shifts. In order to improve the interpretation of fossil diatom records in terms of changing water quality and availability, a study with multiple proxies should be undertaken to help infer environmental conditions in a complex environment that has many potential drivers, such as the Berg River.

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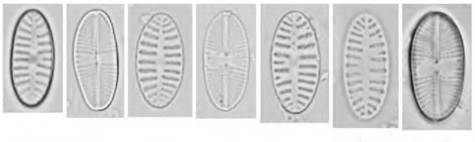
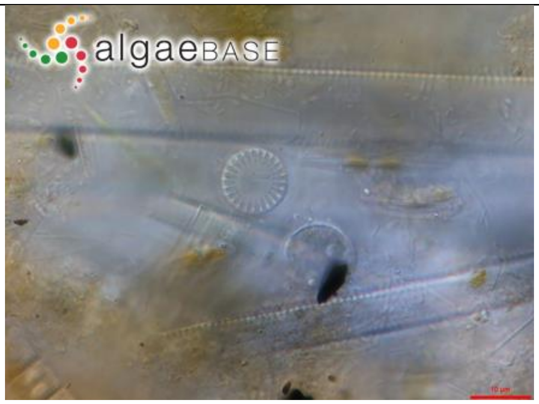
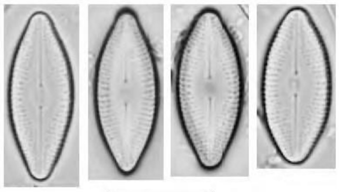
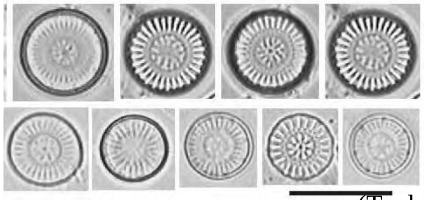
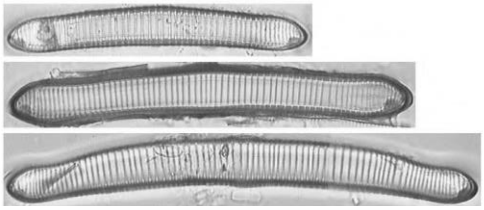
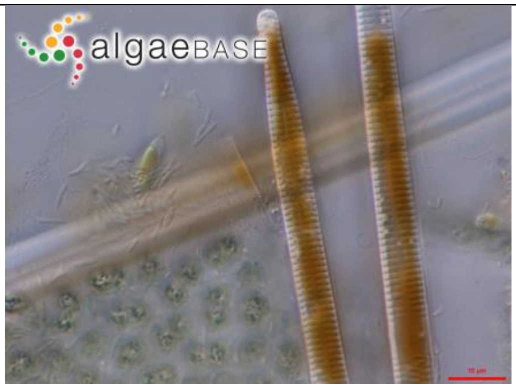
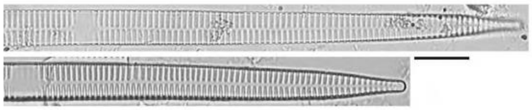
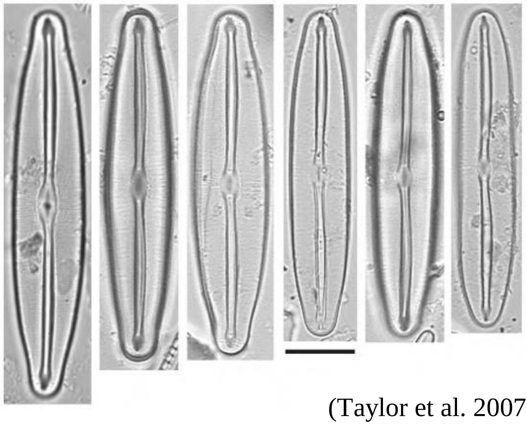
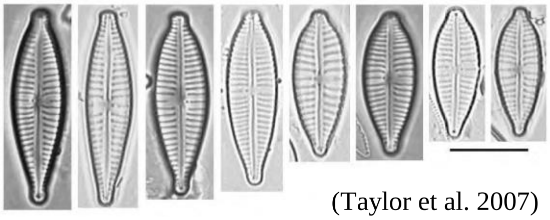
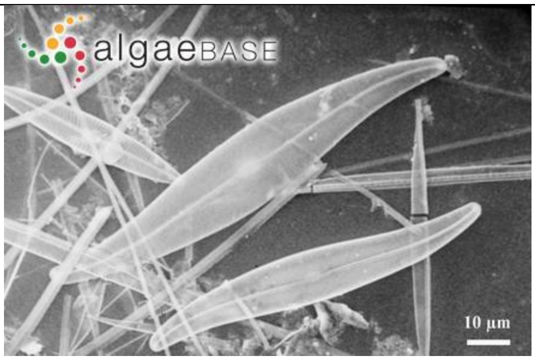
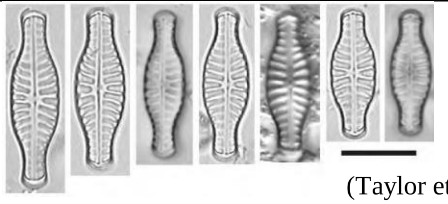
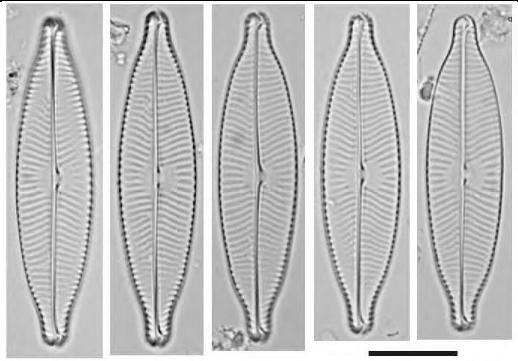
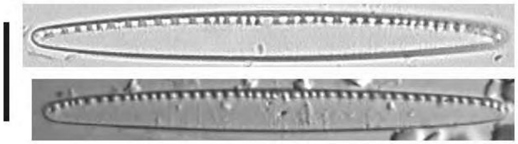
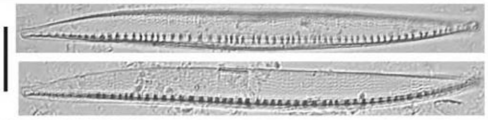
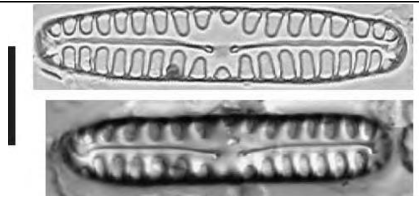
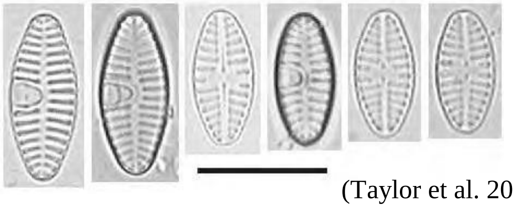
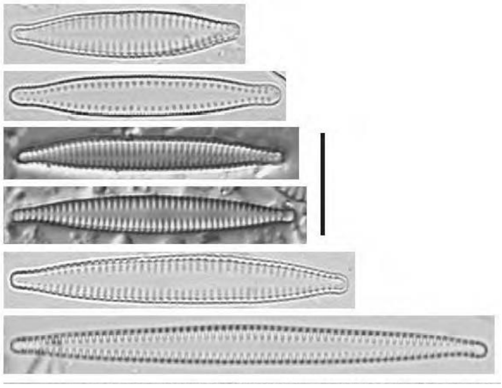
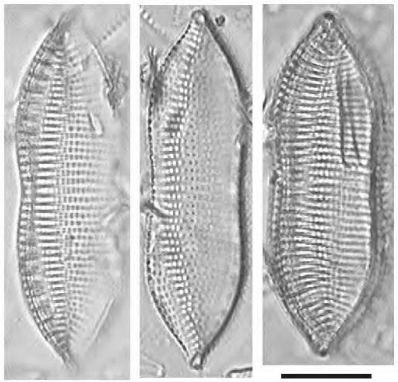
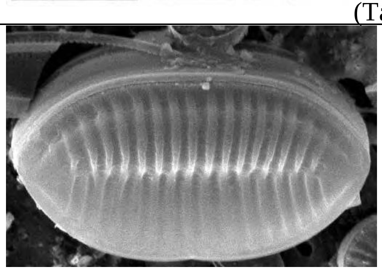
APPENDIX

***Table 1:** The full scientific names (including species authority) of the top 20 diatom species with example images obtained from http://www.algaebase.org/ and Taylor et al. (2007).*

Full scientific name of diatom species	Example images of diatom species
Achnanthes oblongella Østrup	 (Taylor et al. 2007)
Aulacoseira granulata (Ehrenberg) Simonsen	
Cyclotella meneghiniana Kützing	
Diadesmis confervacea Kützing	 (Taylor et al. 2007)
Discostella pseudostelligera (Hustedt) Houk & Klee	 (Taylor et al. 2007)

Eunotia formica Ehrenberg	 (Taylor et al. 2007)
Fragilaria ulna (Nitzsch) Lange-Bertalot	
Fragilaria ulna var. acus (Kützing) Lange-Bertalot	 (Taylor et al. 2007)
Frustulia vulgaris (Thwaites) De Toni	 (Taylor et al. 2007)
Gomphonema parvulum (Kützing) Kützing	 (Taylor et al. 2007)

Gyrosigma acuminatum (Kützing) Rabenhorst	
Hantzschia amphioxys (Ehrenberg) Grunow	
Hippodonta capitata (Ehrenberg) Lange-Bertalot, Metzeltin & Witkowski	 (Taylor et al. 2007)
Navicula rostellata Kützing	 (Taylor et al. 2007)
Nitzschia filiformis (W. Smith) Van Heurck	 (Taylor et al. 2007)
Nitzschia sigma (Kützing) W. Smith	 (Taylor et al. 2007)

Pinnularia borealis Ehrenberg sensu lato	 (Taylor et al. 2007)
Planothidium rostratum (Østrup) Lange-Bertalot	 (Taylor et al. 2007)
Tabularia fasciculata (C. Agardh) D.M. Williams & Round	 (Taylor et al. 2007)
Tryblionella coarctata (Grunow) D.G. Mann	 (Taylor et al. 2007)
Tryblionella levidensis W. Smith	 (Taylor et al. 2007)