Neo- and palaeolimnological investigations in a humic and a clear water lake in the west of Ireland

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Apriti sesamo

Alles hat ein Ende

nur die Wurst hat zwei.

Abstract

Surface waters draining peat catchments often have a characteristic brown colour due to the presence of dissolved organic carbon (DOC) compounds. A rise in DOC concentrations has been documented in rivers and lakes in various parts of Europe and North America over the last few decades. The processes responsible for the increased DOC load are complex and not entirely understood, but it is obvious that this change could be indicative of decreased terrestrial storage of carbon, which has important consequences for aquatic ecology and drinking water quality.

This thesis applies contemporary or neo- and palaeolimnological approaches at different temporal and spatial scales in a humic and clearwater lake in the west of Ireland (Lough Feeagh, Co. Mayo and Lough Guitane, Co. Kerry). An investigation of contemporary auto- (pico- and phytoplankton), mixo- (phytoflagellates) and heterotrophic (bacteria and ciliates) communities was fundamental to this research. The results confirmed that higher loads in suspended solids, and thus a darker water colour, which had a direct effect on light attenuation, depressed autotrophic biomass and simultaneously stimulated heterotrophic bacteria and potentially mixotrophic phytoflagellates. A heterotrophic base for total organic production served as an energy and carbon source. A flash-flood in July 2009 caused an increase in Cryptophyta and bacteria. In contrast, the clear water lake was characterized by lower DOC levels and deeper Secchi depths and thus, more light availability, favouring the autotrophic community and extending the growing season.

Sediment traps installed in three locations within each lake showed contrasting seasonal and inter-annual dynamics of lithological, geochemical and biological variables. C/N ratios reflected a mixture of algal and land-derived organic matter with a major peaty influence in the humic lake. The comparison of the open water phytoplankton community and diatom assemblages with sediment trap fossil pigment and diatom assemblages showed a close agreement and reflected a seasonal pattern. In contrast, the

comparison between sediment trap and surface sediment assemblages revealed different patterns. Pigment and diatom assemblages were influenced by water depth, while inter-annual variability and/or dilution and mixing through bioturbation influenced the surface sediment diatoms.

Lastly, sediment core lithological, geochemical and biological proxies enabled reconstruction of the past environment of the lakes and their surrounding catchments. Both lakes were characterized by contrasting water column and sediment trap responses and consequently their sediment core responses were different. Divergent levels of DOC in the two lakes contribute to different algal community structures and thus fossil assemblages. One of the most striking outputs was shown by an index of ultraviolet radiation penetration that gave an indirect indication of dissolved organic matter (DOM) present in the water column. A decreasing trend in the humic lake indicated an increase in DOM in the water column over the last ca. 70 years. This was paralleled by an increase in Cryptophyta known to tolerate lower light conditions and a shift in diatom assemblages. The trend was concurrent with extensive commercial afforestation and an exponential increase in sheep grazing, however climate change could also have contributed to the transport of suspended sediment into the lake.

Declaration

I hereby declare that this thesis has not been submitted as an exercise for a degree at this or any other university and that it is entirely my own work. I agree that the library may lend or copy this thesis upon request.

Signed,
Karin Sparber

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Chapter 1 - Introduction

1.1 General introduction and research objectives

Dissolved organic carbon (DOC) in lacustrine environments can be derived from both terrestrial (allochthonous) sources or from sources produced within the aquatic ecosystem (autochthonous). The larger fraction of the total DOC in lakes is from decomposed organic matter derived from long-term terrestrial stores or peatlands. Natural brown coloured lakes, known also as humic, dystrophic or bog lakes (Thomas *et al.*, 1996), are very common in the northern temperate zone, where extensive peat soils are characteristic (Dillon & Molot, 1997a; Kortelainen, 1999; Ojala *et al.*, 2011). Long-term observations over the last few decades show a steady increase in DOC in freshwaters across Europe and North America (Freeman *et al.*, 2001a; Evans *et al.*, 2005; Sucker & Krause, 2010). The significance of these upward trends in DOC concentrations and its dynamics and influence of upon aquatic ecosystems is not entirely understood (Tranvik & Jansson, 2002; Roulet & Moore, 2006), but it is certain that they may have wide-ranging impacts on the functioning of aquatic ecosystems (Jones, 1992; Evans *et al.*, 2005; Jansson *et al.*, 2007). Organic matter also affects water treatment processes (e.g. trihalomethane (THM) formation) and therefore, the drinking water quality (Alarconherrera *et al.*, 1994), which can have negative effects on human health (Janus, 2010).

The transfer of carbon from terrestrial to aquatic, and finally marine ecosystems, forms a significant component of the global carbon cycle (Hope *et al.*, 1994). Even small changes in DOC quality and quantity can have considerable significance for carbon cycling and have substantial ecological consequences (Cole *et al.*, 2000; Porcal *et al.*, 2009), including shifts in the structure and function of food webs, especially for the microbial component (Jones, 1992; Sucker & Krause, 2010; Kostrzewska-Szlakowksa & Jasser, 2011). A number of features are shared by humic lakes: brown water colour, low light penetration and consequent low available light energy for primary producers, predominance of the red part of the light spectrum, low pH, low alkalinity, low conductivity together with low concentrations and reduced bio-availability of dissolved

inorganic nutrients (Arvola, 1984; Jones, 1998; Löfgren *et al.*, 2003). The magnitude and proportion of carbon derived from allochthonous and autochthonous sources varies widely among different aquatic ecosystems. In highly productive eutrophic lakes, autochthonous production plays a fundamental role, while in more oligotrophic lakes allochthonous organic matter can affect the entire lake metabolism (del Giorgio *et al.* 1999; Wetzel, 2001). In recent years the traditional concept of lake food webs has been challenged by the evidence, that in spite of their position in the landscape, many water bodies maybe net heterotrophic aquatic systems (Cole *et al.*, 1994; del Giorgio *et al.*, 1997; Ojala *et al.*, 2011). These systems are sources of CO_2, due to the importation and mineralization of allochthonous organic carbon and the resultant degassing of inorganic carbon (Cole *et al.*, 1994; Algesten *et al.*, 2003; Sobek *et al.*, 2003). A relationship between lake trophy and net metabolic balance has been observed, suggesting that the latter is more frequent in oligotrophic than in eutrophic lakes (del Giorgio & Peters, 1994).

1.1.1 European Union Directives

The European Union Water Framework Directive (WFD) (2000/60/EC) (European Union, 2000a) and the Habitats Directive (92/43/ECC) (European Union, 2003b) formulated a legislative framework to promote: sustainable management of fresh- and saline waters, protect and enhance all aquatic environments, prevent future deterioration; achieve "good ecological status" and ensure sustainable functioning by 2015 (European Union, 2000a).

The World Health Organization (WHO) together with the European Parliament Environment Committee set standards for drinking water quality at the tap including the general obligation that drinking water must be wholesome and clean. Many lakes are drinking water sources, which need to be purified (removal of undesirable chemical and biological contaminants from raw water) for human consumption (potable water) and also for other purposes such as medical, pharmacological, chemical and industrial applications. In general, the methods used include a variety of physical (filtration, sedimentation) and chemical (flocculation, chlorination) processes and the use of electromagnetic radiation (UV-light). The processes of water treatment reduce the concentration of particulate matter including suspended particles, parasites, algae, fungi,

bacteria and viruses. The European Drinking Water Directive (1998/83/EC) has sharpened the enforcement of water quality norms and put particular emphasis on the organic matter content by restricting the maximum acceptable concentration of THMs. Where humic waters are used as a potable water source efforts are made to remove organic matter during water treatment for aesthetic reasons and because organic matter reacts with the oxidants (chlorine, ozone, hydrogen peroxide) during disinfection and produces a series of disinfection by-products (DBPs) (Rook, 1974; Reckhow & Singer, 1990; WHO, 2005). DBPs have been associated with adverse health impacts, including congenital abnormalities and an increased risk of cancer (Källén & Robert, 2000; Nikolaou & Lekkas, 2001; WHO, 2005). The organic matter character, organic precursor levels and DBPs formation, nature and reactivity can be characterized by seasonal changes that can cause variations of the water quality over time (Uyak *et al.*, 2008).

1.1.2 Neo- and palaeolimnology

Contemporary aquatic ecology and palaeolimnology (lake sediment reconstructions) are complementary disciplines that contribute to knowledge and understanding of long-term lake responses (Battarbee *et al.*, 2005a; Batterbee *et al.*, 2005b). Generally, contemporary studies (physico-chemical monitoring and ecological data sets) seldom extend beyond 10 years and thus, cannot show how lake ecosystems change over the longer (decadal-centennial) timescales. Longer term datasets are essential in assessing lake history, providing baseline reference information and defining the timing and rate of ecological change (including e.g. lake development, catchment processes) (Likens, 1979; Lotter & Bigler, 2000; Batterbee *et al.*, 2005b; Batterbee *et al.*, 2011). The combination of neo- and palaeolimnological research has provided valuable data (Bennion & Batterbee, 2007; Dalton *et al.*, 2009) and can be helpful in deriving targets for lake restoration and conservation measures to ensure future environmental protection of aquatic systems (Moss *et al.*, 1996; Köster *et al.*, 2005). Additionally, palaeolimnological and standard limnological approaches can be integrated through the application of sediment traps. Sediment traps are a device, which permit quantitative collection of particles falling through the water column and enable high resolution sampling at seasonal, inter-annual and/or decadal time scale, allowing an integration of past and present lake responses (Ryves *et al.*, 2003; Battarbee *et al.*, 2005a; Batterbee *et*

al., 2005b). Sediment traps enable comparisons between sedimenting matter and contemporary water column measurements and with basin sediment records, which offer a continuous long-term archive of lake history (Cameron, 1995; Bennion *et al.*, 2011). The combination of limnological and palaeolimnological approaches are highly complementary and can provide essential information in assessing lake ecosystem response to changes in nutrient loading and the role of several drivers and stressors, such as for example land-use and climate change.

1.1.3 Research objectives

The overall aim of this research is to examine the nature and fate of organic matter and the influence on pelagic organisms through the application of neo- and palaeolimnological approaches in a clear water and a humic lake in the west of Ireland. This aim is achieved through three main objectives:

The purpose of the first part of this research was to establish the present ecological status of pelagic auto- (phyto- and picoplankton), potentially mixo- (phytoflagellates) and heterotrophic communities (bacteria and ciliates). The objective was to track variations in biomass production in relation to abiotic variables, to determine if variations in water colour and DOC input alters the relationships between the pelagic communities over an annual cycle.

Secondly, the relationship between living lake communities in the water column and the records of these communities in suspended sediment traps and basin surface sediment are explored. Spatial and temporal variations in organic matter, total organic carbon and total nitrogen load, pigment concentrations and diatom assemblages are quantified in sediment trap samples collected over circa two years. The factors regulating spatial and temporal variations and their external influences are explored.

Lastly, an examination of lake sediment cores permitted an extension of the timescale examined. By looking at sediment core responses an evaluation of change in system state, including past changes in primary production, algal communities and organic matter was achieved using palaeolimnological techniques. Parameters including fossil pigments, diatom assemblages, total organic carbon and total nitrogen were analysed.

Comparisons between lithological, geochemical and biological proxies in sediment cores and historical catchment and climate changes enable evaluation of potential drivers and pressures. This longer-term context can help determine the onset and magnitude of change and inform predictions of future state.

1.2 Thesis Structure

The body of this thesis is divided into eight chapters. A literature review of key classic and contemporary literature on the sources and role of DOC in aquatic environments, the importance of drinking water quality, recent rises of DOC and its potential drivers, along with an introduction to palaeoecological studies is presented in Chapter 2. This is followed by a description of the study sites in Chapter 3. Materials and methods used in the project are outlined in Chapter 4. The first of three results chapters is presented in Chapter 5 and examines the dynamics of phytoplankton, picoplankton and heterotrophic bacteria along with physico-chemical parameters at the two study sites. Sediment trap seasonal fluxes and a comparison with open water and surface sediment samples are detailed in Chapter 6. Sediment core reconstructions of lithological, geochemical and biological proxies for both lakes are contained in Chapter 7. Each result chapter contains a detailed discussion. Chapter 8 highlights the implications of the research and its the contribution to knowledge.

Chapter 2 - Literature review

2.1 Introduction

This chapter reviews the role and fate of DOC in freshwater ecosystems and its influence on the classification of lakes. The consequences of variation in the quantity and quality of organic matter for the quality of drinking water are explored. The recent changes in DOC concentrations in aquatic systems are outlined and potential drivers of these changes are explored. Finally, palaoelimnological applications are reviewed.

2.2 Sources and sinks of dissolved organic carbon in aquatic environments

Carbon is crucial to life on Earth and is the most actively cycled element in the biosphere (Sulzman, 2000). Biological processes convert organic and inorganic carbon into one another. For example, photosynthesizing organisms convert atmospheric inorganic carbon (CO_2) into organic carbon and respiration converts organic carbon into inorganic carbon and releases it back to the atmosphere. Other inorganic carbon sources, such as bicarbonate (HCO_3^-) and carbonate (CO_3^{2-}), enter aquatic systems through ground- and surface water. Aquatic organic carbon is a component of living and non-living biomass and is used as an energy source for secondary aquatic production (Tranvik, 1988; Karlsson *et al.*, 2003). Organic matter can be divided into dissolved molecules, colloidal suspensions and particulate matter (Kronberg, 1999). DOC is the largest pool of organic carbon in lake water (typically > 90% of the total organic C) (Thurman, 1985; Wetzel, 2001). DOC has been reported to constitute 97% of the TOC in water of boreal lakes (Kortelainen *et al.*, 2006). The natural range of DOC, from < 0.5 mg L^{-1} to > 50 mg L^{-1}, is enough to span the range from crystal clear to dark brown waters (Thurman, 1985; Kortelainen, 1999; Mulholland, 2003). In aquatic systems DOC originates mainly from in-lake (internal or autochthonous DOC) and the surrounding terrestrial catchment (external or allochthonous DOC). A minor contribution is also sourced from exchanges between air and water (Wetzel & Likens, 2000; Reche & Pace, 2002; Bertilsson & Jones, 2003; Kortelainen *et al.*, 2006). The fraction of DOC that is created from in-lake processes is mainly derived from

autochthonous primary production (e.g. macrophytes, phytoplankton, picoplankton). This photosynthetically produced carbon pool, classified as labile dissolved organic carbon (LDOC), constitutes circa 15% of TOC in aquatic environments (Figure 2.1) (Søndergaard & Borch, 1992). This pool of carbon is readily available for bacterial growth, and as the supply of humic substances in lakes is high, they become quantitatively important bacterial substrates (Tranvik, 1989; Moran & Hodson, 1990). A small part of the LDOC fraction (0.2%) is produced photosynthetically and excreted as dissolved extracellular organic carbon (EOC), mainly by phytoplankton in pelagic areas and by macrophytes and epiphytic algae in the littoral zone (Münster & Chróst, 1990). In contrast, allochthonous DOC is derived from the surrounding terrestrial plant matter and/or humic substances (e.g. cellulose, lignin, tannins). The latter is significantly processed as it passes through soil before entering aquatic systems via microbial and abiotic processes (e.g. decomposition and photo-degradation of terrestrial plant biomass) (Wetzel, 2001; Reche & Pace, 2002). In addition, anthropogenic activities can be a source of DOC, which enters the aquatic environment through direct discharge from industrial, agricultural and domestic activity (Apsite & Klavins, 1998), indirect leaching of soil organic matter and aerial dispersal (Hinton *et al.*, 1997; Hudson *et al.*, 2007).

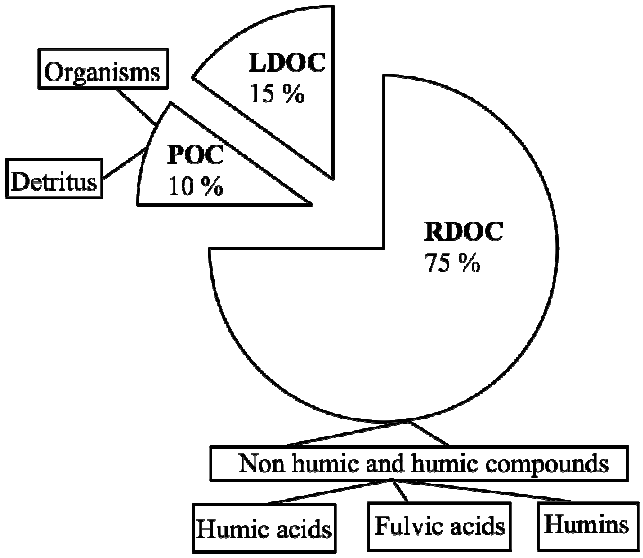

Figure 2.1 – Approximation of the different fractions of the total organic carbon pool in aquatic systems composed of particulate/colloidal organic carbon (POC), labile dissolved organic carbon (LDOC) and refractory dissolved organic carbon (RDOC) (modified from Tulonen, 2004; page 7).

The term organic carbon covers a range of substrates. The dissolved organic fraction is defined as the carbon concentration of water passing through a 0.2 – 0.7 µm filter (Thurman, 1985; Wetzel & Likens, 2000). The carbon retained on the filter constitutes the fraction of colloidal and particulate organic carbon (POC) fraction (McKnight *et al.*, 1997). POC includes both living organisms (bacteria, phytoplankton, protozoa and metazoa) and particulate detritus (dead organic material). The dissolved organic carbon fraction includes soluble organic substances variously named as coloured or refractory dissolved organic carbon (RDOC) or chromophoric dissolved organic matter (CDOM) (Münster & Chróst, 1990). In the older literature it is termed as gilvin and gelbstoff or yellow substance. RDOM is considered to be a mixture of compounds chemically characterized as humic and fulvic acids and/or humus (Kirk, 1994a, 1994b; McKnight *et al.*, 1997; Williamson *et al.*, 1999). This heterogeneous mix of yellow, brown and even black organic compounds present in all natural waters impact the chemistry and biology of water (WHO, 2005; Hudson *et al.*, 2007). RDOM accounts for 75% of the DOC in aquatic environments, which is not easily broken down by bacteria and is composed of non-humic and humic components. Non-humic components are insoluble in aqueous systems and are formed by aromatic and aliphatic hydrocarbons, esters, acids, and even relatively polar structures of microbial origin, such as polysaccharides and glycoproteins (Hayes, 2001). Humic compounds are mostly generated by the partial decomposition of, or exudation from, living plants and animals and soil microorganisms. The organic matter formed by these processes may be stored in the soil for varying lengths of time (e.g. as peat) before decomposition processes render a part of this material soluble. Humic compounds can be sub-divided into three categories, humic and fulvic acids and humins, chemically defined by solubility at different pHs.

In aquatic systems the main sinks of organic matter are *in situ* microbial activity and mineralization, flocculation, coagulation and sedimentation. The microbial utilization of carbon provides an energy source for the food web (Tranvik, 1989; Pace *et al.*, 2004), resulting in a net flow of CO_2 from lakes to the atmosphere (Cole *et al.*, 1994). Small lakes in particular are considered to be hot spots of carbon metabolism (Cole *et al.*, 2007). Flocculation, coagulation and sedimentation of organic matter in the water column and its sequestration into the bottom sediments are additional pathways (Molot & Dillon, 1996; Einsele *et al.*, 2001; von Wachenfeldt *et al.*, 2008b). Storage in lake sediments has been estimated to be a major carbon sink in boreal areas (Arvola *et al.*,

2002; Kortelainen *et al.*, 2004). Finally, roughly half of the organic matter may be exported by stream transport to the sea (Baker & Spencer, 2004; Cole *et al.*, 2007).

2.3 Classification of lakes

Despite early recognition of the importance of allochthonous organic carbon (Thienemann, 1921; Birge & Juday, 1927; Naumann, 1929) many limnologists excluded DOC as a parameter (Jones, 1992). Rohde (1969) tried to incorporate the concept of dystrophy into the established nutrient-based classification system defined by Vollenweider (1968). Rohde's concept was based on a scheme with lake types divided along two gradients: a gradient of autotrophy where lakes went from oligo- to eutrophic, a gradient of allotrophy where lakes went from oligo- to dystrophic and an intermediate group of mixotrophic lakes with auto- and allotrophic conditions. The role of DOC in lakes was largely eclipsed by the chlorophyll-phosphorus relationship and research on the control of eutrophication of lakes formed the basis of many management programs in aquatic systems (Vollenweider, 1968; Dillon & Rigler, 1974; Vollenweider & Kerekes, 1980; Nu rnberg, 1996; Schindler, 2006). This research focused mostly on *in situ* primary production, which is measured via planktonic chlorophyll, total phosphorus and water transparency in lake water (Carlson, 1977; OECD, 1982). Four distinct trophic state levels (oligo-, meso-, eu- and hyper-eutrophic) were determined from summer epilimnetic nutrient, chlorophyll (chl) concentration and Secchi disk transparency (Nu rnberg, 1996). The OECD classification was subsequently modified in Ireland because the usual frequency of sampling of lakes did not generate sufficient data to permit calculation of the annual mean values as specified in the OECD scheme. For this reason, the Irish EPA use a modified, less statistically reliable (Irvine *et al.*, 2001), trophic classification scheme based on the annual maximum chlorophyll concentration (Toner *et al.*, 2005).

While many lake studies concentrated on nutrient state and trophic classification some notable publications also incorporated examination of DOC. Studies by Wetzel (1983), Häkanson & Jansson (1983) and Thurman (1985) were seminal publications. Häkanson & Jansson (1983) modified Rodhe's scheme (1969) relating it to autotrophic production and allotrophic inputs of organic matter, suggesting that dystrophic lakes are rich in humic materials and are generally low in internal autotrophy carbon production.

Thurman (1985) proposed four trophic states based solely on DOC concentrations of lakes (Table 2.1) and showed that DOC varies with the productivity of the lake and increases with trophic status. Coincidentally an increasing understanding of the role of microbial cycling of detrital organic matter (Pomeroy, 1974; Azam *et al*., 1983; Scavia & Laird, 1987) helped to resurrect interest in defining how allochthonous organic carbon supports both the microbial and metazoan food webs in lakes (Sherr, 1988).

Table 2.1 - DOC of lakes of various trophic states (Thurman, 1985).

Trophic State	**Mean DOC (mg L^{-1})**	**Range (mg L^{-1})**
Dystrophic	30	20-50
Oligotrophic	2	1-3
Mesotrophic	3	2-4
Eutrophic	10	3-34

Williamson *et al.* (1999) argued that phosphorus and DOC should both be considered for proper lake characterization, since phosphorus load accounts for the effects of eutrophication, whereas DOC influences light, oxygen and temperature profiles, as well as toxin availability and acidity. The research group used Rodhe's (1969) model as starting point and included total phosphorus (TP; µg L^{-1}) and coloured dissolved organic carbon (CDOC; absorbance at 320 nm m^{-1}) to classify lake trophic state (Figure 2.2). A data-set of seven lakes was assembled to demonstrate the differences that exist among lakes and permit separation into four classes (oligo-, eu-, mixo- and dystrophic).

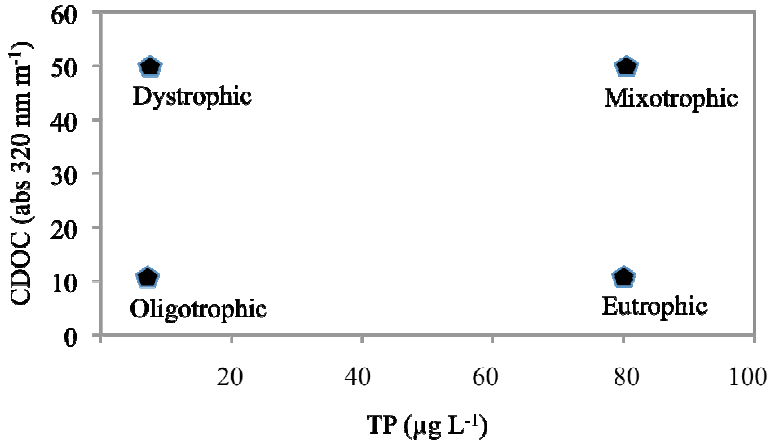

Figure 2.2 – Classification of trophic state according to CDOC and TP (from Williamson *et al.,* 1999; page 799).

In Scandinavian countries water colour has been included historically as a parameter in determining ecological status of lakes, e.g. reference conditions. For example, in

Finland Pilke *et al.* (2002) divided lakes with a surface area > 40 km^2 and a mean depth > 3 m into two types: oligohumic (water colour <30 mg Pt/Co L^{-1}), and humic (water colour 30 – 90 mg Pt/Co L^{-1}) lakes. Similarly, Lepistö *et al.* (2004) differentiated between oligohumic (water colour <60 mg Pt/Co L^{-1}), humic (60-120 mg Pt/Co L^{-1}) and dystrophic (> 120 mg Pt/Co L^{-1}) lakes. In fulfilment of WFD requirements working groups for intercalibration and benchmarking were set up in recognition of the need to differentiate Europe into different Geographical Intercalibration Groups (GIG) characterized by specific descriptors (Poikane, 2009). Ireland now contributes to two geographical groups: the Northern (country members: parts of Finland, Sweden, Norway, UK, Ireland) and the Atlantic GIG (parts of the UK and Ireland). The Northern GIG identified seven lake types characterized by five descriptors, namely lake size, altitude, lake mean depth, alkalinity and water colour, while the Atlantic GIG includes lake size, altitude, mean depth and alkalinity and excludes water colour. Thus, the WFD now explicitly recognises the role of colour in the geography of northern Europe.

2.4 The role of DOC in aquatic ecosystems

Particulate and especially DOC from auto- and allochthonous sources regulate the material and energy fluxes in lake ecosystems. Any change in DOC concentration has an impact on physical, chemical and biological behaviour of the aquatic system. It is important to note that the effects of DOC are inter-linked.

2.4.1 Physical effects

2.4.1.1 Effect of DOC on lake thermal properties

DOC is strongly related to light penetration in aquatic ecosystems and has a direct impact on the heat absorption of humic material, which is linked to thermocline depth, and thus, to the mixing depth of lakes (Williamson *et al.*, 1999; Hudson *et al.*, 2003; Maloney *et al.*, 2005). Thermal stratification in clear lakes develops slower in spring than in humic lakes (Bowling & Salonen, 1990). The establishment of the thermocline in the uppermost water layers, consequently results in steeper and shallower thermal gradients characterized by increased stability (Salonen, 1984; Jones, 1992). Fee *et al.* (1996) found a positive correlation between lake size and light transmission. Surface area represents the most important determinant in epilimnetic depth in lakes > 500 ha.

In contrast, in small lakes (surface area < 500 ha) water transparency is a more relevant factor than surface area (Perez-Fuentetaja *et al.*, 1999; Snucins & Gunn, 2000).

2.4.1.2 Attenuation of solar radiation

The transparency of water is determined by the relative light penetration and is measured by calculating the ratio between the irradiance observed at a given depth and that recorded at the water surface. Transparency, frequently estimated as Secchi disc depth, is used to define the euphotic depth and marks the lower boundary of the layer in which net photosynthetic production is possible (Håkanson & Peters, 1995). This corresponds to approximately 1% of full daylight (wavelengths of 400-750 nm) and is also referred as attenuation depth of photosynthetically active radiation (PAR). In humic lakes the euphotic zone can generally be equated with Secchi disc visibility (Arvola *et al.*, 1999a). In humic lakes the depth of PAR is strongly attenuated. The red part of the light spectrum is dominant, due to the presence of humic substances and thus, the illuminated water layer is shallower than in clearwater lakes (Kirk, 1994a).

While the zone of photosynthesis and the euphotic zone are not synonymous, they often coincide. The illuminated layer corresponds to the photic zone, while the non-illuminated layer is called aphotic zone and the twilight layer between them is termed the dysphotic zone (Arvola *et al.*, 1999a). In the most transparent oligotrophic lakes, the euphotic zone extends to depths > 10 m (Jones, 1992) and in some cases it may extend up to 30-50 m depth (Snucins & Gunn, 2000). In contrast, in humic lakes with DOC concentrations of 10-15 mg L^{-1} the euphotic zone is between 1 and 2 m deep, while in highly humic lakes (DOC > 15 mg L^{-1}) it rarely exceeds one metre (Jones, 1992; Lindell *et al.*, 1996; Löfgren *et al.*, 2003). The presence of DOM in water impacts on the biological and chemical behaviour of the water body by absorbing radiant light from the water column, decreasing that available for photosynthesis (Ferrari *et al.*, 1996)

2.4.2 Chemical effects

2.4.2.1 Acidification and oxygen depletion

Lake water acidification has been attributed to natural and anthropogenic drivers (Schindler, 1996a; Williamson *et al.*, 1999). Acidification of a lake occurs, for example, when a catchment area receives acid loading at levels such that the natural buffering

capacity is exceeded or when the chemical equilibrium is altered as a result of progressive decrease in exchangeable cations (Cresser & Edwards, 1987). When a lake cannot completely neutralise increasing acidity there is a net increase of H^+ ions. This process is known as acidification. In general, humic substances are a natural source of acidity in inland waters (Steinberg, 1991). High concentrations of DOC compounds contribute to the naturally low pH (Kortelainen & Mannio, 1990; Lydersen, 1998). Transient increases in DOC concentration cause a pH decline in surface waters (Laudon *et al.*, 2001) or show no correlation (Worrall & Burt, 2004d). Other studies have documented increased water transparency (associated with declines in DOC and colour and increased penetration of UV radiation) with increases in surface water acidity (Gjessing, 1992; Leavitt *et al.*, 1997).

DOC concentration influences the rate of oxygen depletion in lakes through photochemical oxidation of the organic material (Lindell & Rai, 1994) and can reduce the maximum depths of oxygenation and, therefore, contribute to hypolimnion anoxia and have impacts upon aquatic life (Baker & Spencer, 2004).

2.4.2.2 Nutrient availability and reduced toxicity of metals

DOC serves also as a carrier for nutrients and thus, influences their concentrations and bioavailability (Jones, 1998; Perdue, 1998; Shaw, 2000). Nitrogen (N) is the most common limiting nutrient in terrestrial forest ecosystems in temperate regions. The export of inorganic N to aquatic habitats is therefore small and the bulk of dissolved N is bound in humic substances (Jansson *et al.*, 1996). Total N (TN) concentrations in temperate brown water lakes are typically 300-500 $\mu g\ L^{-1}$, while inorganic N fractions are close to detection limits (Stepanauskas *et al.*, 1999). Total phosphorus (TP) concentrations can be high in humic lakes (10-25 $\mu g\ L^{-1}$), however most of the P binds with humus colloids forming iron-phosphorus-humus complexes and affecting the bioavailability of key limiting elements (Ohle, 1935; Tipping, 1981). According to Sakamoto (1966) and Smith (1982) nitrogen to phosphorus ratios (N/P) can give an indication of which nutrient is limiting in an aquatic system. Lakes can be characterized as phosphorus limited ($N/P > 17$), simultaneously nitrogen and phosphorus limited ($10 < N/P \leq 17$) and nitrogen limited ($N/P < 17$).

DOC also binds and transports pollutants, toxic organics and metals (aluminium, iron, chromium, lead, mercury) and radionuclides, and thereby, reduces their dissolved concentrations and bioavailability to aquatic biota (Francko, 1986; Tessier, 1992; Perdue, 1998; Shaw, 2000). Humic substances also bind with contaminants, including the known carcinogen benzopyrene (a product of combustion) and various pesticides, and are capable of altering their chemical reactivity and of reducing their toxicity and bioaccumulation (Oris *et al.*, 1990; Gensemer *et al.*, 1999; Akkanen *et al.*, 2004).

2.4.3 Biological effects

It is well known that lake ecosystems have two main sources of energy: autotrophic phyto- and picoplankton, that use light as their energy input, and heterotrophic bacteria, that exploit the organic matter available in the water column (Jones, 1992; Salonen, 1992a; Pace *et al.*, 2004; Carpenter *et al.*, 2005; Jones, 2005; Cole *et al.*, 2006). These authors recognized that bacteria are not only involved in a microbial loop within a pelagic ecosystem, but also form a link between external primary producers and the pelagic food web. Consequently, both communities are characterized by similar functional roles and both supply higher trophic levels with energy, whether via a physical (light) or a chemical form (DOM). A considerable part of the bacterial production in lakes may be channelized toward higher trophic levels via micrograzers (Kankaala *et al.*, 1996; Isaksson *et al.*, 1999; Jansson *et al.*, 1999), such as hetero- and mixotrophic flagellates (Isaksson, 1998; Isaksson *et al.*, 1999), ciliates (Sanders *et al.*, 1989; Hessen *et al.*, 1990; Havens, 1991; Carrias *et al.*, 1996), rotifers (Arndt, 1993) and cladocerans (Hessen, 1998). Finally, some of the carbon passing up the food chain will be returned to the carbon pool by excretion (Jones, 1992).

Whether auto- or heterotrophic production dominates in aquatic ecosystems has fundamental consequences for carbon processing (Bass *et al.*, 2010). Net autotrophic systems will be sinks for atmospheric CO_2, while net heterotrophic systems will egress CO_2 to the atmosphere (Cole *et al.*, 1994; del Giorgio *et al.*, 1999; Ojala *et al.*, 2011). In lakes the balance between the two is variable and dependent on several factors, including trophic level (Biddanda *et al.*, 2001) and DOC concentration (Blomqvist *et al.*, 2001). Generally, eutrophic systems are dominated by autotrophic processes, while oligotrophic systems are dominated by heterotrophic processes (Sobek *et al.*, 2005).

However, both net heterotrophy and net autotrophy have been documented in oligotrophic aquatic systems (del Giorgio *et al.*, 1999; Carignan *et al.*, 2000). Consequently, bacterial production and respiration in humic lakes is frequently higher than the authochthonous primary production, even in the euphotic zone during summer (Salonen, 1984, 1992a; Drakare *et al.*, 2002). Jansson *et al.* (2000) suggested that the shift from net autotrophy to net heterotrophy might take place at concentrations of DOC around 5 mg L^{-1}. Recently, Bass *et al.* (2010) reported that the factors affecting the autotrophic/heterotrophic balance are dynamic and additional factors, such as inorganic and organic nutrient supply, may also be influential.

The following sections describe briefly the lacustrine pelagic autotrophic (phyto- and picoplankton), mixotrophic (phytoflagellates) and heterotrophic communities (bacteria and ciliates) highlighting their differences in humic and clear-water lakes.

2.4.3.1 Phytoplankton

In humic lakes phytoplankton communities are typically limited to the uppermost layers of the water column. The presence of humic matter indirectly affects their ability to develop via attenuation of the available light energy (Arvola, 1984; Lindell *et al.*, 1996; Jones, 1998; Arvola *et al.*, 1999b; Löfgren *et al.*, 2003). Regardless of their characteristic physical and chemical features, algal community structure in humic lakes seems to be equivalent to that found in clearwater lakes (Jones, 1998; Arvola *et al.*, 1999b). It appears that the algal community structure is less dependent on water colour (the amount of humic substances) *per se*, but some associated features such as reduced nutrients levels, toxicity of metals and low pH, can influence composition and dominance (Jones, 1992; Jones, 1998). Phytoflagellates have been found abundant in humic lakes (Ilmavirta 1988, Jones 1991), because they are mobile and have the ability to keep and optimize their vertical distribution in the water column in accordance with the quantity and quality of available resources (light and nutrients) (Morgan & Kalff, 1979; Reynolds, 1984; Salonen, 1984; Dokulil, 1988; Jansson *et al.*, 1996). However, the dominance of flagellates is not a universal feature of humic lakes, because they can also be observed in clearwater lakes (Arvola *et al.*, 1999b). Although humic lakes have no characteristic phytoplankton species composition, Cryptophyta and Chrysophyta commonly contribute to high biomass (Jones, 1998; Arvola *et al.*, 1999b).

Phytoflagellates obtain carbon via auto- and/or mixotrophy. (Jones, 1994; Jansson *et al.*, 2000; Jones, 2000). Mixotrophy encompasses a spectrum of nutritional strategies (Jones, 1994). Certain types of phytoflagellates are capable of obtaining energy and/or nutrients by phototrophic autotrophy (using light energy and inorganic nutrients) and phagotrophic (ingesting particulate matter and bacteria into food vacuoles for subsequent digestion and utilization of derived organic compounds) or osmotrophic heterotrophy (utilizing dissolved organic compounds osmotrophically) (Bird & Kalff, 1986; Isaksson *et al.*, 1999; Stoecker, 1999). Mixotrophy is evident during situations of scarce light conditions (Jones, 1997) and enables phytoflagellates to outcompete purely autotrophic species during nutrient limited conditions (Caron *et al.*, 1990; Jansson, 1998; Bergstro m *et al.*, 2001). This is an advantageous strategy in humic lakes, where access to nutrients is restricted due to nutrient competition with heterotrophic bacteria (Ramberg, 1979; Salonen & Jokinen, 1986; Riemann *et al.*, 1995; Jansson *et al.*, 1996; Jansson *et al.*, 2001). Generally, mixotrophy is seen among dinoflagellates (Gymnodiniales) and certain types of Chrysophyta (*Chromulina, Chrysococcus*, *Dinobryon, Ochromonas* and *Pseudopedinella*) (Porter, 1988; Tranvik *et al.*, 1989; Jansson *et al.*, 1996; Geider & MacIntyre, 2002). Other flagellates have been shown to be potential or facultative mixotrophs, which is generally regarded as a facultative ability to supplement nutrients other than carbon (predominantly N or P) under conditions of limited nutrient availability (Riemann *et al.*, 1995; Gervais, 1997). This latter strategy is not a substitute for autotrophy, but it can provide an energetic subsidy that may be stimulated under certain environmental conditions, such as reduced light or nutrient supply (Gervais, 1997; Li *et al.*, 2000). The potential mixotrophic strategy is evident in certain Chlorophyta (Chlorococcales), Euglenophyta and Chryptophyta (*Cryptomonas* and *Chroomonas*/*Rhodomonas*) (Tranvik, 1989; Lewitus & Kana, 1994; Jansson *et al.*, 1996).

2.4.3.2 Picoplankton

Autotrophic picoplankton, the smallest photosynthetic pro- and eukaryotic organisms, (cell size: 0.2-2 and 3 µm, respectively), represents an important component of freshwater ecosystems (Stockner & Antia, 1986; Callieri *et al.*, 2007b). Picoplankton comprises several groups of algae, but is mainly represented by unicellular, coccoid picocyanobacteria from genus *Cyanobium* (Komárek, 1996) and picoeukaryotes of

Chlorella-like chlorophytes (Stomp *et al.*, 2007). The small cell size of picoplankton cause a high surface-to-volume ratio, which induce efficient nutrient uptake (Fogg, 1986; Zevenboom, 1986). For that reason, in oligotrophic clearwater lakes the biomass of picoplankton frequently dominates the summer phytoplankton biomass (Stockner & Shortreed, 1989; Stockner, 1991; Vörös *et al.*, 1998). In comparison, investigations of picoplankton in humic lakes found low biomass compared to clearwater lakes (Craig, 1987; Kukkonen *et al.*, 1997). Their abundance and biomass, as with other groups of algae, rises with nutrient enrichment (Stockner, 1988, 1991) and may reach a very high biomass in highly productive lakes (Ju rgens & Jeppsen, 2000; Callieri & Stockner, 2002a). In humic lakes picoplankton production can be restricted either by poor light availability (Eloranta, 1978; Arvola *et al.*, 1999a) or by inorganic nutrient limitation (Meili, 1992) or by both these factors (Drakare *et al.*, 2002, 2003).

2.4.3.3 Heterotrophic bacteria

In clearwater systems, phytoplankton account for most of the mobilization of carbon that later, via different processes including exudation, autolysis and grazing, becomes available to bacterioplankton growth. In those systems the relationship between phyto- and bacterioplankton can be described as a microbial loop with a strong dependence of bacteria on algae derived organic carbon (Azam *et al.*, 1983). In contrast, high pelagic bacterial biomass and production have been reported from humic lakes, where allochthonous DOM is the dominating carbon source (Tranvik, 1988). In these systems bacteria are no longer dependent on carbon mobilized by primary producers (Tranvik, 1988; Jansson *et al.*, 1999; Bergström & Jansson, 2000a).

In the photic zone of humic lakes the relationship between phyto- and bacterioplankton can be described as a competition for inorganic nutrients between two alternative energy mobilizers at the base of the food chain (Currie & Kalff, 1984; Hessen *et al.*, 1994; Jansson *et al.*, 1996; Jansson, 1998). Because of their larger area-to-volume ratio and high uptake capacity for nutrients, bacterioplankton are generally thought to be the better competitors (Bratbak & Thingstad, 1985; Tranvik, 1992; Jansson *et al.*, 1999; Jansson *et al.*, 2001). Bacterial biomass and production can be an order of magnitude higher in the hypolimnion than in the epilimnion (Arvola *et al.*, 1992). In the hypolimnion, large-sized phototrophic bacteria usually form thin and dense layers at

depths with sufficient irradiation (Salonen, 1992a). These bacterial layers provide extra food for migrating zooplankton (Salonen & Lehtovaara, 1992) and protozoa. If the hypolimnion is included, bacterial biomass and production become dominant in the majority of humic lakes, especially if the whole year is considered (Nürnberg & Shaw, 1999).

2.4.3.4 Ciliates

Most planktonic ciliate taxa are obligate heterotrophs, obtaining resources by phagotrophic ingestion of particles. Šimek *et al* (1996) found that ciliates can meet all their carbon requirements with an exclusive diet of picoplankton, while other studies revealed that ciliates prey also on bacteria (Hessen *et al.*, 1990; Havens, 1991) and algae (Jones, 2000). As a result, ciliates serve as a link to higher trophic levels (Stockner & Porter, 1988) and play an important role in the recycling of nutrients (Berman *et al.*, 1987; Caron *et al.*, 1988; Martin-Creuzburg & Von Elert, 2006). Moreover, some taxa have been shown to sequester plastids from ingested algal prey (Rogerson *et al.*, 1989; Jones, 2000). In these cases, the sequestered plastids continue to photosynthesize and are capable of an appreciable contribution to the carbon requirements of the ciliate (Blackbourn *et al.*, 1973; Finlay & Esteban, 1998).

2.5 Drinking water

An increase in natural organic matter concentrations has implications for the ecology of aquatic environments, and also for drinking water supplies. The removal of DOC from water sources represents one of the major costs of water treatment (Worrall *et al.*, 2004c; Worrall *et al.*, 2008). Dissolved organic substances in drinking water are important primarily because of their potential impacts on human health and secondly due to the aesthetic quality of drinking water (Janus, 2010). Additionally inorganic dissolved compounds, for example iron and manganese, impart a dark colour to water. During the last three decades different water treatment plants in Nordic countries have experienced difficulties in treating humic water due to a change in quality and quantity of organic matter (Rodriguez & Serodes, 2001a; Löfgren *et al.*, 2003; NORDTEST, 2003; Sharp *et al.*, 2006). Drinking water treatment revolves around three main pollutants : bacterial and protozoan pathogens, dissolved substances and organic

precursors of disinfection by-products (DBPs) and nutrient levels, which are regulated by the European Drinking Water Directive (1998/83/EC).

2.5.1 Bacterial and protozoan pathogens

Throughout the twentieth century maintenance of the microbiological quality of drinking and bathing water has been an important means of preventing waterborne diseases. The control of human and animal faecal bacteria (*Escherichia coli, Enterococci*, *Clostridium perfringens*) represents one of the most important human health indicators of drinking water quality. Their presence in drinking water should lead to investigation of potential sources, such as insufficient treatment process or breaches in the distribution system integrity (WHO, 2009). Two examples of pathogenic protozoan are *Cryptosporidium* and *Giardia*. The former is resistant to chlorine disinfections, thus managing this threat to the water supply involves limiting it at its source. This is particularly important in countries were brown waters are common as several studies found a temporal association between turbidity and the incidence of gastrointestinal infections, for example cryptosporidiosis in the treated water (Morris *et al.*, 1996; Schwartz *et al.*, 1997). Several outbreaks of cryptosporidiosis occurred over the last decade in the UK, US (Barrell *et al.*, 2000), Norway, France and Ireland (Glaberman *et al.*, 2002; Pelly *et al.*, 2007; EPA, 2011b). Inadequate filtration processes lead to a major outbreak of cryptosporidiosis in Galway during 2007 for five months, causing illness in over 240 people (EPA, 2011b).

2.5.2 Dissolved substances and organic precursors of disinfection by-products

DOC is of particular concern for drinking water because it has been identified as the principal precursor in the formation of carcinogenic compounds when water is disinfected by chlorination. Chlorine is a frequently used disinfectant in the water treatment process in order to ensure the microbiological safety of the drinking water. However, during disinfection, chlorine breaks down complex and inert organic molecules forming smaller reactive compounds. These compounds react with chlorine to form DBPs, which includes THMs (e.g. chloroform, bromodichloromethane, dibromochloro-methane and bromoform), haloacetic acids (e.g. trichloroacetic acid) and aldehydes (e.g. formaldehyde) (Rook, 1974; Reckhow & Singer, 1990; WHO, 2005). The range of DBPs have been associated with adverse health impacts, including an

increased risk of bladder and rectal cancer (Cantor *et al.*, 1998; Nikolaou & Lekkas, 2001; WHO, 2005) and adverse reproductive outcomes (short gestational duration, low birth weight, short body length and small head circumference) following exposure during pregnancy (Bove *et al.*, 1995; Källén & Robert, 2000).

The European Drinking Water Directive (Council Directive, 98/83/EC) implemented Biocidal Product Guidelines for chemical disinfectants, meant to kill or deactivate harmful or unwanted microorganisms and/or reduce residual concentrations in distribution systems to minimize microorganism re-growth (Rodrigurez & Sérodes, 2001b). The formation of DBPs depends mainly on the amount of raw water DOM (Liang & Singer, 2003), which may vary significantly according to season and geographical location (Clark, 1994). Therefore, several factors, such as water temperature, pH, type of disinfection scenario (e.g. whether coagulation is practiced prior to disinfection), biodegradation of organic compounds amount of chlorine added, travel time of water within the system, can all impact the concentration and distribution of DBPs (Golfinopoulos *et al.*, 1998; Rodriguez *et al.*, 2002; Liang & Singer, 2003; Hong *et al.*, 2007). THM formation increases with an increase in pH, while trihaloacetic acids decrease (Liang & Singer, 2003). Therefore, seasonal variations in DOC require pH corrections during treatment, before treated waters are released for public use (Gregor *et al.*, 1997). An incorrect use of oxidants (e.g. chlorine, hydrogene peroxide) during disinfection may cause damage to human, animal and environment. The European Drinking Water Directive set strict standards for drinking water quality at tap (microbiological, chemical and organoletpic parameters) and restricted the maximum acceptable concentration for total THMs (sum of concentrations of specified compounds) to 100 µg L^{-1} and the concentration of chlorine to less than 250 mg L^{-1} (European Union, 1998).

2.5.3 Nutrient levels

The overabundance of nutrients in water can cause a number of adverse ecological and health effects. Dissolved organic substances generated by phytoplankton can cause taste and odour problems and in some cases, toxicity (WHO, 2009). High levels of nitrate in drinking water may induce “Blue Baby syndrome” (methaemaglobinemia) and may increase mutagenicity, birth defects and contribute to bladder, ovarian and digestive

tract cancer (Camargo & Alonso, 2006). Moreover, eutrophication of surface water often results in algal blooms. Blue green algae or cyanobacteria threaten the drinking water quality by causing physical obstructions to water treatment (Wroath & Fawell, 1995) and some blooms contain species that can produce toxins (WHO, 1999). The most commonly encountered toxins are hepato- and neurotoxins. The former cause acute liver injury on acute exposure (Codd *et al.*, 1999) and the latter induce paralysis of respiratory muscles (Wroath & Fawell, 1995; Codd *et al.*, 1999). A further class of toxic compound associated with some algal blooms are lipopolysaccharides and are capable of causing skin disorders (irritation, rashes and wheals) and various gastrointestinal effects (Codd *et al.*, 1999). The presence of algal toxins in drinking water have been reported annually in different countries around the world (Falconer, 1994; Falconer & Humpage, 2005), including several countries in Europe (Lawton & Codd, 1991; Hoeger *et al.*, 2005; Depla *et al.*, 2009).

2.6 Recent rises in allochthonous organic carbon exports in aquatic ecosystems

A range of studies have demonstrated long-term changes in DOC concentrations in surface waters for a range of sub-boreal countries over the last few decades. Several observations of rising DOC trends in waters draining peatlands have lead to concerns that peatland carbon stores are destabilizing (Forsberg, 1992; Freeman *et al.*, 2001a; Tranvik & Jansson, 2002; Löfgren *et al.*, 2003; Worrall & Burt, 2004a; Worrall *et al.*, 2004b; Evans *et al.*, 2006a; Vuorenmaa *et al.*, 2006). Significant upward trends in DOC concentration in surface water were evident at monitoring sites across northern and central Europe (Freeman *et al.*, 2001a; Hejzlar *et al.*, 2003; Worrall *et al.*, 2004c) and in the northern and eastern US (Stoddard *et al.*, 2003; Monteith *et al.*, 2007; Zhang *et al.*, 2010). Skjelkvåle *et al.* (2001) report increased DOC evident south of 63°N and principally in the west, where snow cover in winter is less. Other studies revealed no overall trend in central Europe (Evans *et al.*, 2005) and in Canada (Jeffries *et al.*, 2003). Worrall *et al.* (2006) suggest that the carbon balance in a peaty catchment is balanced between sink and source, and conclude that peatlands are a smaller carbon sink than previously estimated.

2.7 Potential drivers of change in DOC

The origin and interactions of DOC in hydrological catchments are very difficult to determine because many of the processes occurring are still unknown (Evans *et al.*, 2006a; Roulet & Moore, 2006). The quantity and quality of DOC in aquatic ecosystems varies physically, chemically and functionally from site to site and in time (Thacker *et al.*, 2005). The concentration and contribution of the different carbon sources and the link between terrestrial and aquatic environments depends on the trophic state of lentic ecosystems and on the geographical location. Local variables encompass in-lake and catchment characteristics such as morphometry, lake hydrology, soil factors and land-use and management, while regional variables are related to regional climate (precipitation, temperature, wind) and atmospheric deposition (e.g. decreased sulphur deposition and higher CO_2 levels). A description of each variable is given below. While the variables are discussed individually, it is important to note that they are all interlinked. For example, Roulet & Moore (2006) state that the increases in DOC concentrations should not be attributed to any single factor. Similarly, Evans *et al.* (2006a) argue that the most realistic mechanism to explain the recent rise in DOC concentrations is a complex interaction of changing atmospheric deposition-related and climate-related factors. Also Sucker and Krause (2010) suggest that multiple drivers are required to explain the increases in DOC.

2.7.1 Catchment morphometry and lake hydrology

Several regional studies of humid-zone lakes show that catchment and lake morphometry, are the most important determinants of DOC concentration via their influence on allochthonous inputs (Rasmussen *et al.*, 1989; D'Arcy & Carignan, 1997; Weyhenmeyer & Bloesch, 2001; Sobek *et al.*, 2007). First of all, DOC concentrations are determined by various hydrological characteristics such as riverine inputs and relative rates of loading and in-lake transformations (Engstrom, 1987; Dillon & Molot, 1997b; del Giorgio *et al.*, 1999). Both can vary spatially across lakes (Mazzuoli *et al.*, 2005). Lake DOC is positively related to the lake drainage/lake area ratio and negatively related to catchment slope, residence time, lake area and mean lake depth (Rasmussen *et al.*, 1989; del Giorgio & Peters, 1994; Pace & Cole, 2002; Sobek *et al.*, 2007). Catchment slope directly affects the degree of inundation of catchment soils, which in turn contributes to DOC generation within the catchment (Rasmussen *et al.*,

1989; Xenopoulos *et al.*, 2003). Catchments with steep slopes and porous geological materials tend to deliver their precipitation more directly and rapidly to drainage channels and/or adjacent streams, allowing less soil organic matter to dissolve (Frost *et al.*, 2006). Steep slopes with reduced contact time between water and soil may also limit the potential for removal of nutrients (P and N runoff) to the surface waters (Dillon & Molot, 1990; Maberly *et al.*, 2003). In contrast, lower slopes have impeded drainage and extensive wetlands capable of supplying significant quantities of dissolved humic matter (Gorham *et al.*, 1986; Pace & Cole, 2002; Sobek *et al.*, 2007). Large lakes with long water residence times generally tend to have lower DOC and colour, because of lower areal loading rates and higher in-lake rates of photo-degradation and microbial decomposition (Curtis, 1998; Ko hler *et al.*, 2002; Mazzuoli *et al.*, 2005).

2.7.2 Soil properties and vegetation

Catchment soil characteristics and vegetation affect the terrestrial export of DOC in boreal areas (Hope *et al.*, 1994; Tranvik & Jansson, 2002; Mattsson *et al.*, 2005). Higher concentrations of DOC are common at sites with large stores of soil carbon, such as peatlands, wetland and dense forests, and especially where runoff is low (Dillon & Molot, 1997a; Gergel *et al.*, 1999; Laudon *et al.*, 2004). Low DOC concentrations are found in regions with sparse vegetation and poorly developed organic soils (Löfgren *et al.*, 2003). Spatial variation in the export of DOC among catchments depends on the forest type (Ågren *et al.*, 2007). Leaching is higher from Norway spruce stands compared to Scots pine because of the higher production of litter in the spruce forest floor (Strobel *et al.*, 2001). This is explained by the fact that tree species produce litter with diverse chemical composition and degradability, and these differences influence the composition and reactivity of DOC in soil solutions that get washed out (Strobel *et al.*, 2001). Deciduous tree litter is most easily degraded and yields high DOC run-off (Hongve, 1999; Hongve *et al.*, 2000).

It is estimated that approximately 455 Gt of carbon, representing near to one third of all the carbon present in soils on Earth, is stored in peat (Hope *et al.*, 1994; Moore, 2002). Peat soils are those with an organic content of greater than 25% are formed from the partially decayed remains of living plants in areas of high rainfall and poor drainage (Ingram, 1982). Studies from boreal catchments revealed that peaty soils typically

release CO_2 to the atmosphere and export DOC and dissolved organic nitrogen to the water bodies (Alvarez-Cobelas *et al.*, 2008). Peaty soils export between 10 and 300 kg DOC ha^{-1} $year^{-1}$ into water bodies (Billett *et al.*, 2004; Laudon *et al.*, 2004; Jonsson *et al.*, 2007). Peat soils are found in all latitudes, but the vast majority of them occur at low altitudes. In several Western and Northern European countries as well as parts of Canada, Alaska and Indonesia, peatlands are the most significant wetland environments and represent the largest terrestrial carbon store (Hope *et al.*, 1994; Moore, 2002; Montanarella *et al.*, 2006). Almost one-third (32.6%) of the European peatland resource is present in Finland and approximately 21.5% is in Sweden. The remainder is in Ireland (18.5%), UK (18.3%), Estonia (16%), Faeroe Islands (6.9%), Norway (6.1%), Netherlands (5.9%) and Latvia (5.3%) (Montanarella *et al.*, 2006). Peat soils are composed of two distinct horizons (acro- and catotelm) and are characterized by hydrologic conductivity (Evans *et al.*, 1999). The acrotelm is an upper horizon of roots and decomposing plant material, while the catotelm comprises dense peat and is anoxic for most of the year. When the water table falls, the soil moisture content of the acrotelm decreases and consequently aerobic decomposition and oxidation occur causing a decrease of DOC compounds in soil pore water (Clark *et al.*, 2005).

2.7.3 Land use and management

DOC production and concentrations in freshwater ecosystems may vary according to land use changes and management (Chantigny, 2003; Worrall *et al.*, 2003b). Land use changes, associated with forestry practices, burning of grassland and peatlands, draining and extraction of peatlands, or changes in grazing regimes, industrial activity, agricultural and domestic waste can influence the retention and the export of organic carbon from catchments (Worrall *et al.*, 2003a; Evans *et al.*, 2005; Tetzlaff *et al.*, 2007). Generally coniferous vegetation provides a greater DOC input to adjacent lakes than hardwoods and explains the larger proportion of lake DOC variability over time (Cronan & Aiken, 1985; D'Arcy & Carignan, 1997; France *et al.*, 2000; Xenopoulos *et al.*, 2003). Forest fires, deforestation and afforestation schemes can lead to increases in DOC concentrations and nutrient run-off, which may persist for several years in aquatic systems (Carignan *et al.*, 2000; Cummins & Farrell, 2003; DeFries & Eshleman, 2004). In particular, clear-felling operations have been shown to have a range of impacts including increased runoff (Roberts & Crane, 1997), fine sediment mobilization

(Johnson & Whitehead, 1993), nutrient leaching (Rodgers *et al.*, 2010a) and acidification (Neal *et al.*, 1992; Harriman *et al.*, 2003).

Many peatlands across the world have been drained to allow peat-cutting for fuel and to maximise the area of land for agriculture and forestry, or to alleviate floods (Burt, 1995). Changes in land management, can change the balance between anaerobic and aerobic processes in surface layers result in DOC release (Holden *et al.*, 2004; Worrall & Burt, 2004a). However, investigations of the impact of drainage on DOC concentrations have been contradictory with studies documenting increases, decreases and no change in DOC (Adamson *et al.*, 1998; Chapman *et al.*, 1999; Adamson *et al.*, 2000). Increases in grazing intensity can cause severe and irreparable soil erosion and denudation and influence the export of organic carbon from catchments (Garnett *et al.*, 2000; Bragg & Tallis, 2001; Allott *et al.*, 2005). Industrial activity, agricultural and domestic waste can also contribute to DOC present in aquatic environments. This can enter through discharge from point sources or from diffuse sources from indirect leaching (Apsite & Klavins, 1998; Hudson *et al.*, 2007).

2.7.4 Climate and seasons

Climate change presents one of the most severe threats to the future of human society (Fischlin *et al.*, 2007). A growing body of evidence suggests that climate change over the last two centuries has moved beyond the range of natural variability (Bengtsson *et al.*, 2006; IPCC, 2007). Climate change appears spatially and temporally highly variable (IPCC, 2001; 2007) and may be non-linear (Schindler *et al.*, 1997; Porcal *et al.*, 2009). According to the last IPCC Assessment Report (2007) global surface air temperatures in the last two decades (1995-2006) are among the highest on record since 1850. During the past 100 years precipitation patterns have changed significantly in many parts of the globe with respect to its amount, intensity, frequency and type (Freeman *et al.*, 2001b; Evans *et al.*, 2006a; Frei *et al.*, 2006; Beniston *et al.*, 2007; IPCC, 2007; Planton *et al.*, 2008; Fealy *et al.*, 2010). In northern Europe, average precipitation has increased, while it has decreased in the Mediterranean (IPCC, 2007). These tendencies may be associated with changes in the North Atlantic Oscillation (NAO) (Ottersen *et al.*, 2001), a north-south dipole in sea-level pressure across the Atlantic (high-pressure zone centred over the Azores and low-pressure zone over Iceland), which has its strongest

signature in winter (Hurrell *et al.*, 2003). NAO influences inter-annual and multi-decadal variability in the North Atlantic Ocean (Hurrell, 1995; Hurrell & Deser, 2009). During NAO positive phases, stronger atmospheric pressure gradients between the sub-polar and subtropical region increases winter storm frequency and shifts the Gulf Stream current northward. During NAO negative phases, the Icelandic atmospheric low pressure shifts the winter storm tracks southward, while winter storms tend to be fewer in number and the Gulf Stream current shifts southward (Hurrell *et al.*, 2001; Marshall *et al.*, 2001). Variations in the latitudinal position of the Gulf Stream current is a response to fluctuations in NAO two years previously and, to a lesser extent, to the El Niño/Southern Oscillation (Taylor & Stephens, 1998; Taylor & Gangopadhyay, 2001).

Total solar radiation and climate variables (precipitation and temperature) are key variables affecting lake and stream DOC concentrations (Bertilsson & Jones, 2003; Hudson *et al.*, 2003; Lennon, 2004; Molot *et al.*, 2005). Solar radiation provides the necessary energy to break down the double bonds of DOC (Wetzel, 2001). These photochemical processes (photo-bleaching and photo-degradation) are known to change the optical properties of coloured DOC in lakes and induce a reduction in DOC of c. 20-60% over a period of 11-70 days (Curtis & Schindler, 1997; Moran *et al.*, 2000; Molot *et al.*, 2005; Shiller *et al.*, 2006). Air temperature and precipitation strongly influences both the production and transport of DOC from the catchment to surface waters. Upward trends in air temperature and incident solar radiation may indirectly influence DOC export by altering decomposition processes and mineralization of organic matter. Changes in temperature and consequent soil moisture level have direct impacts on decomposition processes (Worrall *et al.*, 2006). Periods of drought, related to regional changes in climate, may either increase DOC concentrations in lakes (Forsberg, 1992; Worrall & Burt, 2004d) or reduce them (Schindler *et al.*, 1997). Variations in temperature lead to differences in the contribution of aerobic and anaerobic decomposition in high organic soils (Chapman & Thurlow, 1998). Decomposition processes are greater on forested peat than on virgin peat and the differences in rates are linked to the impact of drainage at the forested site (Byrne *et al.*, 2001). However, the concentration of DOC in soils and in stream-waters may not always show an immediate response to a rise in temperature (Clark *et al.*, 2005; Fro berg *et al.*, 2006). This implies lags in either the population size or activity of soil biota or the kinetics of DOC release (Clark *et al.*, 2005). The amount of delivered DOC also depends on the length of

the soil-drying period, particularly in waterlogged soils (e.g. peat bogs) (Fierer & Schimel, 2002). During dry periods the water table falls, aerobic decomposition increases and the solubility of DOC decreases, contributing to lower DOC concentrations (Clark *et al.*, 2005). After a long drying period, rainfall events re-saturate the soil and the DOC, iron and aluminium gets washed out rapidly by hydrophobic re-wetting of the peat matrix (Mitchell & McDonald, 1992; Buffam *et al.*, 2001). This may be explained by longer residence time in the acrotelm and that this is then reflected in the chemistry of the runoff (Evans *et al.*, 1999; Fenner *et al.*, 2001; Hurst *et al.*, 2004; Worrall *et al.*, 2004c; Chow *et al.*, 2006; Worrall *et al.*, 2006).

The effect of precipitation on lake DOC concentrations is complex because catchment properties (e.g. the proportion of wetland and land use, vegetation type and soil properties) influence and affect the DOC loads (Wetzel, 2001; Bertilsson & Jones, 2003). Any variation in timing and intensity of regional precipitation usually alters the water budget and discharge of organic and inorganic matter and nutrient run-off from terrestrial into aquatic systems (Forsberg, 1992; Hongve *et al.*, 2004; Dillon & Molot, 2005; Erlandsson *et al.*, 2008). The relationship between rainfall and/or snowmelt and lake DOC concentration can be strong (Correll *et al.*, 2001) or weak (Spitzy & Leenheer, 1991), positive (Reche & Pace, 2002; Worrall *et al.*, 2002; Arvola *et al.*, 2004), or negative (Sobek *et al.*, 2007). Worrall *et al.* (2002) examined the release of DOC from upland peat in northern England during the autumn flushing and exhibited three hydrologically distinct fractions. The first fraction was low in DOC and was related to rainwater, which had little contact with the soil. The second was also characterized by low DOC levels but originated from old groundwater and had largely been exhausted of DOC. The third fraction had high DOC concentrations supplied by the surface peats, which had become a site of oxidation between rainfall or flushing events and, thus, had a high supply of available carbon.

2.7.5 Atmospheric deposition

A series of studies have proposed that some of the increasing aquatic DOC concentrations may be linked to recent decreases in anthropogenic acidification of surface waters associated with decreases in industrial emissions (Evans *et al.*, 2006a; de Wit *et al.*, 2007; Monteith *et al.*, 2007; Erlandsson *et al.*, 2008). Accumulations of

deposited sulphur and nitrogen potentially increase DOC concentrations because of changes in pH (Schindler *et al.*, 1997). These changes can influence the solubility of DOC compounds, accelerate soil microbial decomposition, stimulate nitrogen limited forests and ground flora and give rise to increased primary production, more litter and consequently, more humic material (Krug & Frink, 1983; Clark *et al.*, 2005; Findlay, 2005; Evans *et al.*, 2006a; Monteith *et al.*, 2007). Ireland has been proposed as an unpolluted reference for European studies (Beltman *et al.*, 1993) as it has limited exposure to trans-boundary air pollution (Aherne & Farrell, 2002).

2.8 Palaeolimnology

Lakes act as a collection point for materials originating within lake basins themselves, their catchment and atmosphere (Likens, 1979; Wetzel, 1983). Lake sediments can provide a temporal perspective (or archive) of a vast range of physical, chemical and biological parameters, and indirectly of their driving factors (Battarbee, 1999). In order to reconstruct a lake's history from sediment cores in an accurate and holistic manner, a range of elements are usually quantified such as the chronology of the sediment core together with the physical (e.g. textural analysis) and geochemical (organic and inorganic) features and preserved biological fossils and/or remains (plant macrofossils, pigments, diatoms, cladocera remains, chrysophyte scales, cysts, pollen and spores) (Blomqvist & Håkanson, 1981a; Battarbee, 1991; Kilham *et al.*, 1996; Lotter & Bigler, 2000; Rautio *et al.*, 2000; Hausmann & Pienitz, 2009). The physico-geochemical and biological changes are then situated in time through the establishment of a core chronology using dating techniques.

Multivariate techniques enable to explore the relationships between and within three taphonomic units (plankton, traps and surface sediments) and to quantify the role of dissolution on diatom assemblages (Cameron *et al.*, 1999; Ryves *et al.*, 2003). Sediment traps enable estimates of loss of material from the trophogenic zone (the upper portion of the lake were photosynthsis occurs) or accumulation of materials in the sediments for both short term and long-term studies (Kirchner, 1974; Smol, 1990; Ryves *et al.*, 2003; Allott *et al.*, 2005) in deep (Ryves *et al.*, 2003) and shallow lakes (de Vicente *et al.*, 2006), rivers (Evans *et al.*, 2006b), fijords (Zajączkowski, 2002) and coastal and marine environments (Rutten *et al.*, 2000; Kato *et al.*, 2003). Generally one or more sediment

traps are installed at a certain or at different water depths in the deepest part of the lake (Lotter & Bigler, 2000; Hausmann & Pienitz, 2009). Sediment traps are important tools for examination of sinking loss rates and measuring daily, seasonal and/or annual fluxes of particles through the water column (Bloesch & Uehlinger, 1986; Horn & Horn, 1990; Agbeti *et al.*, 1997), for the study of the pattern of sediment accumulation (Weyhenmeyer *et al.*, 1995) and sediment resuspension in lakes with different morphometry (Steinman & Parparov, 1997; von Wachenfeldt & Tranvik, 2008a). The sediment trap technique has been used successfully to investigate seasonal dynamics of phytoplankton (Horn & Horn, 1990; Agbeti *et al.*, 1997), water chemistry and diatom assemblages (Kilham *et al.*, 1996; Lotter & Bigler, 2000; Hausmann & Pienitz, 2009), diatom and zooplankton communities (Rautio *et al.*, 2000) and pollen (Blomqvist & Håkanson, 1981a). Sediment traps are of special value providing an integrated sample of the present day lake material that can be compared with sediment core samples and/or with plankton and benthic samples (Cameron, 1995; Lotter & Bigler, 2000; Köster & Pienitz, 2006).

2.8.1 Chronology

In order to evaluate when changes occur in lakes, and how long certain conditions may persist, it is necessary that sediment cores are dated. An important part of the process is estimating sediment accumulation rates (SAR). Radiometric lead (^{210}Pb), caesium (^{137}Cs) and americium (^{241}Am) methods are used for recent chronologies (ca. 100-150 years). The methods provide the key stimulus for the use of lake sediments allowing to define the timing of ecological change in lakes (Krishnaswamy *et al.*, 1971; Pennington *et al.*, 1973). ^{210}Pb is a natural isotope, while ^{137}Cs and ^{241}Am are artificial radionuclides. The presence of the two latter radionuclides in lake sediments in most cases is related to the nuclear weapon testing maximum of 1963 (Ritchie & Mc Henry, 1990). Additionally, the Chernobyl nuclear reactor accident of ^{137}Cs fallout affected most parts of Europe in 1986 and contributes a second peak in lake sediments. Thus, the presence of two distinct artificial radionuclide peaks along a sediment core provides a valuable independent dating technique to validate ^{210}Pb chronology. For longer timescales radiocarbon (^{14}C) dating permits sediment chronologies up to approximately 50,000 years Before Present (BP) to be estimated.

2.8.2 Sedimentary Organic Matter

Sediment organic matter comprises an important fraction of lake sediments that escaped mineralization during sedimentation (Meyers & Lallier-Vergès, 1999). The primary source of organic matter to lake sediments derived is from the particulate detritus of autochtonous and allochtonous primary producers (Rullko tter, 2000). The primary producers can be divided into two distinct biogeochemical groups: nonvascular algae that encompass little or no carbon-rich fibrous tissues and contain a higher organic nitrogen content, and vascular plants (grasses, shrubs, trees) that contain large proportions of cellulose and lignin and a lower organic nitrogen content. The relative contribution from the primary producers to lake sedimentary records is influenced by lake morphology, catchment topography, palaeoclimatic conditions and the relative abundances of lacustrine aquatic and terrestrial plants (Meyers & Lallier-Vergès, 1999). Therefore, the origin of accumulation of sedimentary organic matter in lakes reveal the types and amounts of original materials covering the spectrum of being predominantly algal in some lakes (C/N ratio < 10) to being largely land-derived (C/N ratio > 20) in others (Lami *et al.*, 1994; Meyers & Lallier-Vergès, 1999; Meyers & Teranes, 2001; Leng *et al.*, 2005). Selective degradation can potentially modify the original C/N ratio of the organic matter, but in lake sediment, the signal appears to be preserved (Meyers, 1994). As an accumulation of 'geochemical fossils', the organic matter content of lake sediments provides information that is important for interpretations of lake palaeoenvironments, histories of regional and continental palaeoclimates, and the natural and human induced changes and impacts in the aquatic ecosystem(s), such as for example eutrophication and changes in catchment vegetation and agriculture (Meyers & Lallier-Vergès, 1999; Meyers, 2003). Moreover, accumulations of sedimentary organic matter in lakes reveal also the degree of alteration and degradation of the material (Meyers & Teranes, 2001). Although, diagenetic processes may alter its original composition, generally most lakes preserve organic matter in the sediment (Meyers, 1994; Leng *et al.*, 2005) where remineralisation rates are slow (Meyers, 2003). The processes of alteration and degradation of organic matter are geographically and temporally variable (Meyers & Teranes, 2001), can vary substantially from place to place within a lake (Anderson, 1990; Tenzer *et al.*, 1997) and are influenced by environmental conditions (Meyers & Lallier-Vergès, 1999).

2.8.3 Biological remains

Comprehensive understanding of a lake and its catchment requires analysis of multiple proxy records, including biological remains. Biological fossils including algal pigment and diatoms are commonly used to reconstruct ecological responses to the water column and surrounding source area.

2.8.3.1 Pigments

All photosynthetic organisms contain one or more pigments (or biochromes) in cell chloroplasts or in extra-cellular sheaths in certain cyanobacteria (Proteau *et al.*, 1993). Their role is to absorb visible radiation at different wavelengths of the visible spectrum for either photosynthesis or protection from damaging levels of light (Rowan, 1989; Porra *et al.*, 1997). Different pigments are characterised by separate absorption spectra that provide an useful aid in pigment identification (Leavitt, 1993). The abundance of pigments varies among cells within the same taxon or between different taxa. The cell pigment content can change in response to various environmental conditions, including irradiance, nutrient status, spectral distribution of light, day-length, diurnal cycle and growth phase (Partensky *et al.*, 1993; Schlüter *et al.*, 2000; Henriksen *et al.*, 2002; Tukaj *et al.*, 2003).

The preserved fossil pigments in lake sediments are derived from planktonic and benthic algal communities, phototrophic bacterial populations (Overmann *et al.*, 1993; Steinman *et al.*, 1998), macrophytes (Bianchi & Findlay, 1993) and may be also present in some invertebrates (Leavitt, 1993; Patoine & Leavitt, 2006). In addition, a further source of pigments may be terrestrial detritus transported from the surrounding catchment or from re-suspended material from the bottom of the lake (Winfree *et al.*, 1997). Phytoplankton pigments can be separated into lipid-soluble and water-soluble compounds. The former compounds are generally used in the study of fossil deposits because they preserve much better in the sedimentary records and include chlorophylls, carotenoids (carotenoids and xanthophylls) and UV-absorbing compounds (Leavitt & Hodgson, 2001b). The lipid-soluble compounds are labile and their individual stability in sedimentary environments has been related to four numerical categories starting from most (1) to least (4) stable. Chlorophylls are vulnerable to oxidative degradation processes, causing the formation of various coloured breakdown products (Leavitt &

Carpenter, 1990b; Hurley & Armstrong, 1991; Bianchi & Findlay, 1993). The loss or modification of different compounds of the complex molecule can determine the formation of pheophytins (loss of the magnesium atom), chlorophyllide (loss of the phytol chain) or pheophorbides (loss of both magnesium and phytol chain). Carotenoids are less labile than chlorophylls. However, they are often broken down to colourless compounds that cannot be detected by regular pigment analysis methods. For example, some xanthophylls, such as fucoxanthin (stability 2) and diadinoxanthin (3), can be easily broken down and therefore be only present in the uppermost part of sediment records, whereas peridinin (4) is rarely preserved in sediment records (Leavitt & Hodgson, 2001a).

The study of pigments has been included in limnological studies and multi-proxy palaeolimnological environmental reconstructions. Pigment analyses have been used to determine the phytoplankton community structure in water samples as a supplement or alternative to microscopical counts (Millie *et al.*, 1993; Leavitt *et al.*, 1999). In palaeolimnological investigations fossil pigments provide information that would be impossible to achieve from other proxies (McGowan, 2007) and are fundamental if no historical phytoplankton counts are available. Sedimentary pigments have proved to be valuable indicators of past phototrophic production and communities (Guilizzoni *et al.*, 1983; Sanger, 1988; Leavitt, 1993; Harris *et al.*, 1996; Leavitt & Hodgson, 2001a). Moreover, because many pigments show a degree of taxonomic specificity, they can be used to map the primary producer community to classes (algal divisions) (Lami *et al.*, 1992; Airs & Keely, 2003). Preserved pigments in the sediment records have been used as indicators of food-web interactions, lake acidification (Guilizzoni & Lami, 1992), eutrophication and land-use practices (Mc Elarney *et al.*, 2009; McGowan *et al.*, 2011), changes in the physical structure of lakes (Hodgson *et al.*, 1998), mass flux within lakes (Ostrovsky & Yacobi, 1999) and climate change (Lami *et al.*, 1996; Lami *et al.*, 1997; Guilizzoni & Lami, 1999; Hall *et al.*, 1999). Pigment breakdown products also provide indications of sedimentary and water column characteristics that regulate pigment transformations (e.g. grazing, anoxia, stratification) and are therefore key indicators of changes in the abiotic and biotic aquatic environment (Hodgson *et al.*, 1998). Palaeolimnological analyses have demonstrated that changes in forest and soil development control dynamics of DOM to rivers and lakes and, thus, the exposure of aquatic biota to ultraviolet radiation (UVR) (Leavitt *et al.*, 1997; Laurion *et al.*, 2000;

Pienitz & Vincent, 2000). Surveys of alpine (Leavitt *et al.*, 1997) and boreal lakes (Donahue *et al.*, 2003) have demonstrated that benthic algae produce specific pigments (called UVR-absorbing compounds) in response to damaging levels of UVR. Those pigments have been used to document historical variations in the intensity of incident UVR of lakes (Garcia-Pichel & Castenholz, 1991; Leavitt *et al.*, 1997; Cockell & Knowland, 1999; Quesada *et al.*, 1999; Leavitt *et al.*, 2003a). The occurrence of UVR-absorbing compounds can be indirectly related to the light climate, depth of euphotic zone and the depth of potentially harmful UVR flux in lakes (Schindler, 1996a; Leavitt *et al.*, 1997).

2.8.3.2 Diatoms

A widely employed approach in palaeolimnology focuses on the fossil remains of diatoms (Bacillariophyta). Diatoms often form a major component of freshwater ecosystems and as such, can be used as valuable indicators of water quality (Hall *et al.*, 1999; Battarbee *et al.*, 2001; Clarke *et al.*, 2005; Bennion & Batterbee, 2007). Since fairly distinct, siliceous cell walls (valves) of diatoms are abundant and well preserved in lake sediment cores (Battarbee, 1986), they are valuable proxies for reconstructing past changes in lake water quality (Battarbee *et al.*, 2001; Stoermer & Smol, 2004; Clarke *et al.*, 2005; Bennion & Batterbee, 2007). Several studies have investigated the potential of diatoms as indicators of trophic state (Lotter *et al.*, 1998; Chen *et al.*, 2008) or climate change (Wunsam *et al.*, 1995; Lotter *et al.*, 1998; Battarbee, 2000). Changes in the diatom flora suggest clear increases in humic matter in rivers and lakes (Engstrom, 1987; Pienitz *et al.*, 1997; Turkia *et al.*, 1998), while others suggest only mild responses (Ro nkkö *et al.*, 1988). The development of multivariate statistics has lead to environmental reconstructions including ecological optima and tolerances of diatom species for several environmental parameters, including pH (Cameron *et al.*, 1999), TP (Lotter *et al.*, 1998; Chen *et al.*, 2008), DOC and dissolved inorganic carbon (DIC) (Pienitz & Smol, 1993; Rosén *et al.*, 2000), epilimnetic water temperature (Pienitz *et al.*, 1995; Weckstro m *et al.*, 1997), air temperature (Rosén *et al.*, 2000) and specific conductivity (Gregory-Eaves *et al.*, 1999) in aquatic ecosystems.

Diatoms can be classified into four life-forms/taxa: planktonic taxa spend their whole life-cycle suspended in the water column, meroplanktonic taxa have some of their life-

cycle resting on the sediment, tychoplanktonic taxa have their true habitat in the benthos, but can often be found resuspended in the water column and benthic taxa live near the bottom of a lake or are attached to the bottom substrate (Stevenson, 1996; Battarbee *et al.*, 2001). Some taphonomic studies show a good agreement between the composition of planktonic diatom populations from the water column and from the sediment record in traps and surface sediments (Cameron, 1995; Lotter & Bigler, 2000; Köster & Pienitz, 2006; Hausmann & Pienitz, 2009), while other studies show considerable differences between diatoms found in the water column and the sediment record (Batterbee *et al.*, 2005c). Cameron (1995) found good agreement between the composition of planktonic diatom populations from the water column and from the sediment record in traps and surface sediments, while other studies revealed considerable differences (Rautio *et al.*, 2000; Batterbee *et al.*, 2005c; Köster & Pienitz, 2006). The annual cycle in a lake can be characterized by diatoms collected in sediment traps and preserved in sediments and thus, reflect seasonal changes in sedimentation (Sommer, 1986; Stoermer, 1993; Cameron, 1995; Lotter & Bigler, 2000; Rautio *et al.*, 2000; Köster & Pienitz, 2006; Kirilova *et al.*, 2008; Hausmann & Pienitz, 2009).

Chapter 3 – Study Sites

3.1 Introduction

This chapter outlines site selection and provides a description of the two catchments and the study lakes. A summary of available data on recent lake chemistry, trophic status and ecology is provided. This is followed by an overview of the climate and weather, geology, soil types and land use.

3.2 Study site selection

The study site selection considered a range of characteristics: first of all, the lakes needed to be surrounded by peat bogs, with data available on physical, chemical and biological parameters. The study lakes would ideally be sources for potable supplies and be included in the EU-funded CLIME project (Climate and Lake Impacts in Europe). The CLIME project simulated the responses of lakes to future as well as past changes in the weather and encompassed several lakes throughout eight European countries. Three Irish lakes were included: Lough Feeagh (County Mayo), Leane (County Kerry) and Poulaphuca (County Kildare). The CLIME project highlighted the impacts of climate change on DOC and its ecological consequences and risks associated in water treatment.

Feeagh was selected for this research as the primary study site due to the distinctively high levels of DOC, the availability of high frequency data since 1996 and the infrastructure and support available from the Marine Institute (MI), Newport. The second study site, Guitane is situated within the Leane catchment, is characterized by lower levels of DOC and thus, more transparent waters. Kerry County Council (KCC) has been monitoring the lake on a monthly basis since 1998 and facilitated fieldwork at the site. The lake is one of the most important drinking water supplies in the southwest of Ireland and was highlighted in the CLIME project as one of the lakes that would require a more detailed investigation (Naden *et al.*, 2010).

3.3 Burrishoole catchment

Lough Feeagh (*Loch Fíoch* in Irish) is situated in the Burrishoole catchment (*Bhuréis Iumhaill*) on the northwest Atlantic coast of Ireland in County Mayo (N 53°56'39'', W 9°34'33''; WFD Code 32_510) (Figure 3.1.a). The catchment is in the Western River Basin District (WRBD) and is situated in a designated Special Area of Conservation (SAC) under the Habitats Directive (92/43/EEC). This SAC, called Owenduff-Nephin Beg Complex (SAC site code 534), is one of the largest (total area of 260.33 km^2) and best Irish examples of active blanket bog (NPWS, 2006). For the Central Statistics Office (CSO), the national office responsible for census for agriculture and population, the catchment is included within Srahmore District Electoral Division (DED).

The catchment lies in a north-south direction and extends over an area of 89.49 km^2 (Figure 3.1.b). It can be divided into two main sub-catchments: Feeagh (67.48 km^2) to the north and Furnace (17.2 km^2) to the south and it is drained by at least 70 km of small shallow streams that make up 30 ha of stream surface area (Poole & de Eyto, 2006). The main rivers are Glenamong, Maumaratta, Altahoney, Galaun, Rough and Lodge. The catchment comprises two major freshwater lakes, Feeagh (394.8 ha) and Bunaveela (45.7 ha), the brackish water tidal lagoon Furnace (167.6 ha) and a few smaller freshwater lakes sited in the uplands (Whelan *et al.*, 1998). Burrishoole catchment communicates with the sea through a c. 4 km long tidal estuary and drains into Clew Bay to the sea. The north-western part of the catchment makes part of the Nephing Beg Range (maximum altitude of 627 m a.s.l.) and is characterized by steeper slopes compared to the north-eastern and eastern part (Allott *et al.,* 2005). The lake provides a source of water to approximately 50 households (Jennings *et al.*, 2010). A further abstraction from Moher Lake supplies the population of Westport. This oligotrophic lake is characterized by a good water quality, however the sampling rate for bacteriological parameters exceeded the regulation requirements (Leslie *et al.*, 2010).

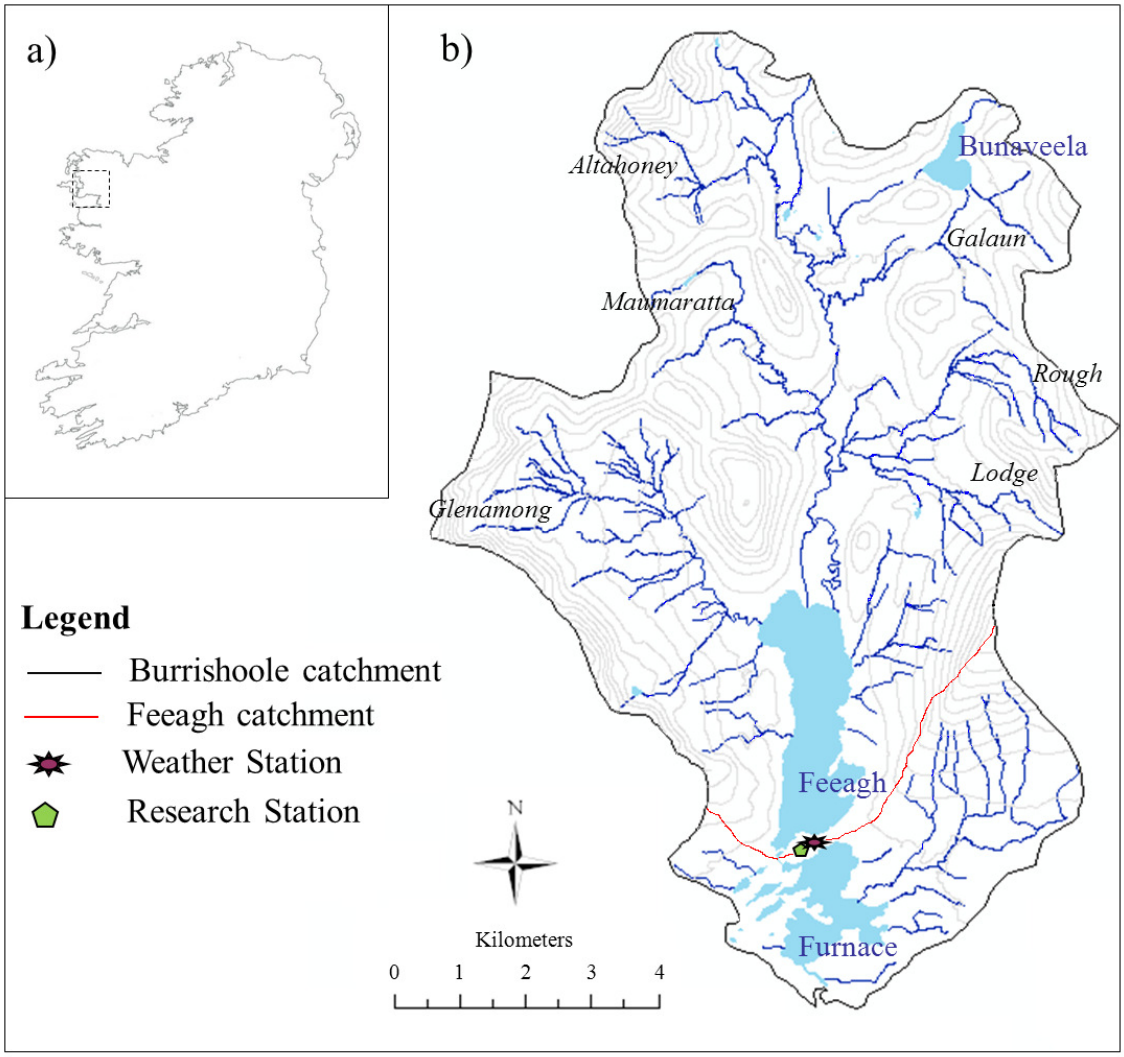

Figure 3.1 – Geographic position of the Burrishoole catchment; b) boundary of the catchment with its lakes, main rivers and location of weather and Research station (Marine Institute).

The Burrishoole is known to be a "data-rich" catchment due to a detailed monitoring programme set up over the last decades. Since 1956 the MI has been an important site for fisheries research and has been recording all migratory fish (salmon, sea trout and eel) to and from the catchment (Whelan *et al.*, 1998; ICES, 2009a, 2009b). Meteorological data have been recorded at the Furnace weather station since 1960. Two high resolution Automatic Water Quality Monitoring Systems (AWQMS) were installed on Feeagh and Furnace in 2003 and 2008, respectively. Monitoring of submerged aquatic plants, macroinvertebrates (White, 2000; Irvine *et al.*, 2001) and pelagic cladocera (MI, unpublished data) has been undertaken. Chydoridae were also investigated and samples are collected monthly (de Eyto, 2000; de Eyto *et al.*, 2002).

Factors influencing the pattern and extent of downstream transport of sediment in the Feeagh catchment were investigated between 2000 and 2001 (Allott *et al.*, 2005). The catchment has also been included in several EU funded international (LIFE, REFLECT, LIFE II, and CLIME) and national research projects (RESCALE, INSIGHT, ILLUMINATE) (Jennings *et al.*, 2000; Allott *et al.*, 2005; George *et al.*, 2005; Livingstone *et al.*, 2005; Rouen *et al.*, 2005; May & Place, 2005a; May *et al.*, 2005b; Leira *et al.*, 2006; Poole & de Eyto, 2006; Blenckner *et al.*, 2007; George *et al.*, 2007; Rodgers *et al.*, 2008; Dalton *et al.*, 2010; Fealy *et al.*, 2010; Jennings *et al.*, 2010; Naden *et al.*, 2010; Rodgers *et al.*, 2010a; Rodgers *et al.*, 2010b; Jennings *et al.*, 2011). In 2007, the catchment joined the Global Lake Ecological Observatory Network (GLEON) (http://www.gleon.org). GLEON aims to collate data from sensors deployed in lakes around the world to address not only local issues for individual lake ecosystems, but also to document regional and global changes in lakes that occur in response to different land-use, latitude and climate regimes.

3.3.1 Lake characteristics

Lough Feeagh (WFD code IE_WE_32_510, Irish Grid Reference F 965 000) lies approximately 200 m upstream of Furnace at an altitude of 11 m a.s.l. The lake has a drainage ratio (drainage area : lake area ratio) of 21.44, a mean depth of 14.5 m and a maximum depth of 45.3 m (Figure 3.2). The annual water residence time is circa 5.4 months (Jennings *et al.*, 2012). The main inflow rivers are the Glenamong, Maumaratta, Altahoney, Galaun, Rough and Lodge (Figure 3.1.b). The main outflows are the Salmon Leap and the man-made Mill Race and both connect Feeagh to the underlying brackish lake Furnace. The lake is composed of two main sub-basins: the deepest one occupies the northern portion of the Lough, while the western side of this basin is steep sided and descends to a depth of 43 m within 180 m of the western shore. A smaller basin lies to the south and reaches a maximum depth of 32 m. The southern and south-western part of the lake is characterized by a undulated floor with a depth varying between 15 and 18 m (Whelan *et al.*, 1998).

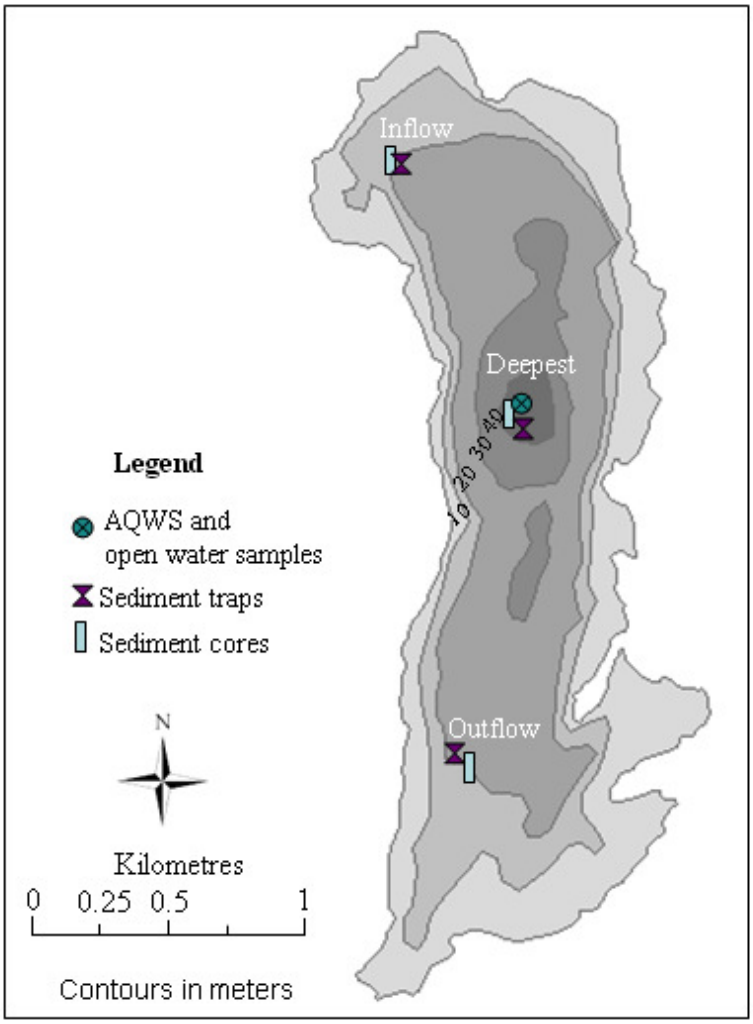

Figure 3.2 – Bathymetric map of Feeagh showing the open water sampling station, sediment trap locations and sediment core collection points.

Feeagh is an EPA typology class 4 lake (deep (average > 4m and maximum depth > 12 m), surface area > 50 ha and low alkalinity (< 20 mg L^{-1} $CaCO_3$) (Taylor *et al.*, 2006)) and its waters are neutral slightly acidic and distinctively coloured, with Platinum Cobalt Units (PtCo) ranging from 80-95 mg L^{-1} and a Secchi depth of 1.6 m (Flangan & Toner, 1975; Free *et al.*, 2006). The waters have low nutrient concentrations with 12 µg TP l^{-1} and < 1 mg TN l^{-1} (Allott *et al.*, 1998; Free *et al.*, 2006). Over 30 years the chl-*a* concentration was low and did not exceed 4 µg L^{-1} (Flangan & Toner, 1975; Free *et al.*, 2006). The most detailed water temperature records available for any Irish lake have been recorded at Feeagh (George *et al.*, 2010). The annual surface temperature generally varies between 3 and 20°C. Feeagh was classified as a monomictic lake (mixes from top to bottom during one mixing period each year) (GLEON, 2008;

Jennings *et al.*, 2012), although the prevailing wind blowing from the sea is readily eroding the seasonal thermocline (Whelan *et al.*, 1998; Poole & de Eyto, 2006). A series of biological surveys were conducted on Feeagh. The first dates back to 1975 when Flanagan and Toner described the planktonic algal communities. A second survey of phytoplankton was conducted in July 2003 (Taylor *et al.*, 2006) and a third between April and October 2007 (Dalton *et al.,* 2010). The EPA included Feeagh in their operational monitoring programme between 2010 and 2012 (EPA, 2010).

3.3.2 Climate

The geographical location of Feeagh on the Atlantic coast favours a typical oceanic climate. The area is highly influenced by the Gulf Stream and the NAO. The mild, moist and extremely changeable type of weather is subject to strong winds, is ice-free during the winter and has relatively cool summers (Jennings *et al.*, 2000; George *et al.*, 2004). Between 1960 and 2009 the weather station measured air temperatures ranging between -8.2°C in February 1969 and 33.9°C in July 2006 (MI, unpublished data). The average annual air temperature was 10.2°C and the annual rainfall was 1,572 mm over the same time-span (MI, unpublished data). The prevailing wind is from the southwest with mean hourly wind speeds of 6 to 7 m sec^{-1} (Healy *et al.*, 1997). Rainfall is generally higher in the northwest of the catchment (c. 1,800 mm $year^{-1}$) and is lower towards the south-east (c. 1300-1400 mm $year^{-1}$) (Allott *et al*., 2005; Dalton *et al.,* 2010). Rainfall can vary considerably from year to year and wet weather can predominate at Burrishoole at any time of the year (Allott, 2005). Typically more precipitation fell during the autumn and winter (September - February) compared to spring and summer (March - August) over the last few decades (MI, unpublished data). An increase in extreme precipitation events during winter, from 3.2 to 7.5, is evident over the period between 1960 and 2009 (Fealy *et al.* 2010).

3.3.3 Geology and soil

The bedrock geology of the Feeagh catchment is dominated by metamorphic rocks of late Precambrian age, consisting of quartzite, schists, gneiss, quartzite and small areas of sandstone and limestone (Parker, 1977; Long *et al.*, 1992). Distinct geological differences divide the western from the eastern sub-catchments: the north-west is composed of quartzite, whereas the west is dominated by a mixture of quartzite, schists

and gneiss, leading to poorly buffered, generally acidic run-off (Whittow, 1974). Carboniferous limestone and sandstone occur on the northern and eastern side of the catchment, specifically around Lough Bunaveela and the Rough River. The eastern part is underlain by quartzite, combined with dolomite bands, volcanic rock, wacke and pure schist (Whelan *et al.*, 1998). Finally, the land bar that separates Feeagh from Furnace is composed of schist and marks the boundary between metamorphic and carbonate lithologies (Whelan *et al.*, 1998). Blanket peat bogs constitute the dominant soil type over the lower slopes of the catchment together with peat podsols, poorly-drained gleys and alluvial deposits (May & Place, 2005a).

3.3.4 Land cover and use

CORINE land cover in the catchment, calculated for 1990 comprises 64% peat bog, 23% forestry, 10% agricultural land and 3% transitional woodland and scrub, natural grasslands and sparsely vegetated areas (Free *et al.*, 2006; Taylor *et al.*, 2006). A comparison of the 1990 and 2006 CORINE data (Appendix A) confirm a decline in forest cover. Pollen records from the Late Glacial suggest the development of forests and woodland, their subsequent decline and the development of peat soils by ca. 5,000 cal yrs BP (Browne, 1986). Census data from CSO show that over the last six decades the primary land-use in the catchment were agriculture and forestry. Mountain sheep grazing, and to a minor extent cattle, represents the most important agricultural activity (Weir, 1996; Whelan *et al.*, 1998; National Parks and Wildlife Service, 2006). The second most important land-use in the catchment is coniferous forestry. Until the 1950s only very small areas of native oak woodlands were present (Fealy *et al.*, 2010). The first important commercial afforestation scheme of Sitka spruce (*Picea sitchensis*), Lodgepole pine (*Pinus contorta)*, Norway spruce (*Picea abies*) and Larch (*Larix* sp.) started in 1951 and continued until the late 1980s (Whelan *et al.*, 1998). Human population has decreased c. 500 to 120 over the last 110 years (Dalton *et al.,* 2010).

3.4 Leane catchment

Lough Guitane (*Loch Coiteain*) is part of the Leane catchment (*Bhuréis Léin*; meaning catchment of learning) and is located in the Killarney Valley in County Kerry in south-west Ireland (52°00’21’N, 9°25’06’’W; WFD Code SW_27_122) (Figure 3.3). The catchment is in the Southern River Basin District (SRBD), lies within the Killarney

National Park, which is Ireland's oldest National Park, and has been recognized as an UNESCO Biosphere Reserve (Fahy & Cross, 2007). Guitane is part of a proposed Natural Heritage Area and part of the Macgillycuddy's Reeks and Caragh River SAC (site code 365) (EPA, 2003; EIS, 2009). The SAC is also part of the NATURA 2000 database (European Council directive, 1992).

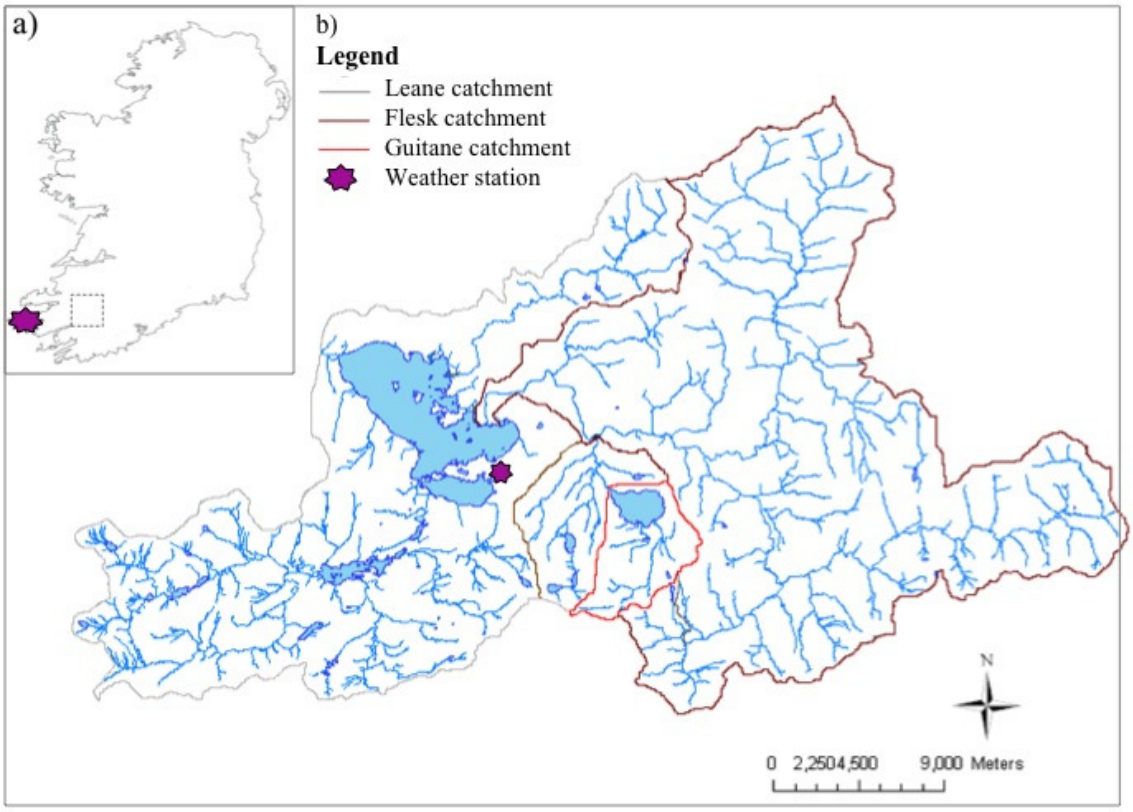

Figure 3.3 a) Geographic position of the Leane and Guitane catchments and Valentia Observatory weather station; b) the Leane catchment and the Flesk (brown line) and Guitane (red line) subcatchments.

The Leane catchment is divided into two main sub-catchments: Leane (210 km^2) and Flesk (325 km^2). The former comprises three main lakes: Lough Leane (Lower Lake; 1987 ha), Upper Lake (1.7 ha) and Muckross (Middle Lake; 275 ha), while the Flesk sub-catchment includes one major lake, Lough Guitane (264 ha), positioned in the southern part of the catchment along with six smaller lakes. For this research the headwater Guitane catchment (12.03 km^2) was considered exclusively. The mountains Stoompa (694 m), Crohane (548 m) and Bennaunmore (454 m) encircle Guitane catchment to the south-east.

KCC has been abstracting water from the Lough Guitane since the early 1980s. Water abstraction from the lake is carried out through a 120 m long pipe, which extends into the lake to reach a depth of 20 m at the northwestern shoreline. The raw water is gravity fed to a storage tank at Sheheree Reservoir, located 4 km to the northwest of the lake. (KCC, 2008). The lake water level is mechanically regulated through a manually operated sluice gate, which ensures fish migration via a fish-ladder (EEA, 2009). Until 1999 an additional water supply was guaranteed from the Owgarriff River, but unacceptable levels of water colour and turbidity forced KCC to use Guitane as the sole source (EEA, 2009). The lake is the largest primary water supply scheme in County Kerry and extracts 51,000 m^3 d^{-1} of water. The treatment plant caters for the water supply requirements of c. 60,000 people. In 2009 a chlorine dioxide disinfection system was installed (EEA, 2009). Guitane is protected under the Drinking Water Regulations (S.I. 439/2000) (European Union, 2000b) and the precautionary principle has been adopted. This prohibits any form of development within the catchment area, precludes new percolation areas for on-site wastewater treatment facilities within 100 m of the shore and requires the installation of additional nutrient reduction measures for all new private development (EEA, 2009).

Detailed limnological and palaeolimnological studies were conducted in the Leane catchment over the last four decades (Murray, 1979; Allott *et al.*, 2001; McClure Morton & Pettit, 2003; Free *et al.*, 2006; Jennings & Allott, 2006; Dalton *et al.*, 2010) as water quality has been declining in recent years (EPA, 2003). A detailed monitoring and management system was set up following severe algal blooms in Lough Leane (Allott *et al.*, 2001; EPA, 2003). The first ecological descriptions for Guitane date back to West & West (1906) and it was not until 1999 that a more complete qualitative and quantitative account of phyto- and zooplankton, benthic profundal and littoral macroinvertebrates was conducted (Twomey *et al.*, 2000). A study on the effect of endocrine disrupting compounds on wild fish populations included Guitane as one of the study sites (Tarrant *et al.*, 2005; Tarrant *et al.*, 2008). KCC have been conducting monthly monitoring of physical, chemical and biological parameters in Lough Guitane since 1999. Guitane has been included in the EPA operational monitoring programme since 2010.

3.4.1 Lake characteristics

Lough Guitane (WFD code IE_SW_22_172, Irish Grid Reference number W 025 845) has a mean depth is 18.7 m and a maximum depth of 56.5 m (Figure 3.4). The lake lies at an altitude of 77 m a.s.l. and its drainage ratio is 7.73. The annual residence time is approximately 5.5 months (KCC, pers. comm.). Four streams discharge into the southern side of the lake. Three are first order streams, while the Cappagh River is the largest stream with a length of approximately 6 km (Figure 3.3.b). The Finow River is the main outflow at the northern side of the lake and flows into the Flesk River, which continues in a south-western direction and flows into Lough Leane. Bare Island, on the northern side of the lake, divides the lake into two sub-basins. The deepest basin lies to the west, while a smaller basin with a maximum depth of 40 m lies to the east.

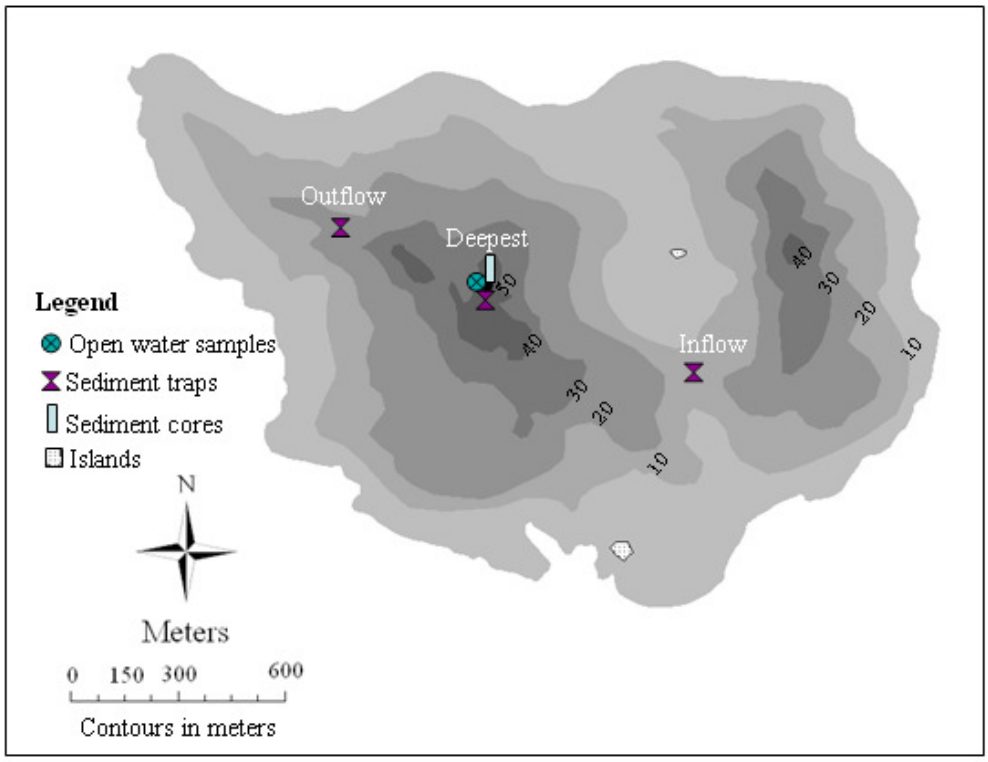

Figure 3.4 – Bathymetric map of Guitane showing the open water sampling station, sediment trap locations and sediment core collection points.

Guitane is an EPA typology class 4 lake (Free *et al.*, 2006) and its waters are described as neutral (6.9-7.1 pH), very soft, transparent in colour (13-20 mg L^{-1} PtCo) with low nutrient and chl-*a* concentrations (TP: 1-5 µg L^{-1}; TN: 0.005-0.25 mg L^{-1} and chl-*a*: 2-2.4 µg L^{-1}) and a Secchi depth of 4.5 m (Caffrey *et al.*, 1999; Free *et al.*, 2006). The

annual surface water temperature in Guitane varied between 5.3 and 25°C between 1999 and 2009 (KCC pers. comm.).

3.4.2 Climate

Similarly to the Burrishoole catchment, the geographic location of Leane catchment is mainly influenced by Atlantic air masses (e.g. NAO) and to a lesser extent, by the latitudinal position of the Gulf Stream (Jennings & Allott, 2006). The more northerly position of the Gulf Stream in early summer contributes to warmer and sunnier weather in the southwest Ireland. The closest weather stations are situated at Muckross and Valentia Island. Valentia Observatory lies off the Iveragh Peninsula in the south-west of County Kerry (Figure 3.3.a) and has been monitoring several meteorological parameters since 1868 (Hickey, 2003). A weather station is located on the south-western shore of Lough Leane (Figure 3.3.b) and is called Muckross House Weather Station. The station is managed by KCC and daily rainfall, minimum and maximum air temperatures (°C) have been recorded since 1999.

Monthly average air temperature data from the Valentia observatory recorded from 1961 to 1990 ranged 6.6°C to 14.8 (NPWS, 2005). A range of -8.8°C and 30.1°C was recorded between 1990 and 1998 at Muckross weather station (KCC, unpublished data). Average annual rainfall of 1,817 mm was measured between 1990 and 2009 (KCC, unpublished data). Allott *et al.* (2008) observed that there was considerable variability across Leane catchment from approximately 1000 mm $year^{-1}$ in the northeast to 2700-3200 mm $year^{-1}$ in the southwest.

3.4.3 Geology and soil

The Guitane catchment straddles a geological fault with its southern part comprising Old Red Sandstone and volcanic rocks that vary in thickness from 90 to 300 meters (Avison, 1984). The northern portion of the catchment is underlain by limestone together with overburden deposits of glacial gravel and boulder clay (Avison, 1984; Pracht & Kinnaird, 1997). The soils in the Guitane catchments are peaty podzols and blanket peat.

3.4.4 Land cover and use

CORINE land cover comprised 75% peat bog, 9% other (sparsely vegetated areas), 8% pasture, 5% agriculture and 3% forestry in 1990 and 2006 (CORINE, 1990; CORINE, 2006) (Appendix B). Blanket peat, together with sparsely vegetated areas and improved agricultural natural grassland occur to the southwest and pasture to the north and northwest. A small patch of broad-leaved forest is present on the south-west shoreline of Guitane. The main land use in the catchment is sheep and cattle farming and to a minor extent, amenity or tourism activities (Jennings *et al.*, 2009). Human population levels encountered in the Flesk subcatchment have increased from 350 in 1996 to 402 in 2011 (CSO, 1991, 2000, 2006, 2011). The population density has been estimated between 10 and 20 people per km^2 with inhabitants more concentrated in the northern sector of the catchment (Clabby *et al.*, 2004). The number of local population kept around 5,000 people over the last eight decades (Dalton, *et al.*, 2010). The Killarney Valley represents one of the most visited tourist venues in the country and attracts approximately 1.5 million visitors per year. Guitane is well known for angling, pony trekking and hiking (National Parks and Wildlife Service, 2005).

Chapter 4 - Materials and Methods

4.1 Introduction

This chapter describes in detail the materials and methods applied in this research. The configuration of the chapter reflects the analytical phases of the research separating the ecological from the palaeoecological methods. Field methods are highlighted first and include instrumental data, open water sampling, sediment trap construction, installation and sample collection and finally sediment core collection and sample extrusion. The second part of the chapter details the laboratory analytical techniques. The final section describes the data analysis techniques used in data exploration. Feeagh was the primary site and was sampled more frequently than Guitane.

4.2 Field methods

4.2.1 Instrumental and measured meteorological and water quality data

Furnace and Muckross House Meteorological Stations, an AWQMS on Furnace and monthly sampling of Guitane generate records of meteorological and water quality parameters collected either on a high (every two-minutes) or low (monthly) frequency (Table 4.1). Data were collated for the relevant time period for this project. Furnace Weather Station collects rainfall data (mm) and the AWQMS records air temperature (°C), wind speed (m s^{-1}), wind direction (°), relative humidity (%), atmospheric pressure (mBar), Photon Flux Density (PFD) (µmol m^{-2} s^{-1}) and photosynthetically active radiation (PAR) (µE m^{-1} s^{-2}). The Muckross House Meteorological Station collects daily rainfall data (mm) and minimum/maximum air temperature (°C). Day length, expressed as hours of light, and daily maximum surface PFD were calculated from this data. Feeagh water quality parameters include Secchi depth (m), thermal vertical temperature profiles from 1 to 42 m depth (with 2 m intervals from 2.5 to 22 m and 5 m intervals to 42 m) and dissolved oxygen (DO) concentration (%), concentrations of chl-*a* (µg L^{-1}), turbidity (Relative Turbidity Unit

(RTU), total suspended solids (mV), pH, conductivity (mS cm^{-1}) at 1 m depth. Water quality parameters measured in Guitane include monthly Secchi depth (m), vertical profiles of temperature (°C) and DO concentrations (%) collected at five-meter intervals from the water surface to a depth of 40 m. High and low frequency ecological data were managed using Microsoft Excel and stratigraphical plots of temperature and dissolved oxygen were constructed with SigmaPlot 11.0 (Systat Software 2008). Thermocline depth was calculated using Lake Analyzer Web (Read & Muraoka, 2011).

Table 4.1 – Overview of the meteorological and water quality parameters, frequency and depth (m) measured in Feeagh and Guitane.

Parameter	**Feeagh**	**Guitane**
Meteorological Data		
Rainfall (mm)	Daily	Daily
Air temperature (°C)	Two minutes	Daily min/max
Wind speed (m s^{-1}) and direction (°)	Two minutes	-
PFD (µmol m^{-2} s^{-1})	Two minutes	-
PAR (µE m^{-1} s^{-2})	Two minutes	-
Water Quality Data		
Secchi depth (m)	Fortnightly	Monthly
Water temperature (°C)	Two minutes (1 – 42 m)	Monthly (0-40 m)
Dissolved oxygen concentration (%)	Two minutes (1 m)	Monthly (0-40 m)
Chl-*a* (µg L^{-1}), turbidity (RTU), TSS (mV), pH, conductivity (mS cm^{-1})	Two minutes (1 m)	-

4.2.2 Water sampling

The fieldwork was organized on both lakes with the support of Marine Institute, Newport and KCC, Tralee. From April 2009 to May 2010 vertically integrated open water samples were collected from the deepest point of each lake on a monthly basis. In Feeagh additional biological samples were collected approximately every two weeks from Feeagh over the whole period. Moreover, monthly samples from March 2008 to April 2009 collected by Marine Institute Newport were also processed for phytoplankton and ciliates. Preserved open water samples were not available for March 2009. In Guitane very poor weather conditions prevented measurements in November 2009 and for that reason a water sample for chemical and biological analyses was collected from the outflow (Finow river). A total of 39 and 12 biological samples were

collected and processed for Feeagh and Guitane, respectively. Water samples were collected using a 2.5 cm diameter tube sampler characterized by two different lengths: 1.5 m for Feeagh and 5 m for Guitane. The different lengths accommodated the average Secchi depths measured over the last 10 years, or the mean depth of the euphotic zone (Håkanson & Peters, 1995; Arvola *et al.*, 1999b). The tube sampler was gently inserted vertically into the water column and the top sealed by covering it tightly with the palm of the hand. The tube and sample were lifted out of the water and the sample was transferred directly into a two-litre polyethylene bottle. Sample bottles were rinsed three times with lake water before use. Four 1.5 m and two 5 m vertically integrated samples provided two two-litre samples.

4.2.3 Construction, installation and sampling of sediment traps

Sediment traps were constructed from a design template from University College London (Cameron, 1995), which followed the recommendations of Bloesch & Burns (1980) and Blomqvist & Håkanson (1981a). Each sediment trap was composed of three open cylindrical PVC tubes (Figure 4.2) with an aspect ratio (height : width) of 5 : 1 in order to avoid loss of collected sediment (Gardner, 1980a; Blomqvist & Kofoed, 1981b). The removable tubes were closed at the lower end with a tight cap and were fixed on a central polypropylene platform embedded with Styrofoam, giving the trap the necessary rigidity and balance. Three sediment traps were placed in the areas adjacent to the main lake in- and outflows and the deepest waters in each lake. The geographical references of the locations of each trap with the water depth are listed in Table 4.2 (see also Figure 3.2 and Figure 3.4). The locations of the three sediment traps were termed “inflow”, “deepest” and “outflow”. The traps were suspended approximately 4 m above the lake-bed. The same distance from the lake-bed was used at all sampling occasions. Each trap was anchored in the sediment using a cement block, buoyed at the surface to mark their position and another buoy was positioned one meter above each trap to keep the rope taut and the trap relatively level. It has been shown that this method works well keeping a constant distance between the trap and the lake bottom, however it necessitates the heavy anchor to be lifted frequently (Zajączkowski, 2002).

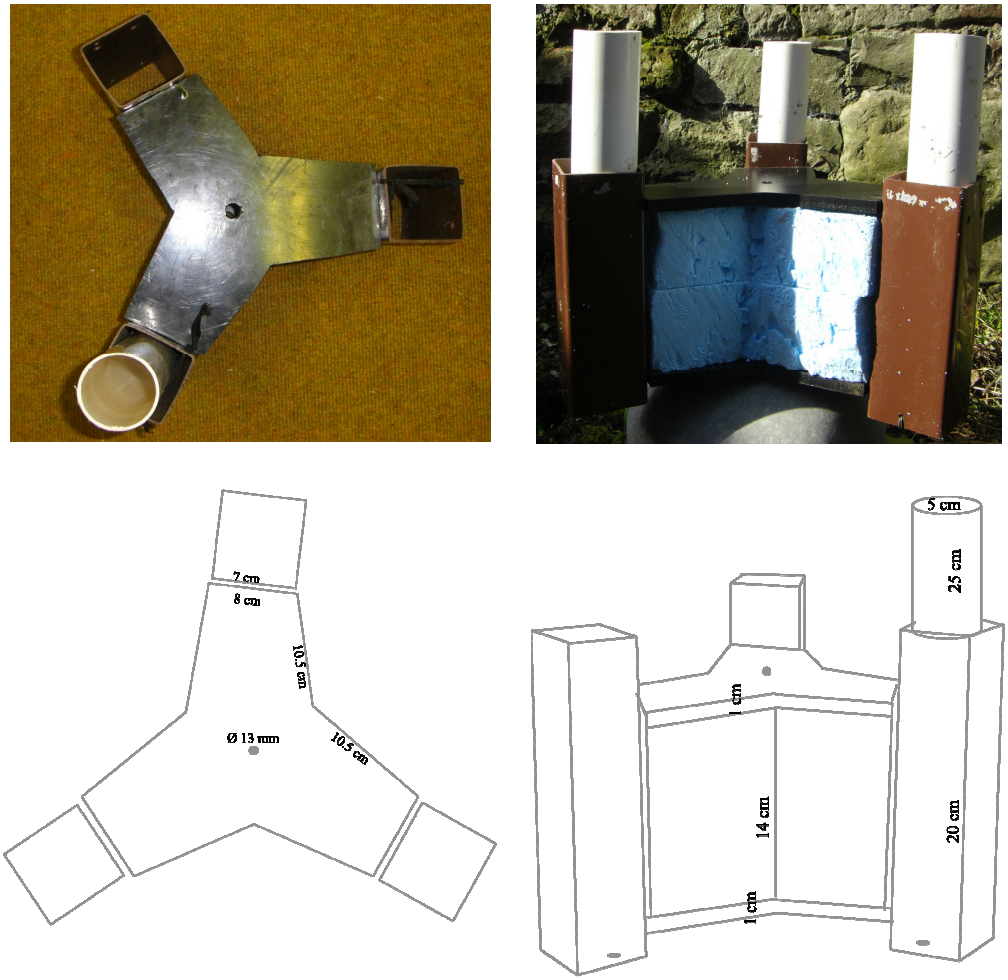

Figure 4.1 – Sediment trap top and lateral view and associated dimensions

Table 4.2- Geographical reference of the location of the sediment traps and sediment cores and water depth (m)

Station	GPS co-ordinates	Depth (m)
Feeagh		
Inflow - Trap	N 53°57'18.78'' W 9°34'54.46''	18
Inflow - Core	N 53°57'17.94'' W 9°34'57.34''	18
Deepest - Trap	N 53°56'33.69'' W 9°34'39.63''	43
Deepest - Core	N 53°56'35.39'' W 9°34'39.62''	43
Outflow - Trap	N 53°56'02.40'' W 9°34'47.00''	21
Outflow - Core	N 53°56'00.29'' W 9°34'47.02''	21
Guitane		
Inflow - Core	N 52°00'14.99'' W 9°24'50.62''	18
Deepest - Trap	N 52°00'31.38'' W 9°24'53.94''	48
Deepest - Core	N 52°00'31.38'' W 9°24'53.94''	52
Outflow - Core	N 52°00'31.38'' W 9°24'53.94''	20

The traps were positioned in Feeagh on the 1st April 2009 and in Guitane on the 26th May 2009. Time-series sediment trap samples were collected from Feeagh at approximately two-month intervals from April 2009 to July 2010 and at a seven-month interval from July 2010 to February 2011. Samples from Guitane were collected at six to eight month intervals from May 2009 to January 2011. The inflow trap in Feeagh in July 2009 and the sediment sample from the deep-water trap in Guitane in January 2010 were accidentally lost during fieldwork. The lost sediment trap in Feeagh was replaced one month later, while in Guitane the trap was re-positioned the same day. On each sampling occasion the water and sediment present in each trap tube was transferred into a single pre-washed and labelled one-litre polyethylene bottle. The sedimentation traps were re-deployed with clean tubes. The triplicate sediment trap samples collected were stored in a cool-box with ice bricks and transported to the laboratory within 1-2 days.

4.2.4 Sediment core collection and sample extrusion

In Feeagh three 40 cm sediment cores were collected adjacent to the sediment traps on the 22nd July 2010. In Guitane a 52 cm sediment core was retrieved from the deepest part of the lake on the 14th July 2010. In addition, three short sediment cores from Feeagh and one sediment core from Guitane were collected from each sampling site between January and February 2011 for pigment analysis. A distance of approximately 7 m was kept from the sediment traps to avoid collecting disturbed sediment. The sampling positions are illustrated in Figure 3.2 and in Figure 3.4 for Feeagh and Guitane, respectively, while the GPS co-ordinates of the sampling positions are listed in Table 4.2. The sediment cores were extracted using a HTH gravity corer (Teknik, Vårvågen 37, SE-95149 Luleå; (Renberg & Hansson, 2008), were sectioned at 1 cm intervals and placed in sealed plastic bags. Subsamples were collected at 2 cm intervals for pigment analysis, avoiding light contact and the inclusion of air bubbles (Reuss & Conley, 2005). Samples for pigment analysis were stored at -20°C within 5 hours. During extrusion of the core sediment characteristics were noted and precise points of any apparent variations in sediment type (Troels-Smith, 1955) or colour change (using Munsell Colour Chart) (Oyama & Takehara, 1967).

4.3 Ecological Analysis

4.3.1 Sample preparation

Water samples were pre-processed at the field-sites for laboratory chemical and biological analysis. First of all, samples for DOC analysis were filtered and acidified. Sub-samples of 100 mL were filtered using Whatman glass microfiber filter (Grade GF/F), pore size 0.45 µm) and 2-3 drops of 2 M HCl were added to remove inorganic carbon by lowering the pH of the sample to 2.0. A two-litre water sample was enclosed in a box filled with frozen ice bricks and sent within one day to the Centre of Environment, Trinity College Dublin, for chemical analysis. A second two-litre sample was used for biological analyses: a 250 mL sub-sample for phytoplankton and ciliates analysis was fixed with 1.5 mL of Lugol's iodine solution (Merck with a composition of I_2 = 3.2 g L^{-1} and Kl = 6.8 g L^{-1}) (European Union, 2009). Samples for pico- and bacterioplankton analysis were fixed with pre-filtered (0.2 µm pore size, Whatman GTTPO2500) 20% formaldehyde buffered with sodium cacodylate 0.1 M to final concentrations of 1% and 4%, respectively (Hayat, 1981) and stored in sterilized amber glass bottles (Callieri and Stockner 2002). The use of 20% formaldehyde is considered less stressful for cells (Callieri *et al.*, 2002b). The samples were kept refrigerated in the dark and were processed as soon as possible after sampling to avoid loss of cell numbers (Turley & Hughes, 1992) and to decrease problems with bleaching of autofluorescent pigments and thus prevent loss of pigment fluorescence (Olrik *et al.*, 1998; Callieri & Stockner, 2002a). The rest of the fresh (unpreserved) sample was stored at 4°C to aid in the phytoplankton identification process and examined within 2-3 days of sampling.

4.3.2 Chemical Analysis

Monthly chemical analyses were carried out at the Centre of the Environment at Trinity College, Dublin by Dr. Mark Kavanagh and under the supervision of Dr. Norman Allott. A total of ten chemical parameters were analysed and are listed in Table 4.3 together with their abbreviations, measurement units and relevant reference for method used.

Table 4.3 - Chemical parameters examined with relative abbreviations and method references

Parameter	**Abbreviation**	**Measurement unit**	**Method**
Alkalinity		mg L^{-1} $CaCO_3$	(Clesceri *et al.*, 1999)
Conductivity		µS cm^{-1}	(Clesceri *et al.*, 1999)
pH		Units	(Davison, 1990)
Colour		PtCo mg L^{-1}	(Clesceri *et al.*, 1999)
Chlorophyll-*a*	chl-*a*	µg L^{-1}	(Standing Committee of Analysts, 1983)
Dissolved organic carbon	DOC	mg L^{-1}	(Clesceri *et al.*, 1999)
Dissolved Molybdate Reactive Phosphorous	DMRP	µg L^{-1}	(Eisenreich *et al.*, 1975)
Total Phosphorous	TP	µg L^{-1}	(Eisenreich *et al.*, 1975)
Total Nitrogen	TN	µg L^{-1}	(Korolef, 1983)
Nitrate Nitrogen	NO_3-N	µg L^{-1}	(Clesceri *et al.*, 1999)

4.3.3 Biological Analysis

4.3.3.1 Sample processing

Preserved phytoplankton samples were processed following the sedimentation technique developed by Utermo hl in 1958 . The standard method was included in the WFD (EN 15204 2006) (European Standard, 2006). Before taking a sub-sample to fill the sediment chamber, the sample (previously acclimatized to room temperature) was gently mixed by overturning. As the composition and concentration of the phytoplankton in the samples was unknown, the samples were set up in different chamber sizes (25, 10 and 5 mL) simultaneously. The 25 mL chamber was adopted as it gave a good overview of the algal composition and the same settling volume was used throughout the whole series of samples from both lakes. Sedimentation chambers were filled to the top with sufficient excess to permit the water to "bead" upward. A glass cover was gently placed across the top of the chamber to remove any excess water and to enclose the exact volume of sample without entrapping any air bubbles. In order to ensure complete sedimentation of all organisms, sedimentation time in hours was at least three times the height of the sedimentation chamber (Margalef, 1969; Vollenweider, 1974).

4.3.3.2 Phytoplankton and Ciliates enumeration

Identification and enumeration of phytoplankton and ciliates was conducted under an inverted microscope (Brunel SP-95-I) at different magnifications. The microscope was

coupled with a digital camera (Leica DFC 290) and Leica Application Suite (Version 2.8.1) software was used to capture and analyse photographic images (Figure 4.2).

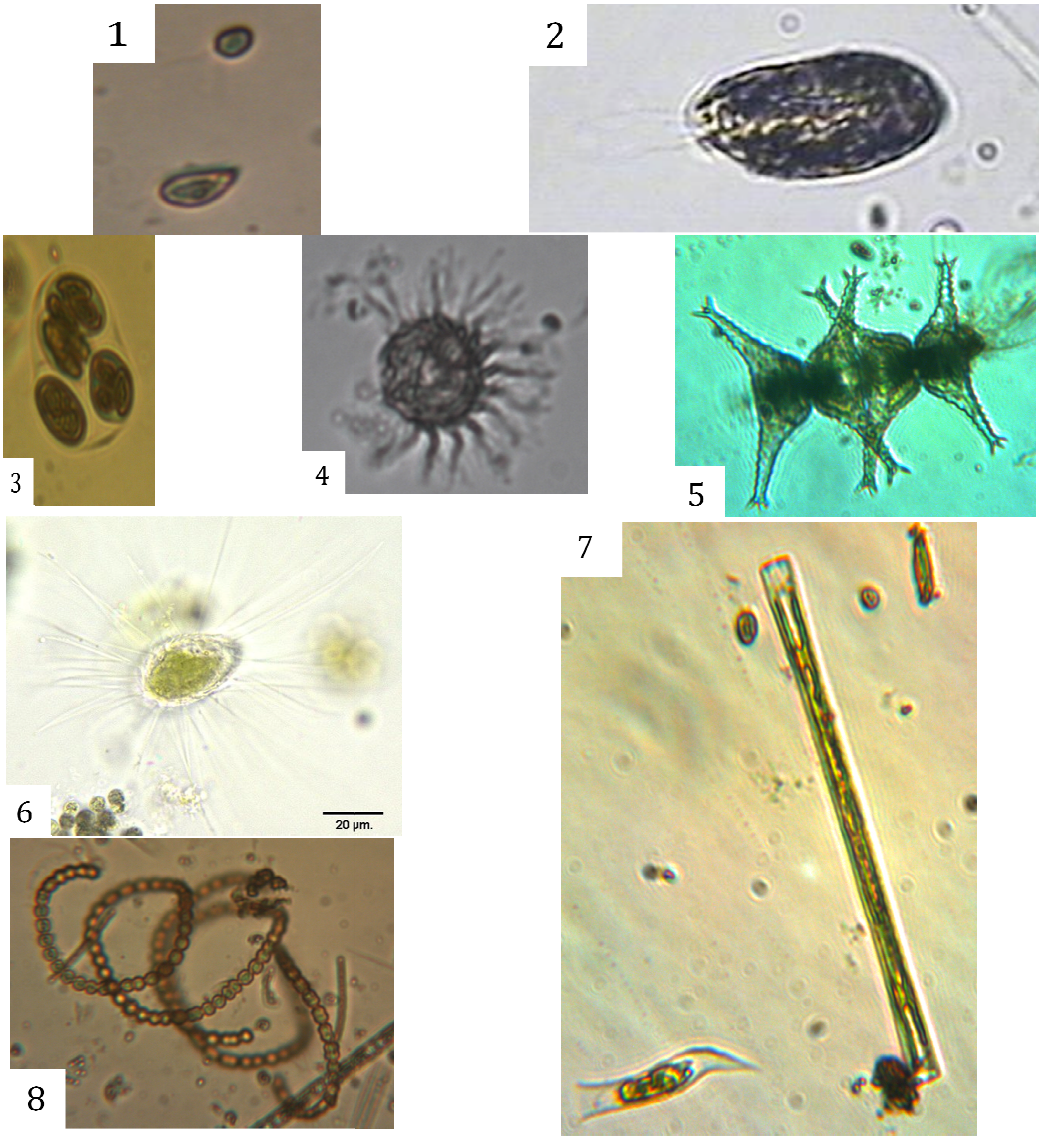

Figure 4.2 - Images of some phytoplankton taxa and Ciliates in Feeagh and Guitane: (1) *Chroomonas/Rhodomonas* minuta (top) and Chroomonas*/Rhodomonas acuta* (bottom); (2); *Cryptomonas* sp. (3) *Oocystis* sp.; (4) Ciliate; (5) *Staurastrum anatium*; (6) *Mallomonas caudata*; (7) *Tabellaria ulna* and *Dinobryon* sp.; (8) *Anabaena flos-aquae*.

Before starting the counting procedure the overall distribution pattern of phytoplankton was checked at the lowest magnification (10x). Only samples with a random (Poisson) distribution were analyzed. Only cells that appeared viable with intact chloroplasts were enumerated and estimates of cell numbers of cyanobacterial colonies were made. Filaments/trichomes and coenobia were counted individually. Empty cells (e.g. empty *Dinobryon* loricas or diatom valves) and unicellular picoplankton (< 2 µm) were not

enumerated. Enumeration was conducted as follows (Figure 4.3): the chamber was scanned at a magnification of 250x in a series of horizontal transects. All ciliates and large taxa (e.g. *Ceratium*, *Staurastrum*), large colonies and filaments (e.g. *Woronichinia*, *Fragilaria, Oscillatoria*) were counted (Figure 4.3.a). The same organisms together with the smaller colonies, *coenobia*, filaments or trichomes (e.g. *Anabaena*, *Merismopedia*, *Aphanocapsa*, *Scenedesmus*, *Crucigenia*, *Sphaerocystis*, *Asterionella*, *Aulacoseira, Djnobryon*) and larger algae (> 15 µm length) (*Cosmarium*, *Cryptomonas*) were identified and counted in the second half chamber (separated by the dashed line in Figure 4.3.a). In addition, small single algae (< 15 µm length) for example *Rhodomonas,* small centric diatoms and single cells of e.g. *Dinobryon, Monoraphidium*, *Chrysochromulina* were counted at a magnification of 400x in diagonal transects (Figure 4.3 b). Total counts of at least 360 - 440 phytoplankton units of the important species were enumerated in each sample. This number of cells corresponds to a confidence limit of 10% (Javornicky, 1958; Lund *et al.*, 1958). To facilitate the enumeration of phytoplankton cells the computer programme Opticount (Hepperle, 2005) was used.

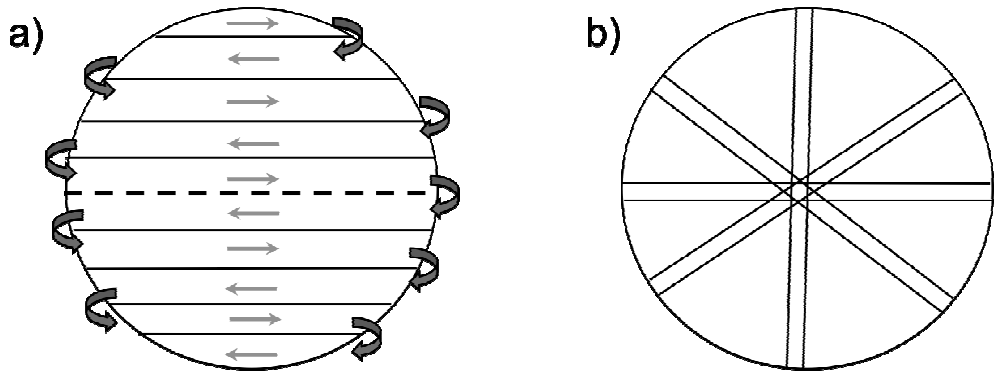

Figure 4.3 – Counting chamber enumeration methods a) horizontal (250x) and b) diagonal (400x).

Identification of taxa to genus, and when possible, to species level was achieved primarily through the use of a range of taxonomic references (Huber-Pestalozzi, 1983, 1942, 1955, 1962, 1972, 1982, 1983; John *et al.*, 2002; Wehr & Sheath, 2003). A training course in the University of Durham in advanced algal identification and taxonomy of green and blue-green algae was attended under the guidance of Prof. Brian Whitton and Dr. David John was attended. Dr. Norman Allott, Dr. Helder Pereira

(Trinity College Dublin) and Pierisa Panzani (Institute of Ecosystem Study, Verbania Pallanza, Italy) aided taxonomical classification. Some taxa were not discriminated beyond general groupings, such as small centric diatoms (considered to be *Cyclotella* spp.) and all pennate diatoms smaller than 15 µm were combined onto one group, and thus represent an understimation of Bacillariophyta species. A common small Cryptophyta with a typical pointed apex was named *Chroomonas/Rhodomonas acuta* (Leitao & Leglize, 2000; Palsson & Graneli, 2004) as it was morphologically similar to *Chroomonas acuta*, but also to *Rhodomonas minuta*/*Plagioselmis nanoplanktonica* (Novarino *et al.*, 1994; Novarino, 2002). *Chroomonas/Rhodomonas minuta* was distinguished by its round apex (Barone & Naselli-Flores, 2003; Javornickŷ, 2003). A further unidentified algae, was a round single cell (with a diameter of 4-5 µm) characterized by the absence of flagella, which could be derived from broken colonies of Chlorophyta. For this study, these cells were enumerated separately as unicellular autotrophs. Unidentifiable broken filaments were present in samples collected over the summer months. The characterisation of auto- and mixotrophic species was carried out according to Tranvik (1989), Lewitus (1994), Jansson *et al.* (1996), Isaksson *et al.* (1999), Geider & MacIntyre (2002). Typically mixotrophic species (Dinoflagellata and certain Chrysophyta (*Chromulina, Chrysococcus*, *Dinobryon, Ochromonas* and *Pseudopedinella*) and potentially mixotrophic taxa (Chlorococcales, *Cryptomonas* and *Chroomonas*/*Rhodomonas)* were put into one group and considered as "potentially mixotrophs".

4.3.3.2.1 Conversion of counting numbers to cell density

Calculation of cell density (cells mL^{-1}) was achieved by dividing the number of algal units (coenobia, colonies, filaments etc.) encountered in the chamber by the sample volume. Cells enumerated in the half chamber were multiplied by two. For the smaller cells (< 15 µm) the calculation required knowledge of the area of the chamber bottom (i.e. 500 mm^2 corresponds to 2599.5 optical fields at a magnification of 400x), the area of the part of the chamber bottom that has been counted (e.g. 0.19 mm^2 x the number of optical fields - 50 in one transect) and finally the number of cells counted for each species. The number of algal cells counted was then converted to give a concentration per unit volume of sample according to:

$$N = X \frac{A}{a \times v}$$

where N is the number per unit volume, X is the number of counted cells, A is the total effective area of the chamber, a is the number of the counting fields and v is the volume of the sample in the chamber. The unit of measurement was algal cells mL^{-1}.

4.3.3.2.2 Estimation of biomass

Detailed analysis of phytoplankton populations requires not only the estimation of cell density, but also algal biomass. Cell numbers do not provide a representative measure because of the considerable variation in cell size among algal species (Smayda, 1978; Wetzel & Likens, 2000). A standard biomass estimate is essential for comparing the relative contribution of different algae between samples and aquatic systems (Potapova & Snoeijs, 1997; Hillebrand *et al.*, 1999). Algal biomass was calculated by multiplying the number of cells of a given species counted in a sample by its average cell volume. Total sample/community biomass was obtained by summing the biomasses of the individual species. Cell dimensions of a species can vary greatly in size between different seasons or geographical location. For this reason, cell volume of each important species was determined for each sample (Wetzel & Likens, 2000). The calculation of biovolume of algae and ciliates was based on geometric approximations. The biovolume of the dominant species were calculated according to 20 different geometric shapes and respective equations taken from the literature (Willén, 1976; Smayda, 1978; Rott, 1981; Hillebrand *et al.*, 1999; Pohlmann & Friedrich, 2001; Sun & Liu, 2003; Vadrucci *et al.*, 2007). The procedure involved the collection of digital photographs (*Leica* DFC 290) and the direct measurement of the linear dimensions (length, width and height) required for calculating the associated geometric cell volumes with a computerized image analysis system program (Leica Application Suite Version 2.8.1). The estimated average biovolume (μm^3 $cell^{-1}$) was compared with literature-based studies from the UK (e.g. Carvalho *et al.*, 2007) and other international publications (Willén, 1976; Makarewicz, 1993; Pohlmann & Friedrich, 2001; Brettum, 2002; Kasten, 2003; Kamenir & Morabito, 2009). The algal biomass for each species was calculated as follows:

$$\text{Algal biomass } (mm^3\ m^{-3}) = \text{density } (\text{cell } mL^{-1}) \times \text{cellular mean biovolume } (\mu m^3\ \text{cell}^{-1}) \times 10^{-3}$$

4.3.3.3 Heterotrophic bacterioplankton and autotrophic picoplankton

4.3.3.3.1 Sample filtration

Formaldehyde fixed open water samples were processed in the laboratory following the method described by Daley & Hobbie (1975), Porter & Feig (1980), Caron (1983), Sherr *et al.* (1993), MacIsaac & Stockner (1993) and Kemp *et al.* (1993). The procedure was similar for bacterio- and picoplankton samples. A wetted white polycarbonate filter (Millipore, Ireland, type HAWPO2500) was placed on the filtering device to support the membrane filter in order to facilitate even distribution of the sample. Subsamples of 1 and 5 mL were filtered onto 0.2 µm pore-sized black isopore membrane filters (Millipore, Ireland, type GTBP 2500) and in semi-darkness 0.1 and 0.5 mL of 0.1 µg mL^{-1} 4'6'-diamidino-2-phenylindole (DAPI) were added. The whole sample was drawn through the filter with a vacuum pump under low pressure (5-10 kPa) (Kuuppo-Leinikki & Kuosa, 1989; MacIsaac & Stockner, 1993). For the picoplankton, two 5 mL subsamples underwent the same procedure without the addition of DAPI. The filters were dried after removal from the holder and mounted on glass slides directly on a small drop of 50% glycerol-water solution (Callieri & Stockner, 2002a). An additional drop of glycerol was then added followed by a round cover slip. Finally, the slide was pressed with caution on paper to absorb the excess of glycerol. The slides were stored at -20°C to minimize bleaching of the autofluorescent pigments (MacIsaac & Stockner, 1993).

4.3.3.3.2 Identification and cell enumeration

The epifluorescence microscopy technique was applied to quantify the abundance and biovolume of heterotrophic bacteria and phototrophic picoplankton. All samples were enumerated on two separate occasions (in December 2009 and August 2010) at the CNR-ISE Institute of Ecosystem Study, Verbania-Pallanza, Italy, under the supervision of Dr. Cristiana Callieri (Figure 4.4). The fluorescent cells caught on the filter were counted under an epifluorescence microscope (ZEISS Axioplan) equipped with objectives specially designed for fluorescence with immersion oil and various filter/dichroic-mirror sets, using a total magnification of 1250x. Both bacteria and picoplankton were encountered using the same methodology with the only difference that for the former a UV filter (G365, FT395, LP420) was used, while the latter were examined using filters for blue (BP450-490, FT510, LP520) and green light excitation

(LP510-KP560, FT580, LP590). The fluorescent cells caught on the filters were enumerated by random fields at the highest magnification (1250x).

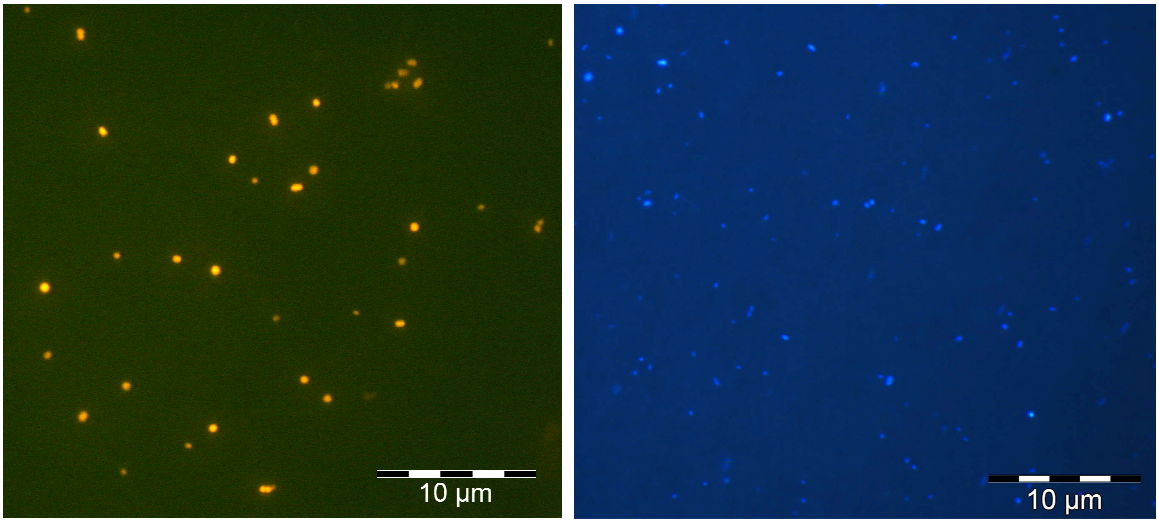

Figure 4.4 – Images of autotrophic picoplankton (to the left) and heterotrophic bacterioplankton (to the right).

At least 400 cells were counted with an upper limit set at 30 random microscope fields to obtain a precision of 10% (Lund *et al.*, 1958). The heterotrophic bacteria appeared bright blue in colour against a dark background, while other particulate matter fluoresced in weak yellow and could therefore easily be distinguished (Porter & Feig, 1980). Solitary cells, loose aggregates and small colonies (< 2 µm) were all considered to be autotrophic picoplankton (picocyanobacteria and picoeukaryotes), while single-celled rod shaped Cyanobacteria and picoeukaryotes with a diameter of 0.8-1.2 µm and a cell length of > 2 µm were not included as they were already counted as phytoplankton in the sedimentation chambers hl, 1958). Generally, yellow-orange picoeucariots can be distinguished from the red picocyanobacteria (Stockner & Antia, 1986), however no clear distinction could be made for the study samples. Both appeared in orange and were therefore counted as one single group.

4.3.3.3.3 Conversion of counting numbers to cell density

The following formula was applied to calculate algal cell densities (cells mL^{-1}):

$$Density = F \times \frac{N}{\text{ml of sample} \times 0.95}$$

€

where *F* is a conversion factor which is calculated from the ratio of active filter area and area of field countered, which is 20,259.0 *N* is the mean number of cells per field and 0.95 to account for the sample : formaldehyde ratio.

4.3.3.3.4 Estimation of biomass

Pico- and bacterioplankton cell size measurements were made for each sample. Digital images of fields with enough bacteria and absence of very bright and yellow particulate matter or particles were selected and digitized with a camera (Olympus DP 72) attached to the epifluorescence microscope at a magnification of 785x. The original and unmodified pictures were saved (Cell^B Version 3.2) and an automated image analysis system (Image Pro Plus Version 4.5.1) was used to assess cell volumes (length, width, volume) of at least 100 cells according to the algorithms given in Massana *et al.* (1997). The software allowed manual selection of individual cells, which were characterized by their colour and light intensity. The undesired objects were removed by comparing the binary image with the original one. The total pico- and bacterioplankton biomass was calculated from the average biovolume measure of each sample using the same equation used with phytoplankton:

$$\text{Pico- and bacterioplankton biomass } (mm^3\ m^{-3}) = \text{total cell density } (cells\ mL^{-1}) \times \text{average cellular biovolume } (\mu m^3\ cell^{-1}) \times 10^{-3}$$

4.4 Sediment core and trap analyses

4.4.1 Sample preparation

The contents of two of the three sediment trap samples were used for Loss On Ignition (LOI), TOC, TN and diatom analyses, while the third sample was kept in complete darkness and stored at -20°C for pigment analyses. Table 4.4 shows the sediment trap sampling dates, months and days of deployment and number of samples (n) of each proxy analyzed together with the number of samples analyzed in each sediment core. For the sediment trap samples sediment deposition rate and LOI (%) were measured for 23 months in Feeagh and for 21 months in Guitane, while organic matter content (TOC and TN (%)) and diatoms were examined over 16 months in Feeagh and 15 months in Guitane. The samples for pigments concentrations were analysed for 9 months in Feeagh (November 2009 – July 2010) and 15 months in Guitane (May 2009 – July 2010). In Feeagh, LOI was analysed at 2 cm intervals in the in- and outflow sediment,

cores and at every centimetre in the deepest core (20 and 40 samples, respectively). TOC and TN measurements were made on 10 samples in each core of both lakes, while pigments were analyzed at 2-cm intervals. The diatom assemblage was enumerated exclusively in the surface sediment. A detailed sediment core diatom reconstruction was available from Dalton *et al.* (2010). In Guitane LOI was estimated at centimetre intervals, while TOC and TN were measured at 2 cm intervals from the surface to 10 cm depth and on a 7-10 cm interval for the rest of the core. Fossil pigments were analysed every 2 cm. Diatom assemblages were enumerated in a total of 10 samples were with higher resolution for the first 10 cm.

Table 4.4 – List of sediment trap sampling dates and their deployment intervals in terms of total months and days and their number of samples (n) analysed in both lakes for each trap. The number (n) of samples analyzed in each sediment core is also given.

	Feeagh				**Guitane**			
	Traps			**Core**	**Traps**			**Core**
	Sampling period	*Months / Days*	*n*	*n*	*Sampling period*	*Months / Days*	*n*	*n*
Sediment deposition	01/04/2009 – 08/02/2011	23 / 679	9	-	19/05/2009 – 19/01/2011	21 / 611	3	-
LOI	01/04/2009 – 08/02/2011	23 / 679	9	20 - 40	19/05/2009 – 19/01/2011	21 / 611	3	52
TOC, TN	01/04/2009 – 22/07/2010	16 / 478	8	10	19/05/2009 – 14/07/2010	15 / 422	2	10
Pigments	20/11/2009 – 22/07/2010	9 / 245	3	20	19/05/2009 – 14/07/2010	15 / 422	2	27
Diatoms	01/04/2009 – 14/07/2010	15.5 / 470	8	1	19/05/2009 – 14/07/2011	15 / 422	2	10

4.4.2 Sediment trap deposition

The sediment trap samples were allowed to settle for four days and then the overlying water was siphoned off using a wide-bore pipette. The samples were transferred into pre-weighed Whirl-Pak bags and dried in an oven at 30°C. The dry weight (DW), expressed in g of DW, was subsequently used to calculate the sediment deposition rate. The daily sinking sediment deposition (or flux) was calculated by dividing the DW by the number of days the trap was deployed *in situ* in the lake. The sediment deposition rate was obtained using the following equation:

$$g \text{ of sediment / day} : 1963.49 \text{ mm}^2 = x : 1 \text{ mm}^2$$

where 1963.49 mm^2 corresponds to the collecting tube area (π x 0.25 mm^2 = 1963.49 mm^2). This was then converted to DW g m^{-2} d^{-1}. Finally, the total sediment deposition was calculating by summing the different samples of dry sediment collected and converted to an area of one square meter:

$$Total = \sum \frac{d}{f}$$

where *Total* is the total sediment deposition, *d* the dry sediment weight and *f* the area of the collecting tube.

4.4.3 Sediment chronologies: ^{210}Pb and artificial radionuclides

One of the most important means for dating of recent sediments (0-150 years) is the natural radioactive isotope of lead (^{210}Pb) (half-life of 22.3 years) and artificially produced radionuclides caesium (^{137}Cs) (half-life of 30.2 years) and americium (^{241}Am) (half-life of 432.2 years) (Appleby, 2001). The former is derived from natural atmospheric fallout, while the latter two were emitted during nuclear weapons testing and nuclear reactor accidents. In particular, two distinctive peaks can be detected in sediment cores: a first peak is linked with the atmospheric weapons tests between 1953-63 and a second peak is associated with the Chernobyl reactor fire in April 1986. Both are extensively used in dating of recent sediments (Appleby, 2001).

A sediment chronology, using ^{210}Pb and ^{137}Cs, was established for Feeagh in the ILLUMINATE project (Dalton *et al.*, 2010). The sediment core collected in this project was correlated with this dated core by matching points on LOI stratigraphies visually, plotting them and adding a trend line (linear regression). This enabled a match between depth *x* in the ILLUMINATE core with depth *y* in the sediment core collected for this project. For cost reasons, no chronologies were established for the two littoral sediment cores collected adjacent to the in- and outflow sediment traps during this project. Results for those two cores were therefore reported according to depth only.

The sediment core extracted from the deepest point of Guitane was analysed for short-life radionuclides ^{210}Pb, ^{137}Cs and ^{241}Am at the Bloomsbury Environmental Isotope Facility (BEIF) at University College London under the supervision of Dr. Handong Yang. Wet sediment core samples (circa 2 g) were evenly picked from the top to the

bottom of the sediment core and oven dried at 50°C for 24 hours. The dried samples (circa 0.5 g) were ground using a mortar and pestle and transferred into labelled Whirl-Pak bags for transport. The samples were analysed by direct gamma assay using an ORTEC HPGe GWL series well-type coaxial low background intrinsic germanium detector. ^{210}Pb was determined via its gamma emissions at 46.5 keV, and ^{226}Ra by the 295 keV and 352 keV gamma rays emitted by its daughter isotope ^{214}Pb following 3 weeks storage in sealed containers to allow radioactive equilibration. ^{137}Cs and ^{241}Am were measured by their emissions at 662 keV and 59.5 keV, respectively (Appleby *et al.*, 1986).

The sedimentation accumulation rate (SAR) was calculated by the unsupported ^{210}Pb and was expressed both as g cm^{-2} y^{-1} and cm yr^{-1}. Dates were determined using the Constant Initial Concentration (CIC) and Constant Rate Supply (CRS) model (Krishnaswamy *et al.*, 1971; Appleby & Oldfield, 1978). The former model provides good results when uniform rate of sediment accumulation (and consequently of ^{210}Pb) occurred (Appleby & Oldfield, 1978). This model assumes that the unsupported ^{210}Pb accumulated on the lake bottom remains unaffected by post-depositional processes and decays exponentially with time. The second model is used when variations in SAR were detected (Appleby, 2001). Therefore, in this case the dates of the older sediments are calculated from the distribution of ^{210}Pb throughout the sediment core (Appleby, 2001). Radiometric chronology of the sediment core taken from Guitane was applied using the CRS model.

4.4.4 Lithology

A preliminary visual inspection enabled broad variations in lithological composition (e.g. presence of sandy layers, changes in colour, presence of macrofossils) to be described. Wet density, dry weight and LOI measurements were conducted on all cores and their measurements were made using standard techniques (Bengtsson & Enell, 1986; Boyle, 2001). In addition, sediment trap samples were measured for dry weight and LOI. A basic classification based on any apparent variations in sediment type (Troels-Smith, 1955) or colour change (using Munsell Colour Chart) was noted in field during the extrusion of the sediment.

4.4.3.1 Wet density

Wet density (g cm^{-3}) reflects changes in sediment composition. Wet density values were necessary to establish sediment accumulation rates from ^{210}Pb analysis. Sediment wet density was determined using a 2 cm^3 capacity brass phial. The phial was completely filled with sediment, paying attention to exclude air spaces. Density values were divided by two and expressed as g/cm^3. To prevent samples cross-contamination the phial was washed with de-ionized water and dried before measuring the next sample.

4.4.3.2 Dry weight

Dry weight (DW) represents sediment water content. Weighted sediment samples were oven dried at 105°C for 12 hours. After a cooling period in a desiccator with silica gel (Merck DIN 55474) samples were re-weighed. Dry weight percentage values were calculated as follows:

$$DW\,(\%) = \left(\frac{DW_{105}}{WW} \right) \times 100$$

where DW_{105} is the weight after oven drying and WW is the wet weight.

4.4.3.3 Loss On Ignition

The Loss On Ignition (LOI) method was applied to determine variations in organic matter content in trap and sediment samples (Dean, 1974; Heiri *et al.*, 2001). The previously dried sediment for dry weight analysis was placed in a muffle-furnace and fired at 550°C for a period of four hours. Ample cooling was required and samples were re-weighed to calculate the percentage of organic matter content lost using the following equation:

$$LOI_{550} = \left\lfloor \frac{\left(DW_{105} - DW_{550}\right)}{DW_{105}} \right\rfloor \times 100$$

where DW_{105} and DW_{550} are dry weight after 105°C and dry weight after 550°C respectively.

4.4.5 Geochemistry

4.4.5.1 Total organic carbon and total nitrogen

Where samples contain inorganic carbon and the organic carbon content of a sample is to be measured using elemental analysis, samples must be pre-treated to remove inorganic carbon. This procedure from sediment trap and sediment core samples followed the vapour acidification method proposed by Harris (2001) and Bianchi (1997). Approximately 30-40 mg of dried sediment was placed together in a beaker filled with circa 150 mL of concentrated HCl (37%) in a desiccator in a fume cup-board. The fumes decomposed any $CaCO_3$ present in the samples (Bianchi *et al.*, 1997). After four hours the samples were dried in an oven at 60°C for six hours. The samples were removed from the oven and reweighed. The weight loss represented the inorganic carbon content of the original dry samples.

The analyses of a total of 29 sediment trap and 40 sediment core samples from both lakes was carried out at the Institute of Technology in Dundalk under the supervision of Dr. Eleanor Jennings. Approximately 5 mg of treated sample was transferred into small silver cylinders, compressed using tweezers and placed on the numbered carousel of the CHNS-O Elementar Analyzer (vario El cube). The instrument combusted each subsample at a high temperature (850°C to 1100°C) in an oxidizing atmosphere and then separated the gaseous products by chromatography (Verardo *et al.*, 1990). Known amounts of standards of sulfanilamide were included at the beginning of each run and after every eight samples. A computer reads the element concentration from the detector signal, and the sample weight on the basis of stored calibration curves. Elemental weight percentage composition of TOC and TN was used to calculate the C/N ratio.

4.4.6 Biological remains

4.4.6.1 Pigments analysis

Even though, a pigment profile from the deepest part of Feeagh was already available in Dalton *el al.* (2010), a second investigation permitted a more detailed analysis of temporal palaeoecological variability and historical catchment change. In Dalton *el al.* (2010) the pigment analysis determined only a small selection of pigments (chl-*a*, chl-*b*, pheophytin-*a*, lutein, diato- and zeaxanthin) without including pigments present in

Cryptophyta (alloxanthin), siliceous algae (fucoxanthin) and Cyanobacteria (e.g. echinenone, cantha- and myxoxanthin). In addition, a modification of the methodology involved the extraction of pigments from defrosted samples in organic solvents with sonication and grinding. While studies highlight that freeze-drying improve pigment extraction (Buffan-Dubau & Carman, 2000), no single method is optimal for all pigments or all sediment types (Buffan-Dubau & Carman, 2000; Reuss & Conley, 2005).

The algal pigment concentrations of sediment trap and sediment core samples were determined in the laboratories of University of Nottingham using High Pressure Liquid Chromatography (HPLC) unit under the supervision of Dr. Suzanne McGowan. The samples were freeze-dried just before extraction and analysis of the pigments. The standardized analysis was carried out in semi-darkness to avoid any degradation. Samples were extracted overnight at -4°C in a mixture of acetone, methanol and deionised water (80 : 15 : 5) following Leavitt & Hodgson (2001a). Extracts were filtered with 0.22 µm PTFE syringe filters, dried completely under nitrogen gas and re-dissolved in a 70 : 25 : 5 mixture of acetone, ion pairing reagent (IPR 0.75 g tetrabutyl ammonium acetate and 7.7 g ammonium acetate in 100 mL water) and methanol before injection into the HPLC system comprised of an Agilent 1200 series quaternary pump, autosampler, ODS Hypersil column (250 x 4.6 mm; 5 µm particle size), Waters 996 photo-diode array detector and Waters Millenium Chromatography Manager Software. Separation conditions were modified from Wright *et al.* (1991). Each sample was injected with 100 µL of solvent and Chen's *et al.* (2001) gradient program was applied. Peak areas were calibrated using commercial pigment standards (DHI, Denmark) and Agilent ChemStation software generated chromatogram reports for each sample. In a total of 100 chromatograms between 22 and 42 peaks were identified, of which 17 peaks were included in the final analysis and interpretation. The remainder was either not successfully resolved by HPLC analyses or did not appear at the right retention time and were therefore considered unidentifiable pigments. Concentrations were reported in nanomoles of pigment relative to the organic material in the dry sediment (nmol g^{-1}) as estimated by LOI at 550°C. Ratios of labile : stable pigments (chlorophyll-*a* : pheophytin-*a*) were used to identify the degree of pigment preservation in each sample (Patoine & Leavitt, 2006; Reuss *et al.*, 2010; McGowan *et al.*, 2011). High ratios indicate good preservation and are often observed when algal production increases

(Leavitt *et al.*, 1997). The UVR-index was calculated as a measure of water clarity by dividing the concentration of UVR-absorbing compound relative to the sum of four abundant and stable carotenoids (alloxanthin, diatoxanthin and lutein/zeaxanthin) and multiplying by 100 (Leavitt *et al.*, 1997; McGowan *et al.*, 2011).

4.4.6.2 Diatom analysis

The preparation of diatom microscope slides from sediment trap and sediment core samples followed the methodology proposed by Battarbee *et al.* (2001). A known quantity of sediment was placed in 12 mL plastic centrifuge test tubes to which 5 mL of H_2O_2 (30% v/v) was added. Digestion of samples was achieved in a water-bath at 60°C until oxidation was complete (Blanco *et al.*, 2007). The volume of the suspension was regularly controlled to avoid complete desiccation. After digestion 1-2 drops of 10% (v/v) HCl were added to eliminate any remaining H_2O_2 and any carbonates. Afterwards the centrifuge test tubes were topped up and washed with deionised water and were put in a centrifuge for 6 minutes at 600 rpm. The supernatant liquid was decanted and the washing procedure was repeated four times. Samples were stored in glass vials and few drops of NH_3 were added to prevent frustule clumping.

To prepare diatom slides a small amount of sample solution was diluted with deionised water and transferred onto microscope slide cover slips. Two different concentrations were prepared for each slide to facilitate enumeration. Samples were dried at room temperature for 1-2 days. A drop of mounting medium (Naphrax) was put on a glass slide on a hotplate at c. 80°C and the inverted cover slip with the completely dried diatoms was placed over the drop. The slide was heated on the hotplate to evaporate the toluene in the Naphrax. The slide was then allowed to cool.

Diatom concentration was determined following the microsphere (divenylbenzene) addition method proposed by Battarbee & Kneen (1982) and Battarbee *et al.* (2001). Using a micropipette 100 µL of 6.23 x 10^6 microspheres mL^{-1} suspension was added to the previously prepared digested samples. Microspheres were counted separately during diatoms counts and frustule concentration was then obtained using the following equation (Battarbee *et al.*, 2001):

$$Frustule.concentration = \frac{\text{Microspheres introduced} \times \text{diatoms counted}}{\text{Microspheres counted}}$$

Frustule concentration was expressed as frustule per gram of dry (trap) and wet (core) sediment.

Mean daily accumulation rate per sampling period was calculated from the calculated frustule concentration in the sediment trap samples divided by the number of days of exposure, in order to take into account the different time periods between sampling dates.

Diatom valve identification and enumeration was achieved using a Leica DME microscope with an oil immersion objective at 1000x magnification (Figure 4.5). The microscope was coupled with a Leica camera (DFC 290) and Leica Application Suite (Version 2.8.1) software to capture and manage photographic images. In order to ensure representative samples, a minimum of 400 valves per sample were enumerated for each sample in horizontal transects. Single valves were used as the basic counting unit. Furthermore, diatom fragments were counted based on a system of recognisable ends for certain species (e.g. *Eunotia incisa*) and central areas of others (e.g. *Tabellaria flocculosa*). Abundances were expressed as percentages of the total diatom count. Taxonomic identification and nomenclature was achieved according to Krammer & Lange-Bertalot (1986; 1988; 1991a; 1991b), Lange-Bertalot (1996), Kelly *et al.* (2005), Houk *et al.* (2010) and Guiry (2007). Moreover, the subdivision of the community into benthic, planktonic and tychoplanktonic taxa was achieved using a number of published sources (Tuchman, 1996; Gibson *et al.*, 2003; Kelly *et al.*, 2005; Jones, 2007; Podaner & Potapova, 2007). This was supplemented by two diatom workshops held by Dr. Nadia Solovieva and Dr. Manel Leira at Trinity College Dublin. Taxonomy identification of *Cyclotella* spp. was confirmed by Prof. John Anderson (Loughborough University, UK). Dr. Barry O'Dwyer (Trinity College Dublin) aided taxonomical classification of *Aulacoseira* spp. Species counts were transformed into percentage abundances and taxa with relative abundance > 1% in at least two samples through the core formed the working datas

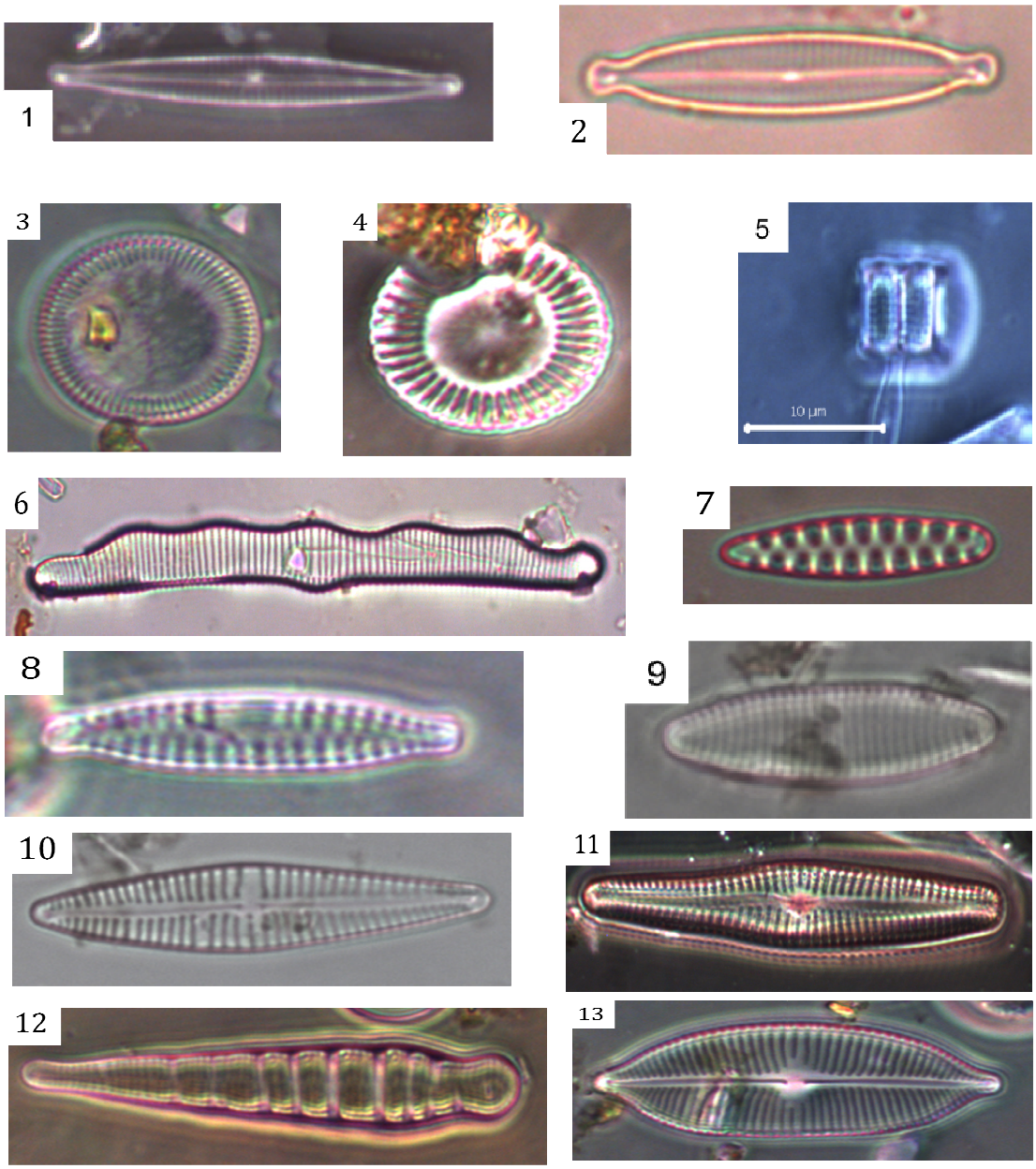

Figure 4.5 - Images of some diatom taxa in Feeagh and Guitane: (1) *Achnanthidium minutissimum*; (2) *Cymbella microcephala*; (3) *Cyclotella kuetzingiana*; (4) *Cyclotella stelligera*; (5) *Aulacoseira alpigena* (girdle view); (6) *Eunotia hexaglypha*; (7) *Fragilaria leptostauron* var. *martyi*; (8) *Fragilaria capucina* var. *vaucheriae*; (9) *Fragilaria exigua*; (10) *Gomphonema angustum*; (11) *Gomphonema clavatum*; (12) *Meridion circulare*; (13) *Navicula placentula*.

4.5 Data analyses

4.5.1 Exploration of environmental and biological data

The analysis of environmental and biological data was performed on the monthly measurements over the annual cycle (April 2009 – May 2010). Frequency histograms were plotted for each measured environmental variable in Data-Desk (Version 6.1). The

plots enabled decisions on whether log_{10} or no data transformation was best (Ebdon, 1977). Several variables showed a skewed distribution and required transformation. TP, DMRP, TN, NO_3-N, chl-*a* and DOC were log_{10}-transformed. Correlation between environmental variables was examined using Spearman's rank correlations in Sigmaplot (Version 11.0). Highly correlated environmental variables ($r \leq -0.618$ or $r \geq 0.618$) were excluded from further analysis.

A series of multivariate statistical analysis were applied to the physical-chemical and biological data using Canoco (for Windows 4.5) and Primer-E 5 (Version 5.2.6 for Windows) software (Plymouth Routines in Multivariate Ecological Research, Plymouth Marine Laboratory, Plymouth, UK) (Clarke & Ainsworth, 1993; Clarke & Gorley, 2001a). First, the biological variables (different algal groups, picoplankton, bacterioplankton and ciliates) were root square transformed. Ordination Analysis and multidimensional scaling (MDS) were applied. Further, patterns in community structure identified by MDS analyses were linked to environmental variables (based on Euclidean distance similarity index) by using the BIOENV method. The procedure calculates a measure of agreement between the two similarity matrices by Spearman correlation, which ranks the subsets of variables that best 'matches' the biological patterns (Clarke & Ainsworth, 1993; Clarke & Gorley, 2001a). Limitations of Primer-E 5 and the unavailability of access to the newer version of Primer (Version 6) prevented determination of significance. This facility was available in ordination analysis, which was adopted as the preferred multivariate data analysis technique.

4.5.1.2 Ordination Analysis

Relationships between biological and environmental variables were assessed using direct gradient analysis. First, a Detrended Correspondence Analysis (DCA) of the biological variables was run to determine whether linear or unimodal ordination methods should be applied (ter Braak & Šmilauer, 2002). Because the length of the first axis resulting from the DCA was less than three (0.905 Std. dev.), a linear method (redundancy analysis or RDA) was applied (ter Braak & Šmilauer, 2002). Significant explanatory variables were determined by automatic forward selection. Monthly samples over one annual cycle (April 2009 – May 2010) from both lakes were used for this analysis.

4.5.2 Palaeolimnological data

Stratigraphic plots for each lithological, geochemical and biological proxy were created using C^2 software (version 1.3) (Juggins, 2003). Stratigraphically Constrained Incremental Sum of Squares cluster analysis (CONISS) using Euclidean distances was used to reveal the timing of major changes in the sediment pigments and diatom assemblages from Guitane. Pigment and diatom abundance (in %) data were input in PSIMPOLL 4.27 software (Bennett & Willis, 2002) and only significant zone boundaries were selected. The statistical significance of the zone boundaries was tested using the broken-stick model (Bennett, 1996).

Chapter 6 - Spatial and temporal changes in sediment deposition

6.1. Introduction

The installation of three sediment traps (inflow, deepest and outflow) in Feeagh and Guitane permitted the collection of authochtonous and allochthonous matter falling through the water column and enabled the calculation of daily and total sediment deposition rates. Trap samples were examined for lithology, geochemistry and biological characteristics. Surface sediments from adjacent sediment cores were also analysed for the same parameters. Measurements of water column chl-*a* and contemporary diatom assemblages (detailed in Chapter 5) were compared with the trap and surface sediments. The term "flux" is used in this chapter and relates to the deposition of sediment, diatom valves and algal pigments collected in each sediment trap.

6.2 Sediment deposition

The rates of sediment deposition were estimated on a daily basis (g m^{-2} d^{-1}) and are depicted separately for Feeagh (Figure 6.1 a) and Guitane (Figure 6.1 b) (Appendix J). In Feeagh the daily sediment deposition rate was calculated over 23 months and 9 sample periods from 1st April 2009 to 8th February 2010 (see Table 4.3) and ranged from 0.6 to 7.93 g m^{-2} d^{-1}, with means of 4.1 g m^{-2} d^{-1} (inflow), 3.8 g m^{-2} d^{-1} (deepest) and 2.6 g m^{-2} d^{-1} (outflow). The deposition rates were clearly higher at the inflow compared to the deepest and outflow traps on seven of the nine occasions sampled. The highest sediment deposition was measured between May and July 2009, with estimated deposition of 7.9 g m^{-2} d^{-1} (inflow), 6.9 g m^{-2} d^{-1} (deepest) and 5 g m^{-2} d^{-1} (outflow). The second highest rate was measured between December 2009 and January 2010, with 6.1 g m^{-2} d^{-1} (inflow), 5.4 g m^{-2} d^{-1} (deepest) and 4.4 g m^{-2} d^{-1} (outflow), whilst the lowest was between January and March 2010, with 1.6 g m^{-2} d^{-1} (inflow), 0.8 g m^{-2} d^{-1} (deepest), and 0.6 g m^{-2} d^{-1} (outflow).

The sediment trap deposition rates were lower in Guitane relative to Feeagh and only minor differences were evident between the three collecting stations (Figure 6.1 b). The deposition rate was calculated over 21 months and three sampling periods between 10th May 2009 and 19th January 2011. The fluxes ranged from 0.3 to 1.5 g m^{-2} d^{-1} with means of 0.9 g m^{-2} d^{-1} (inflow), 0.5 g m^{-2} d^{-1} (deepest) and 0.8 g m^{-2} d^{-1} (outflow). The highest sediment deposition was measured between May 2009 and January 2010 with 1.5 g m^{-2} d^{-1} of sediment at the inflow and outflow traps. No data are available for the deepwater trap as the sample was lost on retrieval on the 25th January 2010. The lowest sediment load was accumulated between January and July 2010 with 0.4 g m^{-2} d^{-1} at the inflow and 0.3 g m^{-2} d^{-1} at the outflow.

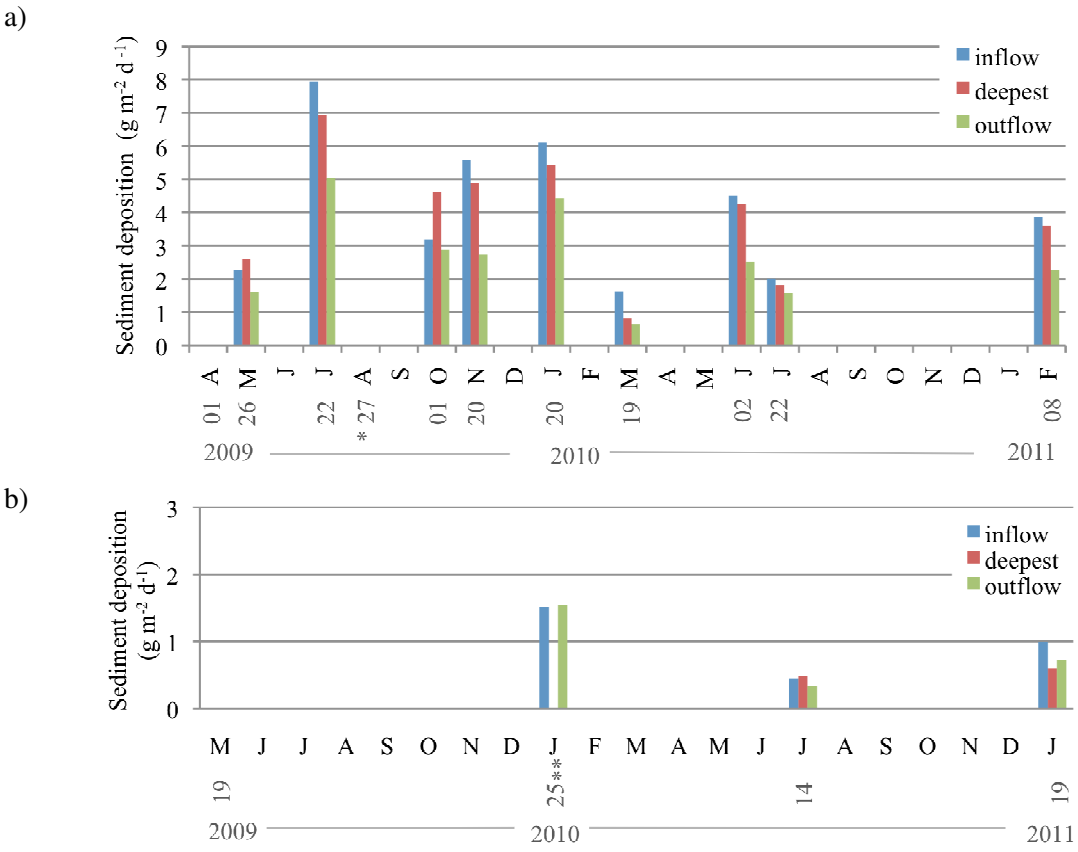

Figure 6.1 – Daily sediment deposition (g m^{-2} d^{-1}) at inflow, deepest point and outflow traps in a) Feeagh during periods of deployment between April 2009 and February 2011 and in b) Guitane between May 2009 and January 2011. Specific dates correspond to trap sample collection and re-deployement. The asterisk (*) indicates the shorter sampling period of the inflow trap. No data for deep water trap from Guitane January 2010 (**).

The estimated cumulative sediment deposition rates (g m^{-2}) in both lakes are shown in Figure 6.2 and are presented in Appendix J. No deep trap samples from Guitane were

available from May 2009 to January 2010. The cumulative deposition in the inflow trap was 4.1 times lower in Guitane than in Feeagh. The total deposition rate ranged between 1,750 and 2,656 g m^{-2} in Feeagh and was highest at the inflow and lowest at the outflow. In comparison, the total deposition rate ranged between 584 and 648 g m^{-2} in Guitane at the out- and inflow, respectively.

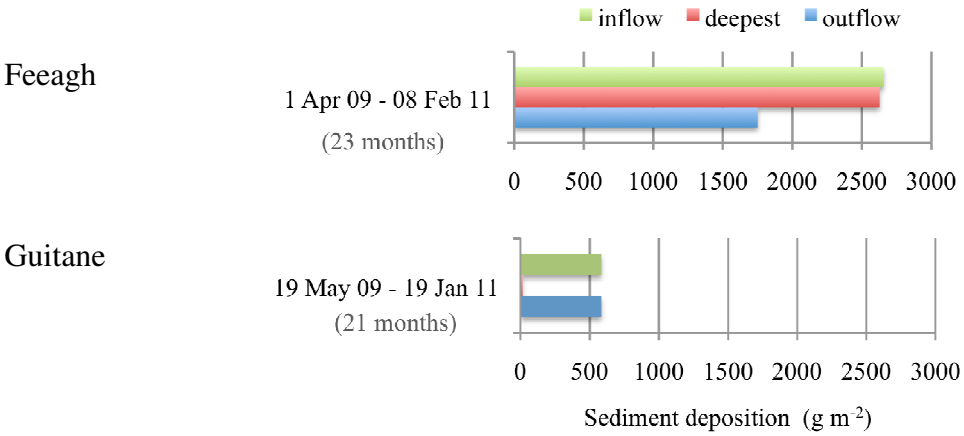

Figure 6.2 – Cumulative sediment sinking flux (g m^{-2}) for inflow, deepest and outflow traps for Feeagh and Guitane. No data for deep water trap from Guitane.

6.3 Organic matter content

In Feeagh the average organic matter content (LOI_{550}%) was 23.9% (inflow), 27.3% (deepest) and 30.7% (outflow). The overall trend of organic content co-varied in the three sampling stations over time (Figure 6.3 a). Raw data are presented in Appendix J. The highest percentages were evident in the samples collected from April to May 2009 and from January to March 2010, while the lowest organic matter content was measured between May and July 2009 (after the flood event on 2nd July 2009). The organic matter content of the top centimetre of the three corresponding sediment cores were generally higher with 32% LOI (inflow and deepest) and 45.4% LOI (outflow).

In Guitane the average organic matter content of the sediment trap samples (Figure 6.3 b) at the inflow, deepest, and outflow positions was 28.8%, 26.0% and 30.1%, respectively. Little variation was evident in both in- and outflow traps over the whole sampling period. The organic content of the trap records from the deepest point decreased from 29.1% to 22.9% in the sediment collected between two sampling periods (January to July 2010 and July 2010 to January 2011). The organic matter content of the surface sediment from deepest point was similar with 21.6% LOI.

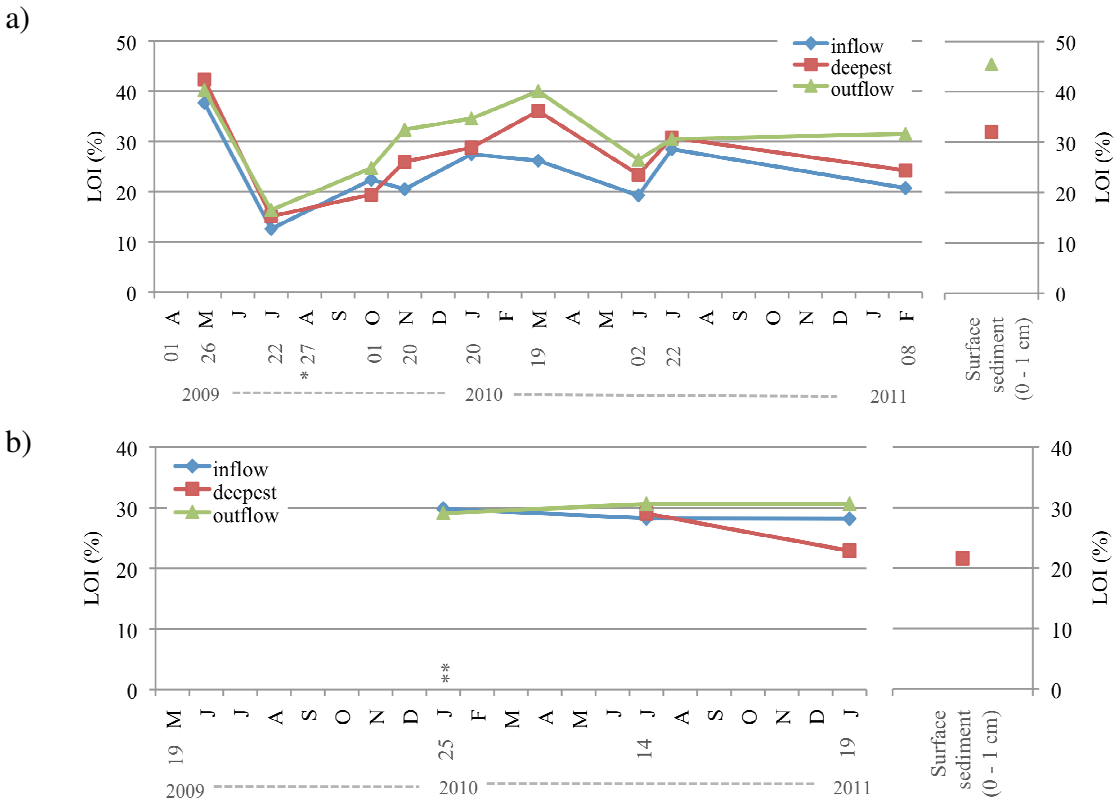

Figure 6.3 – Organic matter content (LOI%) of sediment trap samples collected at the inflow, deepest and outflow traps in a) Feeagh between April 2009 and February 2011 (n=9) and in b) Guitane between May and July 2011 (n=3). The asterisk (*) indicates the shorter sampling period of the inflow trap. No data for deep water trap from Guitane January 2010 (**). The organic matter content of the adjacent surface sediments (0-1 cm) is shown to the right of each graph.

6.4 Total organic carbon and total nitrogen

TOC content of the sediment trap samples from Feeagh (Figure 6.4 a) ranged from 4.9% to 18.2%, with means of 9.3% (inflow), 11.5% (deepest) and 13.1% (outflow). Narrower oscillations and lower TN concentrations of 0.3% and 1.3% were evident (Figure 6.4 b), with overall averages of 0.5% (inflow), 0.7% (deepest) and 0.9% (outflow). Raw data are presented in Appendix J.

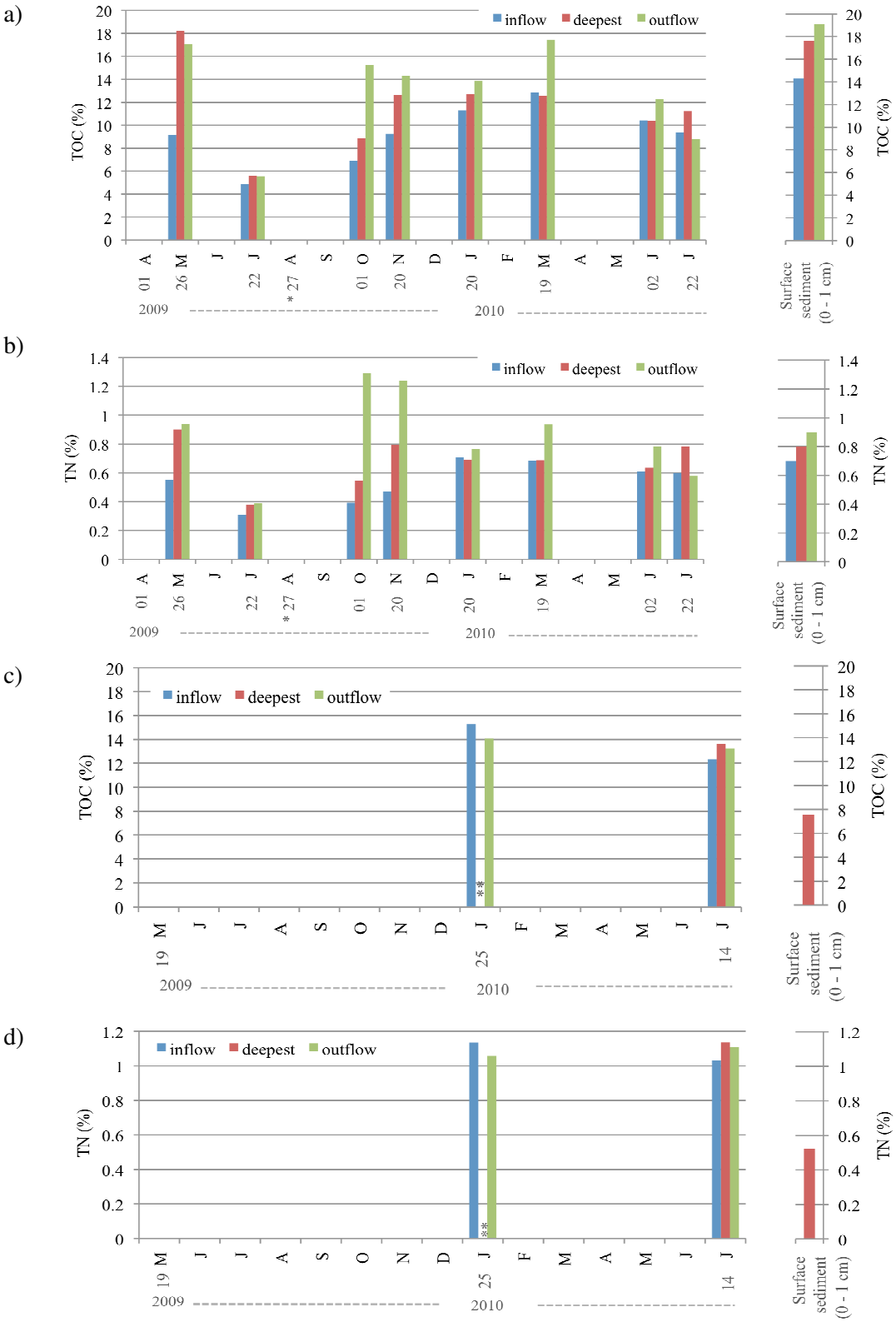

Figure 6.4 – TOC (%) and TN content (%) of sediment trap samples collected at the inflow, deepest and outflow traps in a) and b) in Feeagh between April 2009 and July 2010 (n=24) and in c) and d) in Guitane between May 2009 and July 2010 (n=5). The asterisk (*) indicates the shorter sampling period of the inflow trap. No data for deep water trap from Guitane January 2010 (**). The organic matter content of the adjacent surface sediments (0-1 cm) is shown to the right of each graph.

TOC and TN varied spatially and temporally. Generally lower values were recorded at the inflow and TOC and TN were lowest in the sediment samples collected between May and July 2009. Concentrations increased gradually during the following months in the three traps. The percentages of TOC and TN measured in the adjacent surface sediments had slightly higher concentrations (13.4-19.1% TOC and circa 0.8% TN).

In Guitane TOC in the sediment trap records (Figure 6.4 c) ranged from 12.4% to 15.3%, with a mean of circa 13.8% in the inflow and outflow traps. TOC concentrations decreased from 15.3% to 12.4% and from 14.1% to 13.2% in the outflow trap between January and July 2010. TN values generally fluctuated around 1% (Figure 6.4 d). The TOC and TN in the surface sediment was half the concentration of the sediment trap samples: TOC was 7%, while TN was 0.5%.

The C/N ratio of the sediment trap samples and surface sediments from Feeagh are shown in Figure 6.5 a. The C/N ratios of the trap samples ranged from 11.5 to 20.2, with means of 17.2 (inflow), 16.8 (deepest) and 15.4 (outflow). The greatest variation in C/N ratios was evident at the ouflow trap ranging from 11.5 to 18.6. The spatio-temporal variation was clearly evident with highest ratios at the deepest point from April to May 2009 and the inflow trap from October to November 2009. Lowest ratios of c. 11 were measured at the outflow trap between July and November 2009. The surface sediments in Feeagh revealed C/N ratios of 20.0 (inflow), 16.8 (deepest) and 21.1 (outflow).

In Guitane the C/N ratio of trap samples were similar in each sampling location and showed minor temporal variation (Figure 6.5 b). The C/N ratios ranged from 12.0 to 13.4 and the overall averages were 12.7 (inflow), 12.0 (deepest) and 12.6 (outflow). The surface sediment from the deepest part of the lake had a C/N ratio of 13.4.

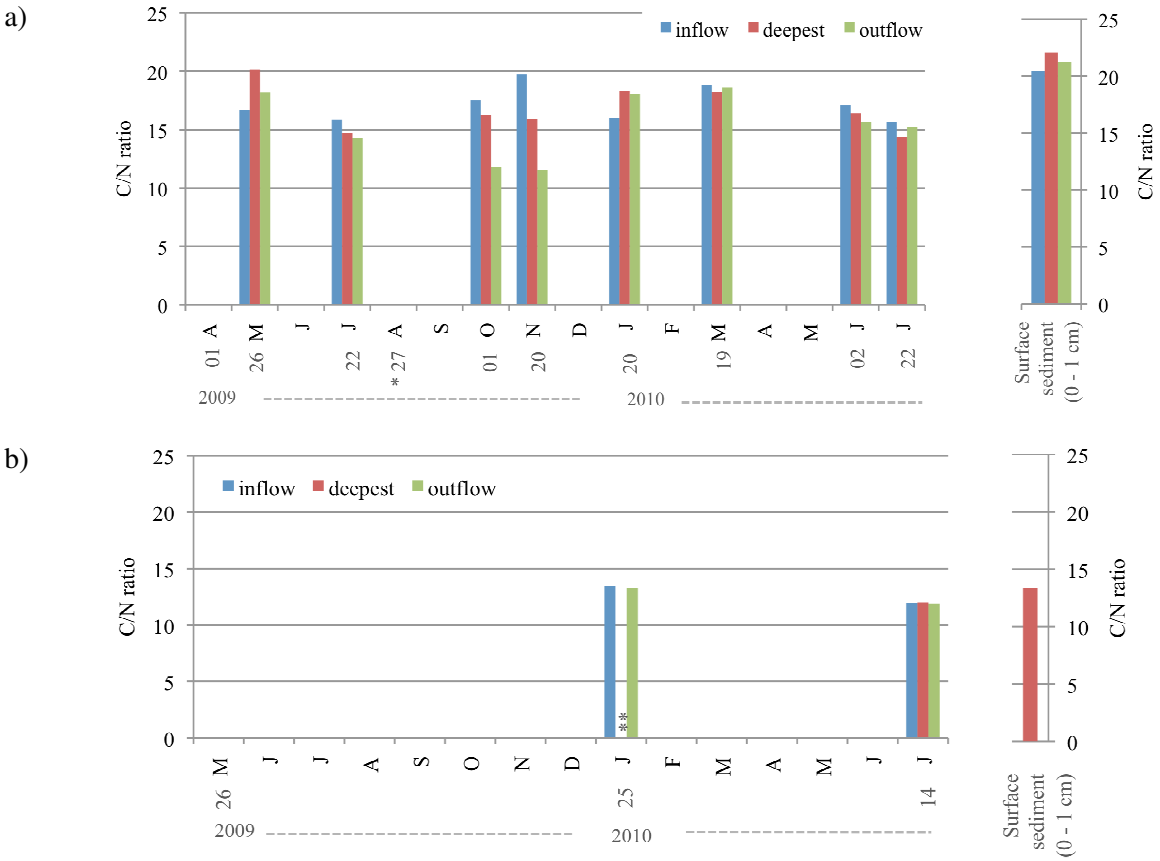

Figure 6.5 – Carbon/Nitrogen ratio of sediment trap samples at the inflow, deepest and outflow traps a) in Feeagh collected between 1st April 2009 and 22nd July 2010 (n =24:) and b) in Guitane between 19th May 2009 and 14th July 2010 (n=5). The C/N ratios of the adjacent surface sediments in both lakes (n=3 and 1, respectively) are depicted to the right. The asterisk (*) indicates the shorter sampling period of the inflow trap. No data for deep water trap from Guitane January 2010 (**).

6.5 Pigments

Pigment extracts from sediment trap, surface sediment and sediment core samples (see Chapter 7) revealed complex mixtures of pigments and their derivatives. Of the 17 pigments identified seven belonged to chlorophylls, nine to carotenoids and one was an UV radiation-absorbing compound (Table 6.1). The chlorophyll pigments included chl-*a*, chl-*b*, *chl-c2* and their derivatives chl-*a*', pheophorbide *a*', phaeophytin-*a* and phaeophytin-*b*. Carotenoids included β-carotene, alloxanthin, aphanizophyll, canthaxanthin, diatoxanthin, echinenone, fucoxanthin, lutein/zeaxanthin (appeared as one peak and could not be differentiated) and myxoxanthin. The UV-pigment was classified as an UV-absorbing compound A-type (McGowan pers. comm.).

Table 6.1 – Pigments identified ( ) in sediment trap and surface sediment (0-1 cm depth) in Feeagh and Guitane

Affinity		**Feeagh**		**Guitane**	
Pigment type	**Pigment**	**Traps**	**Surface Sediment**	**Traps**	**Surface Sediment**
All algae and plantae					
Chlorophyll	chl-*a*				
Chl-derivative	chl-*a*'				
Chlorophyll	chl-*c*2		-	-	-
Chl-derivative	Pheophytin-*a*				
Chl-derivative	Pheophorbide-*a*'				
Carotenoid	β-carotene				
Diatoms, Dinophyta, Chrysophyta					
Carotenoid	Fucoxanthin				
Carotenoid	Diatoxanthin				
Chlorophyta, Euglenophyta, all plantae					
Chlorophyll	chl-*b*				
Chl-derivative	Pheophytin-*b*				
Chlorophyta/Cyanobacteria					
Carotenoid	Lutein/Zeaxanthin				
Cyanophyta					
Carotenoid	Aphanixophyll	**-**	**-**		
	Canthaxanthin				
	Echinenone	**-**		**-**	
	Myxoxanthin	**-**		**-**	**-**
Cryptophyta					
Carotenoid	Alloxanthin				
UV-compound					
UV-compound	Compound-A type	-		-	

The ratio of chl-*a* to pheopigment-*a*, a measure of preservation conditions, was very low in the Feeagh trap and surface sediment core sample (range 0.03 – 0.99). A total of 13 pigments were identified in the trap samples in Feeagh (Appendix K for more details). The total amount of pigments (Figure 6.6) ranged from 59.9 nmol g^{-1} to 468.7 nmol g^{-1}, with means of 160.7 nmol g^{-1} (inflow), 226.3 nmol g^{-1} (deepest) and 136.7 nmol g^{-1} (outflow). The total pigment concentration of each trap sample increased progressively from November 2009 to July 2010, with generally lower concentrations in the outflow trap. A large increase in concentration (to 468.7 nmol g^{-1}) was evident between June and July 2010 in the deepest water, while in- and outflow traps registered smaller rises to 232.9 nmol g^{-1} and 206.9 nmol g^{-1}, respectively. A parallel increase in chl-*a* concentrations was measured over the same period in the surface waters (Figure 5.9 and 6.6). A major rise of chl-*a* was evident from 0.2 µg L^{-1} to 0.8 µg L^{-1} between March and

June 2010 and a minor increase from 0.8 µg L^{-1} to 1.45 µg L^{-1} between June and July 2010.

Chl-*a* and its derivation products dominated each sediment trap sample and pheophorbide-*a'*, was the most prominent degradation product. Pigments belonging to diatoms, Dinoflagellata and Chrysophyta (diatoxanthin and fucoxanthin) reached the highest abundance in the trap sediment collected between March and June 2010 (40.6 nmol g^{-1} at the inflow and 64.6 nmol g^{-1} at the deepest point). Pigments present in Chlorophyta, Euglenophyta and plantae (chl-*b* and pheophytin-*b* peaked in the inflow trap in two samples collected between March and July 2010 and in the deepest and outflow traps accumulated from November 2009 to January 2010. Pigments belonging to Chlorophyta/Cyanobacteria (lutein/zeaxanthin) and to Cryptophyta (alloxanthin) showed a progressive increase between June and July 2010. Traces of cyanobacterial pigments (canthaxanthin) were found in the three traps on two occasions in the late summer samples (June-July 2010).

A total of 15 pigments were identified in the three surface sediments in Feeagh. The total pigment concentrations were similar in the in- and outflow surface sediments (83 nmol g^{-1} and 81 nmol g^{-1} respectively) and highest at the deepest point with 150 nmol g^{-1}. Pigments present in all algae and plantae (chl-*a* and its by-products) dominated the surface sediment sample at the deepest part of the lake (63.9 nmol g^{-1}), while pigments present in Chlorophyta, Euglenophyta and plantae (chl-*b* and pheophytin-*b*) were the most abundant pigments in the in- and outflow surface sediments (35.2 and 31.0 nmol g^{-1}, respectively). The concentrations of siliceous algal (diato- and fucoxanthin) and Cryptophycean pigments (alloxanthin) were always higher at the deepest compared to the other two sites.

The ratio of chl-*a* to pheophytin-*a*, of the trap and surface sediment samples in Guitane ranged from 0.06 to 3.7. The highest ratios (3.4 and 3.7) were found in deepest trap samples collected between January and July 2010. In Guitane a total of 13 pigments were identified in five sediment trap samples (Figure 6.7 and Appendix K for raw data). The total pigment abundance ranged from 172.1 to 1,019.1 nmol g^{-1}, with means of 284.2 nmol g^{-1} (inflow) and 542.9 nmol g^{-1} (outflow). The first period of accumulation from May 2009 to January 2010 showed similar total pigment concentrations in the in-

and outflow traps (172.1 and 189.0 nmol g^{-1} respectively). The total concentrations increased in trap sediments accumulated between January and July 2010, with lowest values at the inflow (396.3 nmol g^{-1}) and highest at the deepest point (1,019.1 nmol g^{-1}). The trap samples appear to track the monthly open water chl-*a* concentration in Guitane, which showed a typical seasonal pattern reaching the highest levels over the summer months (Figure 5.21 and Figure 6.6).

A more detailed examination of the abundance of pigments identified shows that chl-*a* and its derivation products dominated the inflow (87.6 and 246 nmol g^{-1}) and the deepest (652.8 nmol g^{-1}) trap samples that were collected between May 2009 and July 2010. The sediment at the outflow was dominated by pigments belonging to Chlorophyta, Euglenophyta and plantae (chl-*b* and pheophytin-*b*) with 356.7 nmol g^{-1}. Relatively high amounts of siliceous algal pigments (fucoxanthin (227.5 nmol g^{-1}) and diatoxanthin (132.9 nmol g^{-1}) were recorded in the deepest and outflow traps between January and July 2010. Pigments belonging to Cryptophyta (alloxanthin) and Cyanobacteria (aphanizophyll and canthaxanthin) were present with low concentrations in each sample.

Fifteen pigments were identified in the surface sediment sample at the deepest point in Guitane. The total pigment concentration in the surface sediment was 341.7 nmol g^{-1}. Chlorophyll and its derivation products were the most abundant pigments, followed by pigments present in Cyanobacteria (aphanizophyll, echinenone and canthaxanthin), Cryptophyta (alloxanthin), Chloro- and Euglenophtya and plantae (chl-*b* and pheophytin-*b*), Chlorophyta/Cyanobacteria (lutein/zeaxanthin) and siliceous algae (fuco- and diatoxanthin).

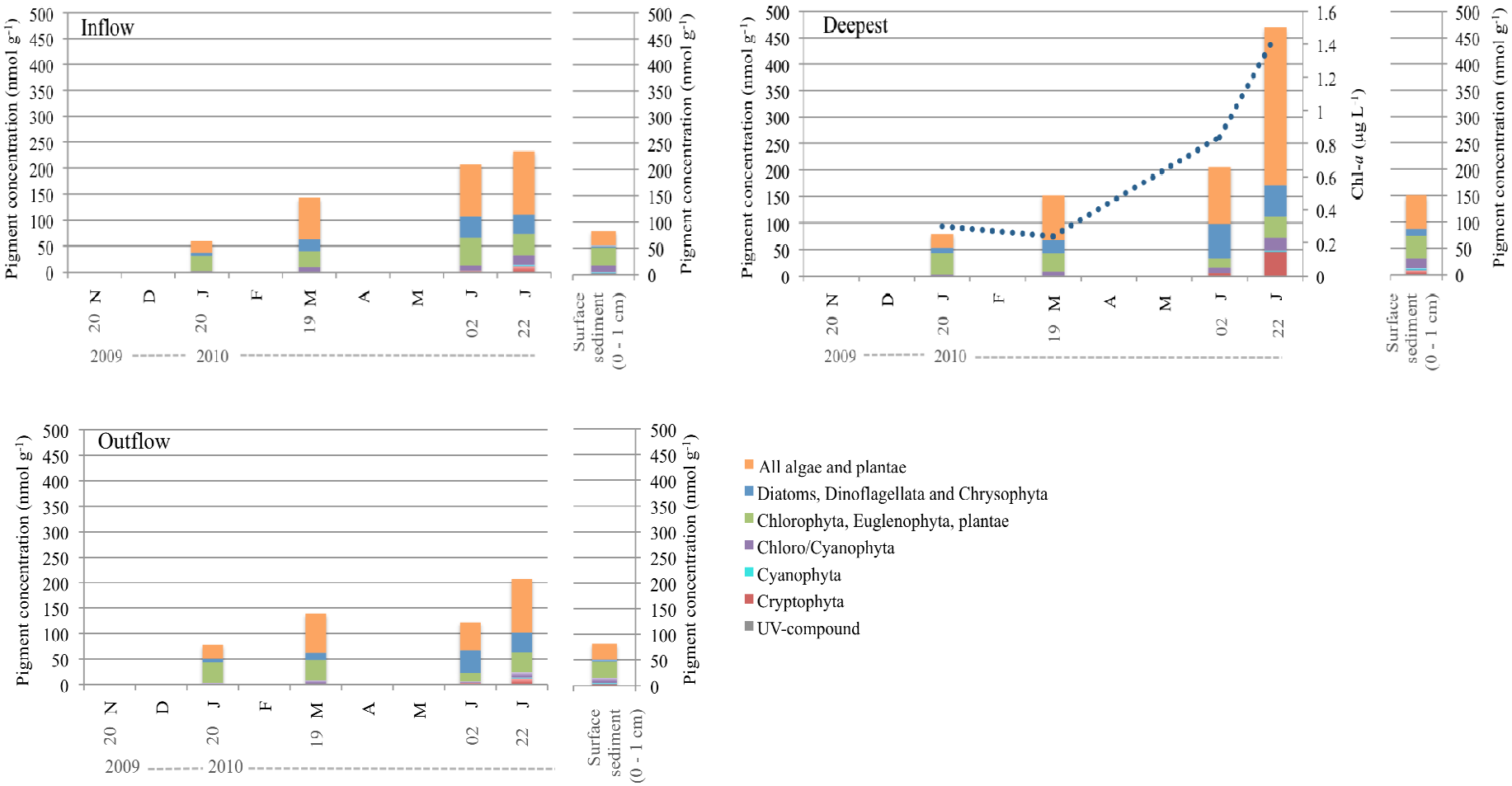

Figure 6.6 – Algal pigment concentrations (nmol g^{-1}) of the identified taxonomic groups (see legend) measured in sediment traps and surface sediment samples from inflow, deepest and outflow sampling stations in Feeagh. Open water chl-*a* concentrations (µg L^{-1}) (dashed line) are included for the deepest sampling station

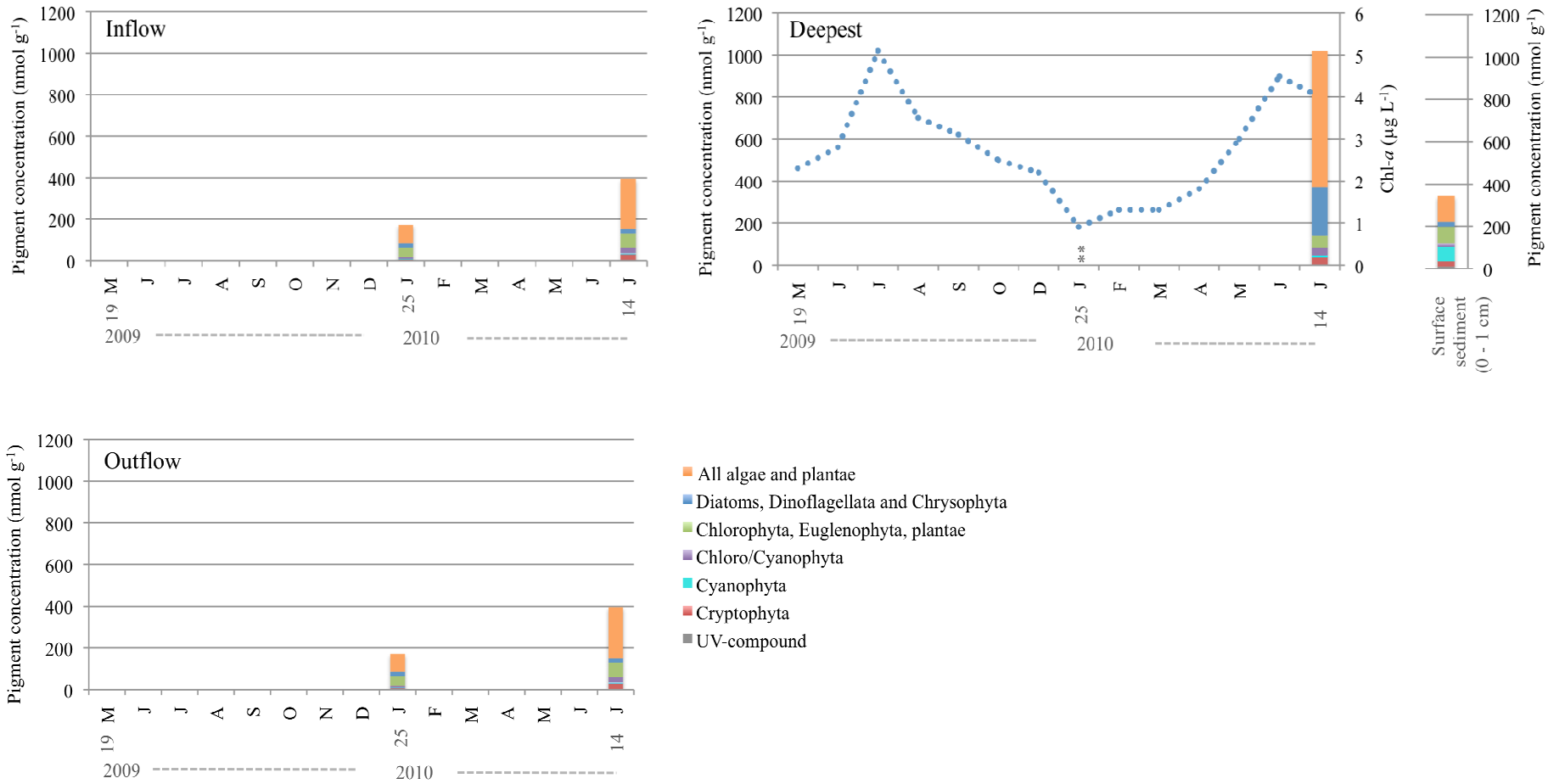

Figure 6.7 – Algal pigment concentrations (nmol g^{-1}) from inflow, deepest and outflow sediment traps and surface sediment sample (0-1 cm) from the deepest waters in Guitane. The legend shows the identified pigments and their taxonomic affinities. Open water chl-*a* concentrations (µg L^{-1}) (dashed line) are included for the deepest sampling station. No data for deep water trap from Guitane January 2010 (**).

6.6 Diatoms

As already shown in the previous chapter, 14 diatom species were identified in the open water samples in Feeagh. A total of 127 diatom taxa were enumerated in 24 sediment trap samples, while 69 diatom taxa were identified in the adjacent surface sediments. A full list of taxa and relative abundances (%) and accumulation for both lakes are presented in Appendix L. The discrepancy in numbers of species identified reflects the different sampling mediums and microscopy constraints. The open water diatom samples were collated to reflect similar time periods to the trap accumulation periods. Six diatom species in the water samples had abundances higher than 5 cells mL^{-1} and included mainly pelagic species such as *Asterionella formosa, Aulacoseira alpigena, Aulacoseira subarctica, Cyclotella radiosa, Cyclotella kuetzingiana* and the pelagic/epiphytic/epilithic *Tabellaria flocculosa.* Four of these species (*Asterionella formosa, Aulacoseira* spp., *Tabellaria flocculosa*) also predominated (≥ 5%) in the trap and surface sediment samples along with *Achnanthidium minutissimum* and *Achnanthes oblongella*. The mean abundance for the relative diatom assemblages in open water, trap, and surface sediment samples are shown in Figure 6.8.

In Guitane a total of 8 species were identified in the 12 open water samples collected between May 2009 and April 2010. Four species identified (*Tabellaria flocculosa*, *Cyclotella* spp., *Asterionella formosa*, *Aulacoseira subarctica*) and the pennate group were encountered in at least five samples with densities greater than 5 cells ml^{-1}. A total of 63 diatom species were identified in five sediment trap samples, while 33 species were encountered in the deepwater surface sediments. In the sediment trap and surface sediment samples the following species had mean percentages greater than 5%: the pelagic taxa *Cyclotella kuetzingiana*, *C. comensis*, *C. radiosa*, *Aulacoseira subarctica*, the pelagic/epiphytic/epilithic *Tabellaria flocculosa* and the epiphytic *Achnanthidium minutissimum* (Figure 6.9).

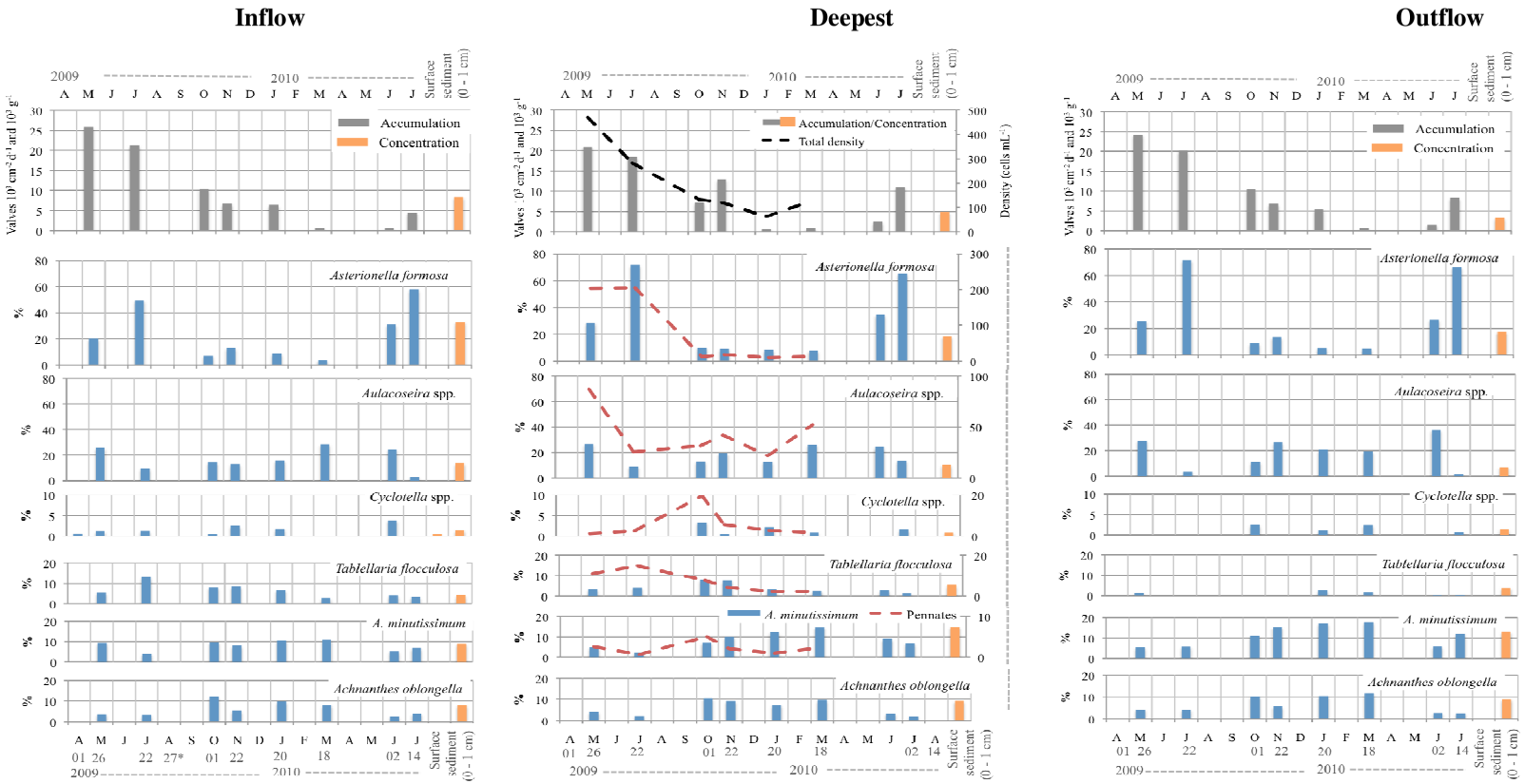

Figure 6.8 – Diatom accumulation (valves 10^3 d^{-1} cm^{-2}) in trap samples (grey columns), diatom concentration (valves 10^3 g^{-1}) of surface sediment (orange columns), total diatom density (cells mL^{-1}) in open water (black dashed line) and relative abundances (%) of the most dominant diatom species encountered in sediment trap (blue columns) in the inflow, deepest and outflow locations and diatom densities (red dashed line) in open water samples in Feeagh. A one-month shorter collecting period is given by the inflow trap in Feeagh (*).

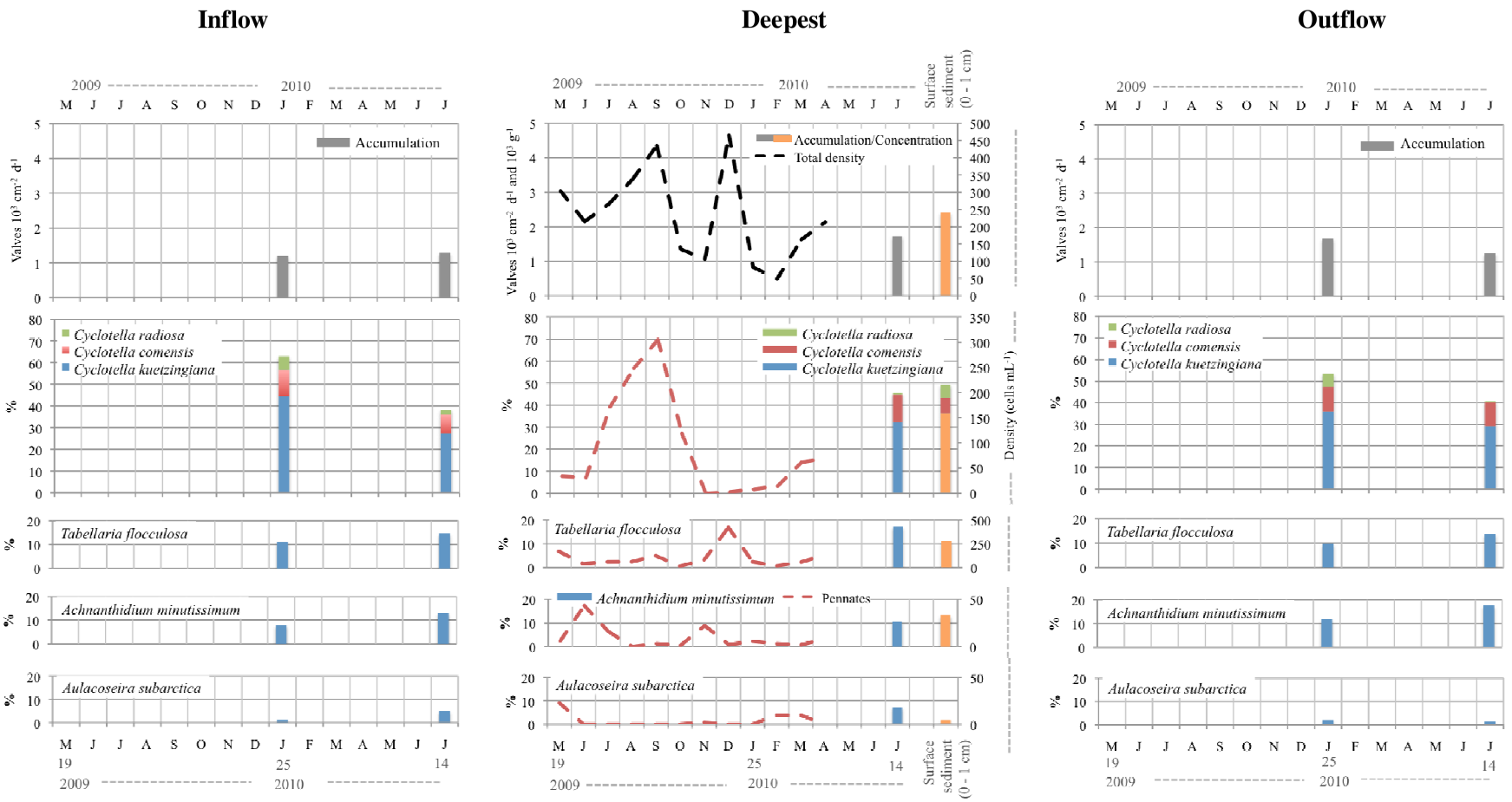

Figure 6.9 – Diatom accumulation (valves 10^3 d^{-1} cm^{-2}) in trap samples (grey columns), diatom concentration (valves 10^3 g^{-1}) of surface sediment (orange columns), total diatom density (cells mL^{-1}) in open water (black dashed line) and relative abundances (%) of the most dominant diatom species encountered in sediment traps (blue columns) in the inflow, deepest and outflow locations and diatom densities (red dashed line) in open water samples in Guitane.

Benthic and planktonic diatom taxa encountered in both sediment trap and surface sediment samples are depicted in Figure 6.10 and raw data are presented in Appendix L. In Feeagh a seasonal pattern in the trap samples was evident, with an increase in planktonic diatoms from April to July in both years, while benthic diatoms dominated over the rest of the period. The surface sediments were mainly composed of benthic species in the inflow and deepest waters, while planktonic taxa predominated in the outflow surface sediment. In Guitane the planktonic taxa were predominant in each sediment trap sample and in the surface sediment from the deepest waters.

Overall mean diatom accumulation rates in the sediment traps in Feeagh were generally less then 12.8 x 10^3 valves cm^{-2} d^{-1} (dry sediment). However, during planktonic peaks in diatom abundance between April and May 2009 accumulation rates of over 21.1 x 10^3 valves cm^{-2} d^{-1} were evident (Figure 6.8 and Appendix L). The highest average diatom concentrations were observed over the same periods in the water column samples. The diatom concentration accumulation in surface sediments ranged between 8.4 x 10^6 valves g^{-1} and 3.2 x 10^6 valves g^{-1} (wet sediment) at the in- and outflow, respectively.

The seasonal occurrence of diatoms in the open water cell (density) and sediment traps (relative abundance) showed similarities and differences (Figure 6.8). *Asterionella formosa* was dominant in the open water samples in the late spring (May-July 2009) and was prominent in trap samples in July. The relative abundances of *Asterionella* ranged from 49.6% at the inflow trap to 71% and 66% at the deepest and outflow trap. A similar pattern was evident with pelagic colonies of *Aulacoseira* spp. Water column densities increased between April-May 2009, October-November 2009 and January-March 2010 and corresponded to the peaks (26.6%, 19.5% and 26.2%) encountered in the deepest sediment trap samples. Similar data were obtained at the in- and outflow sediment trap samples. In contrast, high abundances of *Tabellaria flocculosa* were found in the open water samples between March and July 2009, while trap samples showed peaks later in the season (July-November 2009). Periphytic species (*Achnanthidium minutissimum* and *Achnanthes oblongella*) had low abundances in the open water samples and were consistently present in trap samples with percentages ranging from 2% to 18%. The Feeagh surface sediment samples were dominated by *A. formosa* with 32.9% at the inflow and 17.5% at the outflow. The abundances of *A. minutissimum* and *A. oblongella* were higher at the deepest point (15% and 9.3%,

respectively) compared to the in- and outflow samples (range of 8.1% and 13.1%). *T. flocculosa* ranged between 5.4% (deepest) and 3.9% (outflow). *A. alpigena* had similar percentages at the inflow and deepest trap (3.9% and 4%, respectively), while abundances were low (0.4%) at the outflow.

In Guitane mean diatom valve fluxes ranged between 1.1 x 10^3 and 1.7 x 10^3 valves cm^{-2} d^{-1} of dry sediment in the five sediment trap samples and reached a diatom concentration of 2.4 x 10^3 frustules per g of wet sediment in the surface sediment. The relative abundances of the six most dominant taxa found in the open water, sediment trap and surface sediment samples are plotted in Figure 6.9 and are presented in Appendix M. The lower sampling resolution of the sediment traps in Guitane precluded detailed examination of seasonality, however the open water assemblages are illustrated as monthly densities for comparison. The species relative abundances co-varied at the three sampling stations. *Cyclotella kuetzingiana* was the most dominant taxon in each trap sample with a maximum of 44.7% in the sediment collected between May 2009 and January 2010 at the inflow trap. The relative abundances of *C. comensis* and *C. radiosa* were higher in the trap samples representing May 2009 to January 2010 (45-62%), compared to January to July 2010 (38-49%). Similarly, densities of *Cyclotella* spp. in open water were higher during this first period. *Tabellaria flocculosa* and *Achnanthidium minutissimum* abundances increased at the inflow from 11% to 14.7% and from 7% to 13% respectively for the same periods. Similar percentages were evident for the deepest water trap records collected between January and July 2010 (17.2% and 10.5%, respectively). In comparison, *Tabellaria flocculosa*, the most abundant taxon in open water increased in December 2009, was found in relatively low concentrations in the trap and surface sediment samples. The diatom assemblages found in the deepest surface sediment were similar to the trap assemblages and were composed of *Cyclotella kuetzingiana* (36.3%), *Achnanthidium minutissimum* (13.3%), *Tabellaria flocculosa* (11.6%) *Cyclotella comensis* (7%) and *Cyclotella radiosa* (6%).

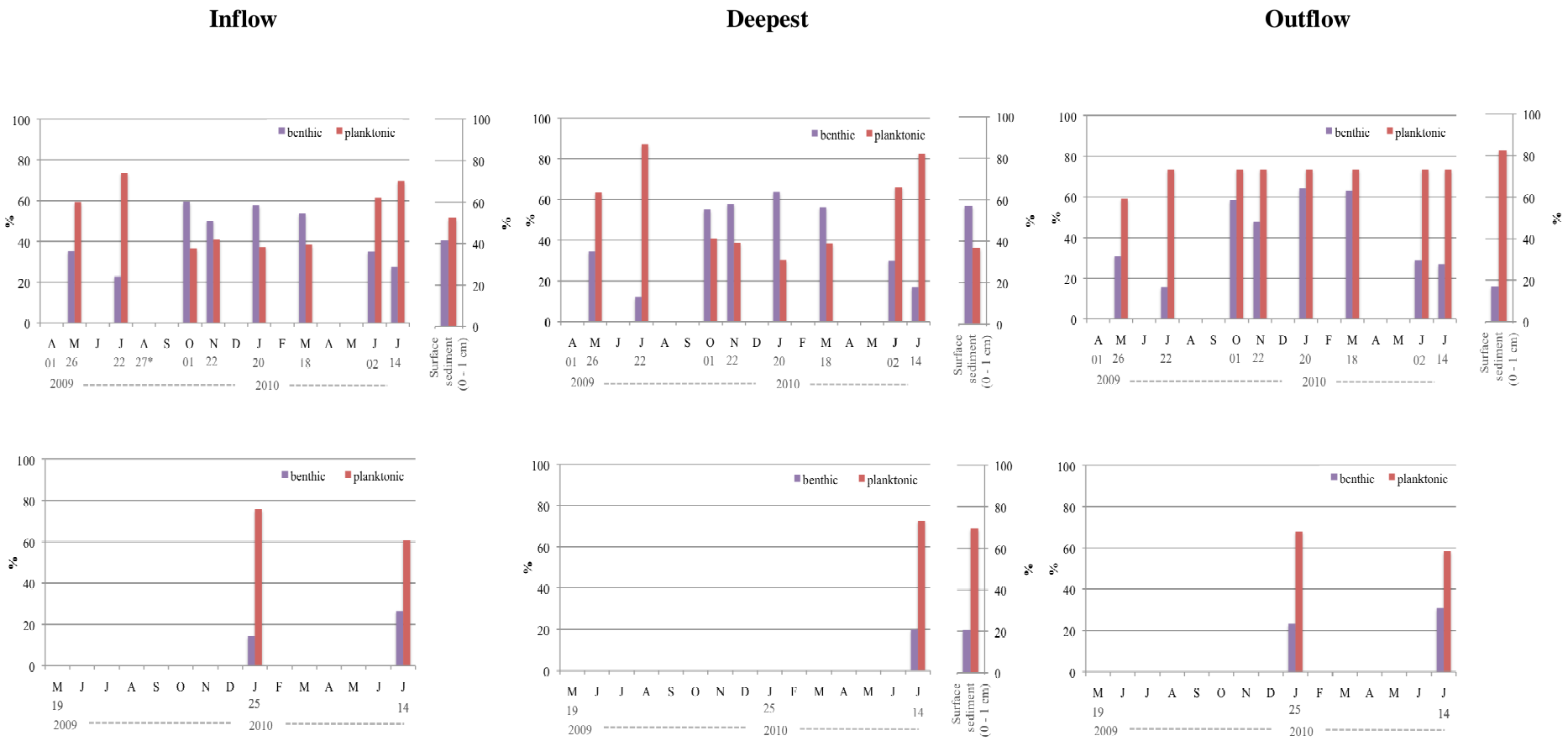

Figure 6.10 - Benthic and planktonic diatom taxa in both sediment trap and surface sediment samples in Feeagh (top) and Guitane (bottom) between April/May 2009 and July 2010. The asterisk (*) indicates the shorter sampling period of the inflow trap.

6.7 Discussion

This chapter has described the within lake variability of sediment trap and surface sediment data in Feeagh and Guitane. The following discussion focuses on how the balance of material falling from the water column, collected in suspended traps and arriving in the surface sediment can be used to augment knowledge of contemporary limnological processes and palaeolimnological reconstructions. Moreover, a range of processes that influence sedimenting material are examined.

6.7.1 Comparability of open water and sediment trap data

Open water samples encompass living communities and represent biomass in a spatially restricted area and at an instant in time. Sediment trap samples in contrast can potentially provide an integrated sample with a broader spatial and temporal coverage (Cameron, 1995; Rautio *et al.*, 2000). The comparison of data from these different spatial and temporal sources is not straightforward as there are multiple influences, such as for example inflowing streams, resuspension from the lake-bed and seasonal algal succession. These processes can affect and alter the biological assemblage and lake sediment composition.

The inflowing streams present in the catchment represent a significant source of allochthonous organic and inorganic sedimenting matter. An example of the latter is given by the mostly inorganic contribution of carbon (low LOI and TOC) collected in all trap samples after the flood event in Feeagh. Moreover, the strong spatiotemporal variation of C/N ratios in the sediment trap samples confirmed the influence of allochthonous contribution, which ranged from a typical sub-equal mixture of algal and vascular plant content to major peat/land plant influence (Ertel & Hedges, 1984; Meyers, 1994, 2003; Lamb *et al.*, 2007; Diefendorf *et al.*, 2008). The higher C/N ratios registered at the inflow trap, together with highest concentrations of chl-*b* and pheophytin-*b*, indicated major terrestrial sources and plant-derived pigments. Moreover, benthic diatoms dominated in the inflow and deepest sediment trap samples, when compared to the outflow sediment trap, where the pelagic community was better represented. In Guitane, there was no evidence of major terrestrial inputs from inflow streams and the C/N ratio indicated a sub-equal mixture of algal and vascular plant content in both trap and surface sediments (Meyers & Lallier-Vergès, 1999; Meyers &

Teranes, 2001; Meyers, 2003). It therefore appears the sedimenting matter and sediment characteristics of Feeagh are more strongly structured by its inflowing tributaries than in the case of Guitane.

Resuspension of material from the lake-bed and horizontal transport, which includes both older sediment and contemporary material and/or in some cases also dead and alive cells, can add a fossil component to trap samples and influence seasonal dynamics (Cameron, 1995; Köster & Pienitz, 2006). For example, epiphytic diatom species (*Achnantidium minutissimum Achnanthes oblongella* and *Brachysira* spp.) in the trap samples encountered in both lakes may indicate resuspension of the sediments. However, large, deep lakes are generally better study sites for combined sediment trap/palaeolimnological studies because traps are generally situated below the photic zone, and thus epipelic and epiphytic growth are of minor importance and thus constitute a minor component (Köster & Pienitz, 2006). If the diatoms in the trap samples originated mainly from resuspension, then we could expect seasonal homogeneity of taxa. This was not the case in either study sites where seasonal succession was clearly evident. Seasonal succession of diatoms in open water and sediment trap samples has been observed in shallow (Cameron, 1995; Lotter & Bigler, 2000; Köster & Pienitz, 2006; Hausmann & Pienitz, 2009) and deep lakes (Rautio *et al.*, 2000; Kirilova *et al.*, 2008). Highest diatom concentrations in the water column and maximum daily diatom sedimentation and peaks in relative diatom abundances in the sediment traps reflected seasonality in Feeagh. The small number of samples from Guitane precluded similar conclusions. However, both lakes experienced an increase in pelagic taxa in the spring-summer samples and in benthic taxa in the autumn-winter samples. Moreover, in Feeagh peaks in diatom abundance encountered in each sediment trap sample corresponded to maximum cell densities in the open water. The percentages of the major diatoms in the sediment traps were also comparable to those in the water column in most trapping periods. For example, *Aulacoseira* species inhabited the surface waters and was reflected in trap samples during overturn (from May 2009 and from October to March 2010), when the water turbulence was sufficient to keep the species in the euphotic zone. Turbulent conditions are especially important for *Aulacoseira subarctica* because of its rapid sinking rate, resulting from its high silica content (Round *et al.*, 1990). A further example is given by the abundance of lightly silicified, spindle-shaped *Asterionella formosa* in the open water during early spring and

summer 2009, that corresponded to a peak in the sediment traps collected between May and July 2009. Peaks in *Asterionella* are typically related to thermal stratification (Köster & Pienitz, 2006). In contrast, this seasonal pattern was not reflected in *Tabellaria flocculosa*, which has also been described as an indicator of thermal stratification (Köster & Pienitz, 2006; Hausmann & Pienitz, 2009). *Tabellaria* increased in the water column in summer, but reached highest abundances in the sediment trap samples in winter. In Guitane *Tabellaria flocculosa* dominated the open water samples, while *Cyclotella* spp. dominated in the trap samples. *Tabellaria* had its highest cell densities in the open water in December, when the whole water column was mixing and available nutrients were released from the hypolimnion. Phytoplankton, including planktonic diatoms, are known to benefit from these nutrient rich conditions with increased turbulence in the water column (Gaedke & Weisse, 1998; Hausmann & Pienitz, 2009).

Several studies have described seasonal variation in the abundance of algal pigments recorded in sediment trap samples (Livingstone & Reynolds, 1981; Cole *et al.*, 1985. Even in the deepest trap (at 3,560 m depth) of an array of trap sets in the Panama Basin, the flux or deposition of algal pigments varied seasonally (Cole *et al.*, 1985). Algal remains recovered from sediment traps reflected the annual phytoplankton succession in many lakes (Livingstone & Reynolds, 1981; Reynolds *et al.*, 1982; Hamilton-Taylor *et al.*, 1984; Bianchi *et al.*, 2002; Yacobi & Ostrovsky, 2008). Progressive seasonal increases in total pigment concentrations were also evident from March to July 2009 in Feeagh and between January and July 2009 in Guitane, confirming a record of seasonal succession in sediment trap samples.

6.7.2 Comparability of sediment trap and surface sediment data

The sources and alteration of organic matter can vary substantially from place to place within a lake (Tenzer *et al.*, 1997; Talbot & Laerdal, 2000) and can also vary temporally (Meyers & Teranes, 2001). For example, differences in TOC concentrations and C/N ratios in surface sediments with increasing distance from shore (Talbot & Laerdal, 2000) and with greater water depth (Tenzer *et al.*, 1997) have been described. Lotter & Bigler (2000) attributed decreases in TOC and TN content of surficial sediments in shallow shore regions to the presence of coarser mineral particles. Lower mean TOC

concentrations were found at the northernmost and shallower site in Feeagh in both sediment trap and surficial sediment samples, while concentrations were higher in the deepest and southernmost sites, where fine-grained sediments slowly settled to the lake bottom. In Guitane LOI and TOC were spatially and temporally homogenous in the sediment traps, while concentrations were 50% lower in the surface sediments. The lower TOC concentrations could result from the diagenesis of organic matter (Meyers & Lallier-Vergès, 1999). It is known that organic matter consumption is extensive in the surface layers of sediments and several studies showed that generally more than 50% of the organic carbon reaching the lake- or seafloor is destroyed in the bioturbated layer (Cobler & Dymond, 1980; Prahl *et al.*, 1989). However, comparisons of the C/N ratios of trap and sediment samples in both lakes showed similar values (range 11.5-19.7 versus 18.6-20.2 in Feeagh and 11.9-13.5 versus 13.4 in Guitane), which suggests that these bulk parameters retain source information even when incorporated into the sediment record (Meyers, 1994).

Major differences between the total pigment concentration in the sediment traps and surface sediment were observed and could probably be attributed to the sampling period and in part to the sampling locations. For example, surface sediment pigment concentrations collected in January 2011 were most similar to the winter sediment trap samples. In addition, pigment concentrations in the surface sediment samples from the deepest waters were twice as high as the shallower in- and outflow samples. This is probably due to the deepwater bathymetry of the depositional area as shallow, oxygenated sediments are known to dilute the sediment pigment record (Sanger, 1988). Another possible explanation of the higher algal pigment concentrations in the deep water sediments could be related to higher depositional rates, which can increase the effective sedimentation rate of pigments into the fossil record, and thereby, decrease degradation at the sediment surface (Leavitt, 1993). A further explanation, suggested by Moss (1968), is that carotenoid pigment concentrations of sediments increase with water depth.

Diatom assemblages in superficial sediments can vary with water depth. Lotter & Bigler (2000) found that assemblages within the littoral zone (8-10 m depth) were dominated by periphytic diatoms (mainly *Fragilaria* spp.), whereas in the deeper surficial sediments (> 10 m depth) valves of planktonic taxa (*Cyclotella comensis*)

predominated. The influence of water depth on diatom assemblages in both study lakes was observed and assessed. In Feeagh, the deepest surface sediment sample was dominated by benthic taxa, while the southernmost surface sediment sample near the outflow was dominated by planktonic taxa. Similarly, the sediment trap samples were characterized by a clear rise in benthic taxa between July 2009 and March 2009 in the inflow and deepest trap samples, while the outflow trap was dominated by planktonic taxa. In Guitane planktonic diatoms dominated both the trap and superficial sediment samples. Comparisons between trap and sediment samples in a shallower lake showed that high relative abundance of *Asterionella* spp. during summer months were not reflected in the sediments, while higher abundances of *Cyclotella* sp. were found in the sediments compared to the traps (Köster & Pienitz, 2006). Similar phenomena in *Asterionella* and *Cyclotella* sp. were observed in Feeagh and Guitane. Köster & Pienitz (2006) together with Rautio *et al.* (2000) point out that these differences may be caused by inter-annual variability in the seasonal cycle of the lake. Also Cameron (1995) suggests that surface sediments are subjected to physical (e.g. currents) and biological (e.g benthic organisms) bioturbation with existing surface sediment assemblages, which could account for a degree of dilution and (upward and downward) mixing, caused by the technical processes of retrieving sediment samples (e.g. smearing by the sides of the corer and time averaging of samples by core slicing thickness).

6.7.3 Sedimentation dynamics in lakes

Particle settling flux or sediment deposition in sediment trap samples between the two study sites differed spatially and temporally. In particular, the estimated cumulative sediment deposition rate was four times higher in Feeagh compared to Guitane. Feeagh had pronounced spatial and temporal variation, while the sediment deposition in Guitane was more homogenous, however, the number of samples collected was lower.

Studies of settling particles intercepted by sediment traps in northern Estonia (Terasmaa & Punning, 2006) and in many boreal lakes (von Wachenfeldt & Tranvik, 2008a) displayed marked seasonal variation and increased particulate deposition concurrent with the onset of stratification in spring and summer. The sediment deposition rates or particle settling flux estimated in Feeagh and Guitane over nearly two years showed clear temporal variation. If the flash flood event is excluded in Feeagh the sediment

deposition was generally higher during autumn-winter when complete overturn of the water column occurred. In Guitane higher sediment deposition rates were measured in late summer-autumn and early winter 2009 and autumn-early winter 2010, compared to the late winter-spring and early summer 2010. This suggests that sediment deposits are mainly allochthonous in nature (shown by their C/N ratios) and are influenced by a combination of morphometric, climatic and land-use characteristics.

Several regional studies showed that morphometric catchment properties, such as drainage ratio, fluvial inputs, catchment slope and presence of upstream lakes have been found to be related to allochthonous inputs of sedimenting matter (Engstrom, 1987; Rasmussen *et al.*, 1989; D'Arcy & Carignan, 1997; Weyhenmeyer & Bloesch, 2001; Xenopoulos *et al.*, 2003; Sobek *et al.*, 2007). Lakes situated within climatically homogeneous regions (Sobek *et al.*, 2007) and with a large drainage area compared to the lake area, and consequently with a large drainage ratio (catchment : lake area), are thought to receive high inputs of allochthonous particulate and dissolved matter (del Giorgio & Peters, 1994; Sobek *et al.*, 2007). High concentrations of allochthonous suspended particle matter in lakes are known to be a precursor of sinking particles (von Wachenfeldt *et al.*, 2008b), that facilitate and contribute to the flocculation, coagulation and subsequent sedimentation in traps and sequestration in lake sediments (Schindler, 1971; Rasmussen *et al.*, 1989). Feeagh has a drainage ratio of 21.4, while the ratio for Guitane is 7.7. This difference clearly has consequences for the terrestrial carbon inputs and consequently the diverse sediment deposition rates in each lake.

Lake fluvial input can be higher adjacent to inflowing streams or in the centre of cone shaped basins (Moss, 1998). Fluvial input to Feeagh enters the lake at the northern end through two main inflow streams. Higher rates of deposition are generally found near inflows due to the rapid settling of the heavier mineral fraction. Allott *et al.*, (2005) estimated sediment deposition in Feeagh over 14 months (December 2000 – January 2002) and found 1,741 g m^{-2} at the inflow and 610 g m^{-2} at the outflow. In the current study cumulative deposition rates were estimated over 16 months (November 2009 - February 2011). Similar rates were found in the inflow trap (1,968 g m^{-2}), while a higher sediment deposition was calculated at the outflow trap (1,170 g m^{-2}). Guitane, in contrast, has one main and three small inflow rivers, however little spatial variation in sediment deposition rate was evident and thus no fluvial influence was apparent.

Lakes draining relatively large and flat catchments tend to have higher inputs of allochthonous suspended solids into lakes as a result of greater importance of shallow flow-paths through soils and greater percentages of wetland (Rasmussen *et al.*, 1989; Pace & Cole, 2002; Sobek *et al.*, 2007). In the Burrishoole catchment the presence of upstream lakes and the lower relief of some sub-catchments offers higher water storage capacity. Allott *et al.* (2005) describe steep slope tributaries as "hydrologically flashy" due to the rapid transfer of rainfall to streams by quick-flow processes. Steep subcatchments are characterized by a very limited water storage capacity due to the presence of impermeable rock types and relatively impermeable peats and peat-podsols and thus, favour higher in-wash of terrestrial carbon sources. Similar flow regimes were described also in North Wales (Bird *et al.*, 1990). In Guitane the steepest tributaries enter the lake to the south-east. The catchment contains more permeable sandstone and volcanic rocks overlain with peaty soils, known to be more capable of filtering through flows and are characterized by a higher water storage capacity.

The latest Intergovernmental Panel on Climate Change (IPCC, 2007) pointed out that aquatic and terrestrial ecosystems are being strongly affected by climate change, particularly in the form of increases in regional temperature and precipitation. Several studies demonstrate that changes in the magnitude and seasonality of precipitation and runoff are expected to have significant effects on dissolved, colloidal and particulate carbon concentrations (Andersson *et al.*, 1991; Pace & Cole, 2002) and water quality in lakes (Whitehead *et al.*, 2006; Jennings *et al.*, 2010; Naden *et al.*, 2010). In Ireland a shift has been observed to increased total annual precipitation amounts for the west coast stations for the period 1957-2006 (Kiely *et al.*, 2010). The years 2008 and 2009 experienced the breaking of many rainfall records throughout the country (Lennon & Walsh, 2008; Walsh, 2010). In particular, two distinct heavy rainfall events or periods occurred within the two study sites: an extreme rainfall and flash-flood event occurred in the Burrishoole catchment on the 2nd July 2009, (Fealy *et al.*, 2010) and prolonged heavy precipitation period was recorded over three weeks in November 2009 in Kerry, including Guitane catchment. The Burrishoole flash flood event is described as a once in 250 year event (Fealy *et al.*, 2010), while the daily rainfall in Kerry in November was more than twice the national average and the wettest month on record (Kiely *et al.*, 2010; Walsh, 2010). In Feeagh the extreme rainfall event was linked to the maximum sediment deposition rate collected from trap samples. In contrast, Guitane had low and

relatively homogenous sediment deposition rates. This shows that the intensity of the rainfall in Feeagh gave rise to increased inflow of allochthonous inorganic suspended solids (shown by low LOI and TOC), and consequently in higher sediment deposition rates into the lake. The availability of suspended solids is also associated with the length of the soil-drying period, which is related to air temperature, solar radiation and wind conditions, leading to impacts on soil moisture levels (Naden & McDonald, 1989; Davidson & Janssens, 2006). The soil drying period determines the decomposition and mineralisation rates of organic matter, and hence affects the transport of sediment into surface waters (Reynolds & Fenner, 2001a; Tranvik & Jansson, 2002; Hudson *et al.*, 2003; Worrall *et al.*, 2003a). The heavy rainfall event was preceded by approximately six weeks of dry settled weather and it is likely that the water was unable to soak into the ground. Consequently, only the superficial soil strata were re-saturated and sand, silt and other solids were washed out rapidly (Mitchell & McDonald, 1992; Buffam *et al.*, 2001). Jennings *et al*. (2010) describes evidence of organic carbon flushing from high frequency measurements of CDOM fluorescence, which were used to quantify the fluorescent fraction of these coloured organic particles in inflows to Feeagh.

Land-use in both study catchments is characterized by extensive amounts of peat soil (64% in Burrishoole and 83% in Guitane) representing significant carbon stores (Free *et al.*, 2006). Forest cover accounts for nearly one quarter (23%) of the Feeagh catchment, while Guitane has none. Forested catchments generally contribute higher terrestrial carbon and nutrient delivery in the receiving water (Rodgers *et al.*, 2008; Rodgers *et al.*, 2010b; Rodgers *et al.*, 2011). Additionally, extensive grazing by livestock in the catchments has been an issue historically (CSO, 1991; Weir, 1996; CSO, 2000, 2006, 2011).

6.8 Conclusions

The investigation of the water column chemical and biological parameters (Chapter 5), sediment trap and surface sediment samples (this chapter) from Feeagh and Guitane revealed results that are relevant for longer term palaeolimnological examination of lake sediment archives. The study of seasonal ecological responses and sedimentation of particulate and dissolved matter from both lakes informs interpretations of the sediment record. The within lake and between lake spatial and temporal variability reflect how

differences in catchment, lake size and morphometry influence sediment deposition. Additionally, trap samples clearly reflected seasonal algal succession (in fossil pigments and diatom assemblages) and interactions with climate parameters were demonstrated when lake ecosystem responses were evident following heavy rainfall events. The results of this study emphasize the interdependence of water column parameters, the downward flux of particulate matter and associated constituents and the balance of material arriving in the surface sediments. With time this material accumulates to form sediment archives, which are explored in the next chapter.

Chapter 7 - Sediment core reconstructions for Feeagh and Guitane

7.1 Introduction

The collection of sediment cores from Feeagh and Guitane allowed detailed reconstructions of lithological, geochemical and biological proxies. Three sediment cores were collected from Feeagh to examine spatiotemporal responses across the lake, while one representative core from the deepest point of Guitane was retrieved. Historical change in organic matter (LOI_{550}, TOC, C/N ratio) and variations in algal pigments and diatoms are outlined. The reconstruction of the UVR index gave an indication of the depth of penetration of UV radiation within both lakes. Fossil diatoms were enumerated for Guitane, while a fossil diatom profile for Feeagh was assembled during ILLUMINATE project (Dalton *et al.,* 2010).

7.2 Sediment Chronology

The sediment core collected from the deepest point of Feeagh was cross-correlated (Appendix N a and b) with a radiometric (^{210}Pb, ^{137}Cs and ^{241}Am) chronology established by Dalton *et al.* (2010) from the same sampling location. The lowermost sample in the 40 cm long core collected for this Ph.D. project was estimated to date to 1942. The estimated sediment accumulation rate ranged from 0.171 to 0.288 g cm^{-2} yr^{-1}. No chronology was established for the inflow and outflow cores. Results for these cores are reported according to sediment depth and compared with the estimated dates for the deep water core.

The 53 cm sediment core collected from the deepest point of Guitane was analysed for a natural radioactive isotope of lead (^{210}Pb) and for two artificial fallout radionuclides (^{137}Cs and ^{241}Am). Their concentrations measured throughout the sediment core are listed in Appendix O. The equilibrium depth between the activity of total and supported ^{210}Pb was reached at c. 13.5 cm depth (Figure 7.1 a). Unsupported ^{210}Pb activity (the subtraction of supported ^{210}Pb from total ^{210}Pb activity) can be divided into two phases: in the first 4 cm it declined irregularly with depth (Figure 7.1 b), suggesting an increase

in sediment accumulation, while below 4 cm depth it declined more or less exponentially with depth, indicating a relatively uniform sediment accumulation rate. The radionuclide ^{137}Cs peaked at 3.5 cm depth and traces of ^{241}Am were detected in the samples between 4.5 and 6.5 cm depth reaching a maximum at 6.5 cm. As the ^{137}Cs peak is not in the depth range where ^{241}Am appears, it is very likely that the ^{137}Cs peak reflects the fallout from the 1986 Chernobyl accident. This peak along with sedimentation rates and the sediment sub-sampling resolution of 1 cm has probably obscured the ^{137}Cs fallout maximum in 1963 from the atmospheric testing of nuclear weapons.

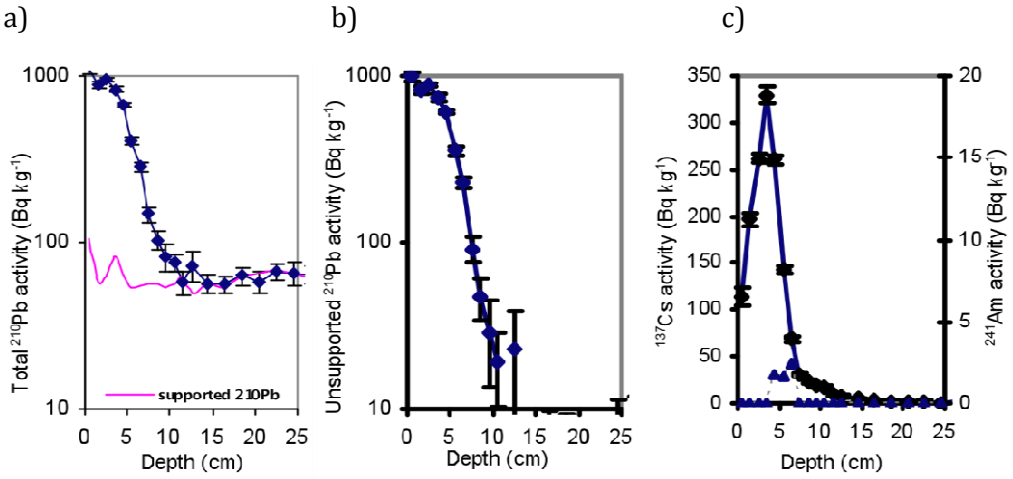

Figure 7.1 – (a) Total and supported ^{210}Pb, (b) unsupported ^{210}Pb, and (c) ^{137}Cs (diamond) and ^{241}Am (triangle) concentrations versus depth for Guitane (Graphs provided by H. Yang, UCL).

Both CRS and CIC dating models (Appleby, 2001) were applied to calculate ^{210}Pb dates. The CRS model placed 1986 and 1963 at 3.5 and 5.5 cm, respectively, which conforms with the ^{137}Cs and ^{241}Am records, while the CIC model places 1986 and 1963 at 5 and 6.5 cm deeper than those suggested by the ^{137}Cs and ^{241}Am records. The disagreement between the CIC model and the ^{137}Cs/^{241}Am records is possibly due to the non-monotonic variation in unsupported ^{210}Pb activities in the top 4 cm that diluted unsupported ^{210}Pb activities (Appleby, 2001). The chronology calculated using the CRS model is shown in Figure 7.2 and Table 7.1. Results show that the top 11.5 cm dates from 1840 and covers the past 170 years. The sediment accumulation estimate was relatively stable with an average of c. 0.0093 g cm^{-2} yr^{-1} from the 1850s to the 1980s, followed by a slight increase in the last thirty years. For the purpose of this research the recent sediments are of most interest. The base of the core was not dated, but based on

linear regression of radiometric dates c. 1200 *anno Domini* is suggested. An accelerated mass spectrometry radiocarbon analysis would provide the radiocarbon age of the core base. The pre-1840 period (11.5 - 53 cm) changes are reported according to sediment depth.

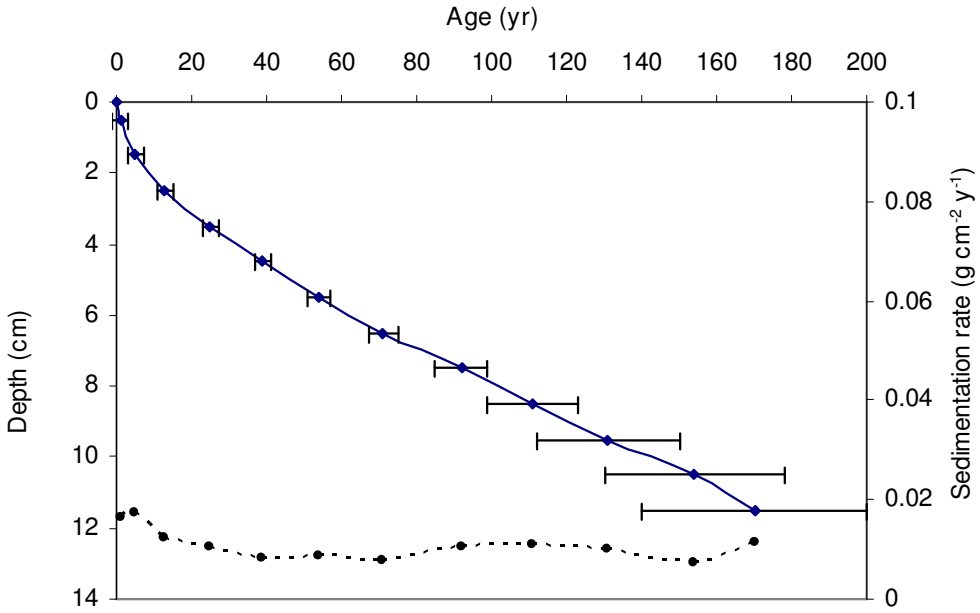

Figure 7.2 - Radiometric chronology of the sediment core taken from Guitane, showing the CRS model, ^{210}Pb dates and sedimentation rates. The solid line shows age, while the dashed line indicates sedimentation rate (Graph provided by H. Yang, UCL).

Table 7.1 - ^{210}Pb chronology of sediment core from Guitane.

Depth	**Dry mass**	**Chronology**			**Sedimentation Rate**		
		Date	**Age**				
cm	**g cm^{-2}**	**AD**	**yr**	**±**	**g cm^{-2} yr^{-1}**	**cm yr^{-1}**	**± %**
0	0	2010	0				
0.5	0.0185	2009	1	2	0.0163	0.275	7.3
1.5	0.0890	2005	5	2	0.0172	0.183	5.3
2.5	0.2055	1997	13	2	0.0125	0.101	4.6
3.5	0.3365	1985	25	2	0.0104	0.079	6.3
4.5	0.4690	1971	39	2	0.0081	0.063	6.7
5.5	0.5940	1956	54	3	0.0088	0.065	10.1
6.5	0.7380	1939	71	4	0.0079	0.048	14.9
7.5	0.9235	1918	92	7	0.0104	0.053	27.9
8.5	1.1255	1899	111	12	0.0111	0.054	46.1
9.5	1.3310	1879	131	19	0.0099	0.049	79.1
10.5	1.5295	1856	154	24	0.0072	0.038	100.3
11.5	1.7125	1840	170	30	0.0112	0.062	123.0

7.3 Sediment Description

The sediment cores extracted from Feeagh and Guitane did not show any apparent variation in sediment type and were dark in colour. The colour of the sediment cores from Feeagh ranged from olive black (Hue 5 Y 3/1) to brownish black (Hue 2.5 Y 3/1) (Oyama & Takehara, 1967) from the core bottom to the core top. The sediment from Guitane ranged from dark brownish (Hue 7.5 YR 2/2) to very dark brown (Hue 7.5 YR 2/3) from the core bottom to the core top. The sediments were composed of homogeneous soft peaty-mud with no particular visible textural changes.

7.4 Sediment Lithology

The organic matter content (LOI_{550}) of the three sediment cores collected in Feeagh ranged from 15% to 47.6%, with means of 23.2% (inflow), 36.7% (deepest) and 30.6% (outflow) (Figure 7.3 and Appendix P). The lowest organic matter content was evident in the inflow sediment core (range 15.0 – 35.2%), while the highest was measured in the deepest core (range 24.6 – 47.6%). The three cores showed a steady increase of organic matter from the core bottom to c. 10 cm. The highest organic matter content was evident at c. 10 cm depth in the inflow and outflow core and at approximately 1990 (c. 14 cm depth) in the deep water core. A decreasing trend is then evident with a minimum at c. 2 cm depth in each core (c. 2009 in deep water core), before LOI_{550} increases again at the top of each core. In Guitane, organic matter content was characterized by multiple minor peaks and troughs ranging from 10.8% to 21.6% (Figure 7.3 and Appendix P). Four peaks between 18% and 22% LOI at 52, 36, 13 cm depth and at c. 2009 (0.5 cm depth) are interspersed with troughs of 10% and 11% at 17 cm depth and at c. 1899 (8 cm), respectively.

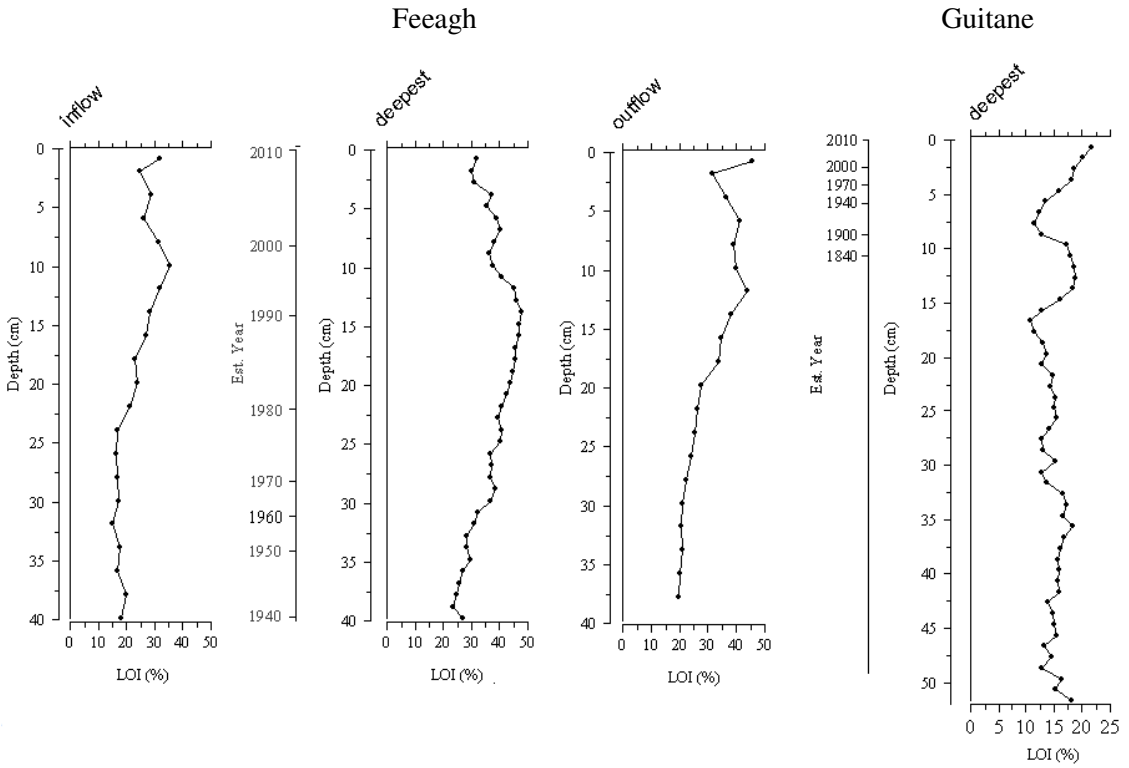

Figure 7.3 – % LOI_{550} from the inflow, deepest and outflow sediment cores collected in Feeagh (on the left) and the deepwater sediment core from Guitane (on the right). Estimated chronologies are available for the Feeagh and Guitane deepwater cores.

7.5 Geochemical proxies

7.5.1 Total organic carbon, total nitrogen and C/N ratio

TOC content of the three sediment cores from Feeagh ranged from 7.7% to 27.1%, while total nitrogen concentrations varied from 0.3% to 1.3% (Figure 7.4). Raw data are presented in Appendix P for both lakes. TOC and TN co-varied in each sediment core. An increasing trend was evident in the inflow core with a peak of 17.8% in TOC and 0.9% in TN at c. 4 cm depth. A similar increasing trend was recorded in the deepest core with two peaks of 25.8% and 27.1% at c. 1975 (20 cm) and c. 1991 (14 cm) respectively, after which followed a gradual decline to 14.3% of TOC in the surface sediment. The outflow sediment core also exhibited an increasing trend (11.6-25.0%) from the core bottom to c. 12 cm and a progressive decreasing trend to 19.1% at the core top. The C/N ratios ranged from 16.8 to 23.1. Each sediment core was characterized by stable C/N ratios with a light decrease in the uppermost strata.

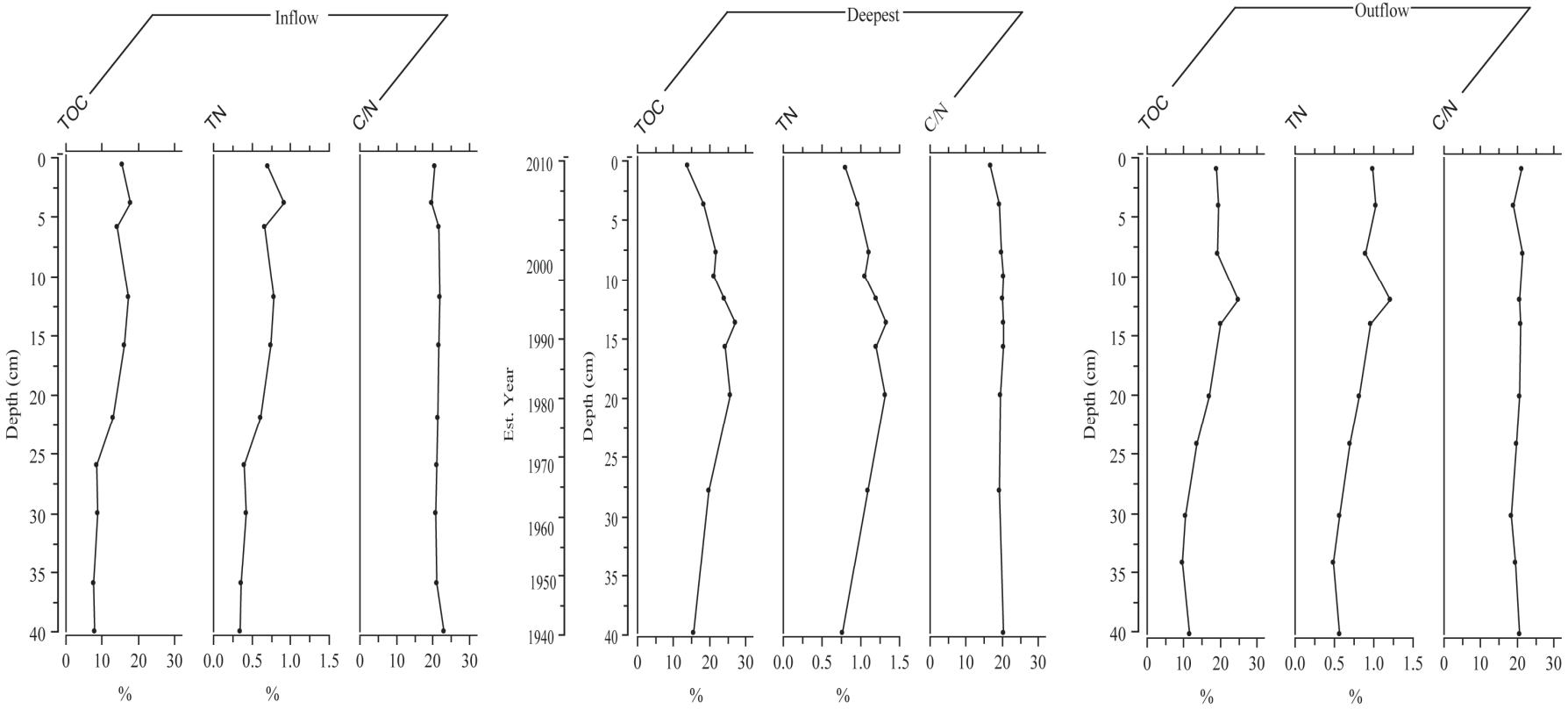

Figure 7.4– TOC (%), TN (%) and C/N ratio in sediment cores from inflow, deepest and outflow sampling location in Feeagh.

The TOC content of the deepest sediment core from Guitane varied between 4.9% and 11.3%, with a mean value of 8.1% (Figure 7.5). TN ranged from 0.4 to 1.4% with an average of 0.8%. TOC and TN did not co-vary in the Guitane core. The TOC content gradually decreased from 11.3% at the core bottom to 4.9% at 17 cm depth, increased to 8.6% at approximately 1856 (10 cm) and maintained levels between 5.0% and 7.7% at the top of the core. TN (%) peaked with 1.3% at 40 cm depth and decreased constantly to a minimum of 0.3% at approximately 1899 (8 cm). The C/N ratio ranged from 7.7 to 16.2 throughout the core. Lowest values were evident (between 11 and 7) from the core bottom (52 cm) up to 17 cm. A distinct change is evident from this point with increasing C/N ratios to a maximum of 16.2 at c. 1880 (10 cm). These higher ratios are maintained to the core top.

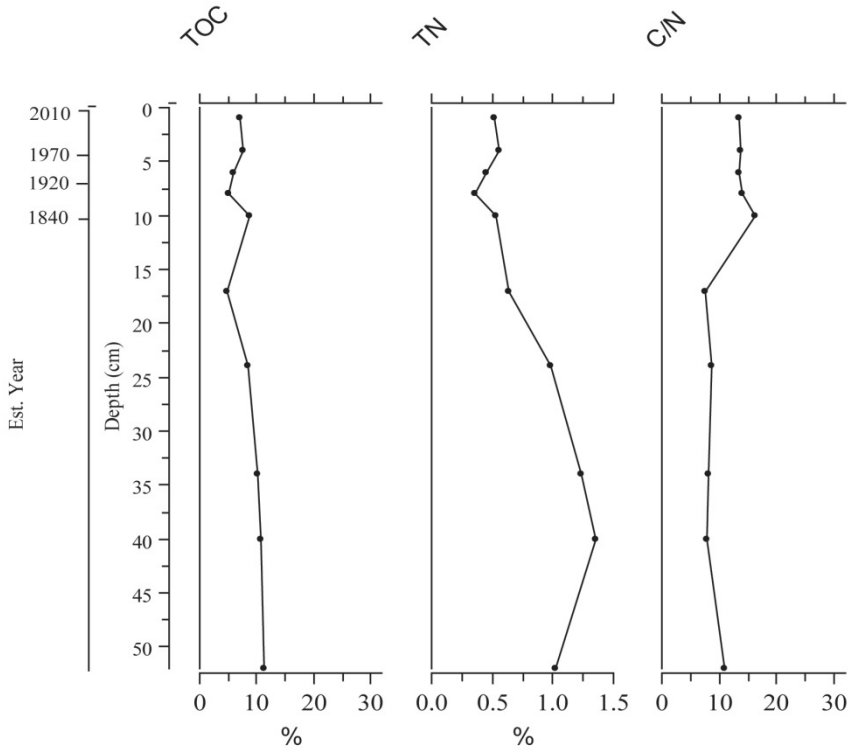

Figure 7.5 - TOC (%), TN (%) and C/N ratio (n=10) in Guitane.

7.6 Biological proxies

7.6.1 Pigments

A total of 15 fossil pigments were identified in the sediment cores from Feeagh and 14 pigments in Guitane (Table 6.2). Pigment data for both lakes are shown in Figures 7.6 – 7.9 and raw data are presented in Appendix Q and Appendix S. In Feeagh the lowest pigment concentrations were found in the outflow (range 29.8 – 127.9 nmol g^{-1}) (Figure

7.8), while highest concentrations were evident in the deepest core (36.0 – 173.2 nmol g^{-1}) (Figure 7.7). This corresponds to the trends expressed in the surficial sediments (Chapter 6). In the inflow sediment core the algal pigment abundance ranged between 48.6 and 132.3 nmol g^{-1} (Figure 7.6). The profiles from the deepest core peaked twice in the 1990s (14 and 10 cm) and peaks were evident in the outflow core (14 cm) and in the inflow core (8 cm and 2 cm). In the deepest core a major trough of 36 nmol g^{-1} at c. 1980 (20 cm depth) was evident. In Guitane total pigment concentration ranged from 35.0 to 323.4 nmol g^{-1}, with minor peaks at 34 and 23 cm depth, followed by a progressive increase to maximum concentrations at the core top (Figure 7.9).

The ratio of labile precursor compounds (chl-*a*) to chemically stable products (pheophytin-*a*) was used to describe the pigment preservation at each site, knowing that high ratios indicate good preservation (Figure 7.6 - Figure 7.9). The degree of pigment preservation varied among the sediment cores and ranged between 0.06 and 0.77 in Feeagh and between 0.13 and 2.45 in Guitane. In Feeagh the ratios were more stable in the inflow sediment core (0.15-0.52) compared to the deepest (0.22-0.77) and outflow (0.06-0.62) sediment cores. The deepest core had a peak of 0.77 at c. 2003 (6 cm), while the outflow core showed two peaks of 0.56 and 0.62 at 20 and 2 cm depth, respectively. In contrast, in Guitane the ratios showed a decreasing trend from 0.49 at the core bottom to 0.13 at 17.5 cm depth followed by an increase in the surface sediments to 2.45. An indication of good pigment preservation conditions throughout the sediment cores is given by the fact that the all pigments remained in relatively stable proportions. This was supported by the presence of labile chlorophylls (e.g. chl-*a*) and carotenoids (e.g. diatoxanthin) throughout the sediment cores. These pigments also have higher concentrations in older strata. More details are given in the following stratigraphic descriptions.

Constrained cluster analysis in CONISS (Grimm, 1987) was performed to facilitate interpretation of pigment stratigraphy and identify zones of major change. The comparison of CONISS with the broken stick model suggested that the deepest and outflow sediment cores can be divided into three distinct zones in Feeagh (Appendix R). No significant zones were identified for the Feeagh inflow core (Appendix R).

The pigment stratigraphic record of the Feeagh inflow sediment core is illustrated in Figure 7.6. Pigments present in all algae and plantae (chl-*a* and its derivation products) dominated the lower (40-34 cm) and central part (22-20 cm) of the core and reached a maximum at 8 cm depth. Pigments belonging to Chlorophyta, Euglenophyta and plantae (chl-*b* and pheophorbide-*b*) were the most dominant pigments in the rest of the core (between 34 and 24 cm depth and from 18 cm up to the core top). Chlorophyta/Cyanobacteria (lutein/zeaxanthin) co-varied with the latter pigments and peaked between 32 and 28 cm depth and 22 and 14 cm depth. Also siliceous algae (fuco- and diatoxanthin) increased in concentrations in this latter part of the core and peaked a second time at c. 2 cm depth. Cryptophyta (alloxanthin) showed an increasing trend from 22 cm to the core top. Low concentrations of Cyanobacteria (canthaxanthin) (0.4-0.9 nmol g^{-1}) were present throughout the core. The UV-absorbing pigment showed a gradual increase upcore with the highest concentrations of 6.6 nmol g^{-1} at 14 cm depth and a gradual decrease to the core top to 0.7 nmol g^{-1}.

The deepest sediment core in Feeagh was divided into three significant algal pigment zones: Zone 1, (c. 1940-1976), Zone 2 (c. 1976-1998) and Zone 3 (c. 1998-2010) (Figure 7.7). Zone 3 was dominated by Chlorophyta, Euglenophyta and plantae (chl-*b* and pheophorbide-*b*) from c. 1940 (40-36 cm) and by Chlorophyta/Cyanobacteria (lutein/zeaxanthin) from c. 1950s to the mid-1970s (34-26 cm). Siliceous algae (in particular diatoxanthin) and Cryptophyta (alloxanthin) did not vary. Zone 2 was characterized by fluctuations with lutein/zeaxanthin together with chl-*b* and pheophorbide-*b* and the UV-absorbing compound successively reaching their highest concentrations. Nearly all the pigments experienced a minor peak at c. 1990 and a major increase at c. 1998 (16 and 10 cm respectively). In Zone 1 chl-*b* and pheophorbide-*b* dominated each sample with the exception of the surface sample in which pigments present in all algae and plantae reached the highest abundance. Chloro-/Cyanophtya pigments did not vary while siliceous algae pigments fuco- and diatoxanthin reached highest concentrations at c. 2002 (6 cm) and 2007 (2 cm). The Cryptophyta pigment (alloxanthin) peaked in the surface sediments (2011).

The three zones identified in the southernmost (outflow) sediment core were Zone 1 (38-22 cm), Zone 2 (22-18 cm) and Zone 3 (18-0 cm) (Figure 7.8). Zone 1 was characterized by a progressive increase of Chloro- and Euglenophyta and plantae related

pigments (chl-*b* and pheophorbide-*b*) and Chlorophyta/Cyanophyta pigments (lutein/zeaxanthin). These were also the most abundant pigments. This zone showed an increase in lutein/zeaxanthin, chl-*b* and pheophorbide-*b* at 30 and 24 cm. Siliceous algae pigments (fuco- and diatoxanthin) and Cryptophyta pigments (alloxanthin) increased at 30 cm and remained constant levels. Zone 2 had the lowest pigment concentrations. In the uppermost zone (Zone-1) Chloro- and Euglenophyta and plantae pigments (chl-*b* and pheophorbide-*b*) reached the highest levels and peaked at 14 cm together with Chloro/Cyano- and Euglenophyta and plantae pigments (lutein/zeaxanthin, chl-*b*, pheophorbide-*b*). Fuco- and diatoxanthin increased slightly at 2 cm, while alloxanthin increased progressively and reached highest concentrations at the core top.

The stratigraphy of the pigment concentrations from Guitane is illustrated in Figure 7.9. No significant zones of change were identified (Appendix R). Pigments present in all algae and plants (chl-*a* and its derivation products) contributed to the highest concentrations in the lower part of the core (52-36 cm). Increases were evident in chl-*b* and pheophytin-*b* (12-30 nmol g^{-1}), lutein/zeaxanthin (5-17 nmol g^{-1}) and canthaxanthin (2-5.5 nmol g^{-1}). The Cryptophyta pigment alloxanthin peaked at c. 36 cm with 8 nmol g^{-1}. All the pigments identified showed a decreasing trend from c. 34 to 28 cm and increased again from c. 24 to 20 cm. A peak of N_2-fixing colonial Cyanobacteria (Aphanizophyll) (40 nmol g^{-1}) was evident at c. 22 cm. The UV-absorbing compound had low concentrations throughout the core and increased only between 24 and 22 cm depth and at c. 1997 (2 cm depth). Algal and plant pigment concentrations were low from c. 1840 to 1985 (12 – 4 cm) but increased progressively over the last decade reaching highest concentration in the surface sediments (2-0 cm depth).

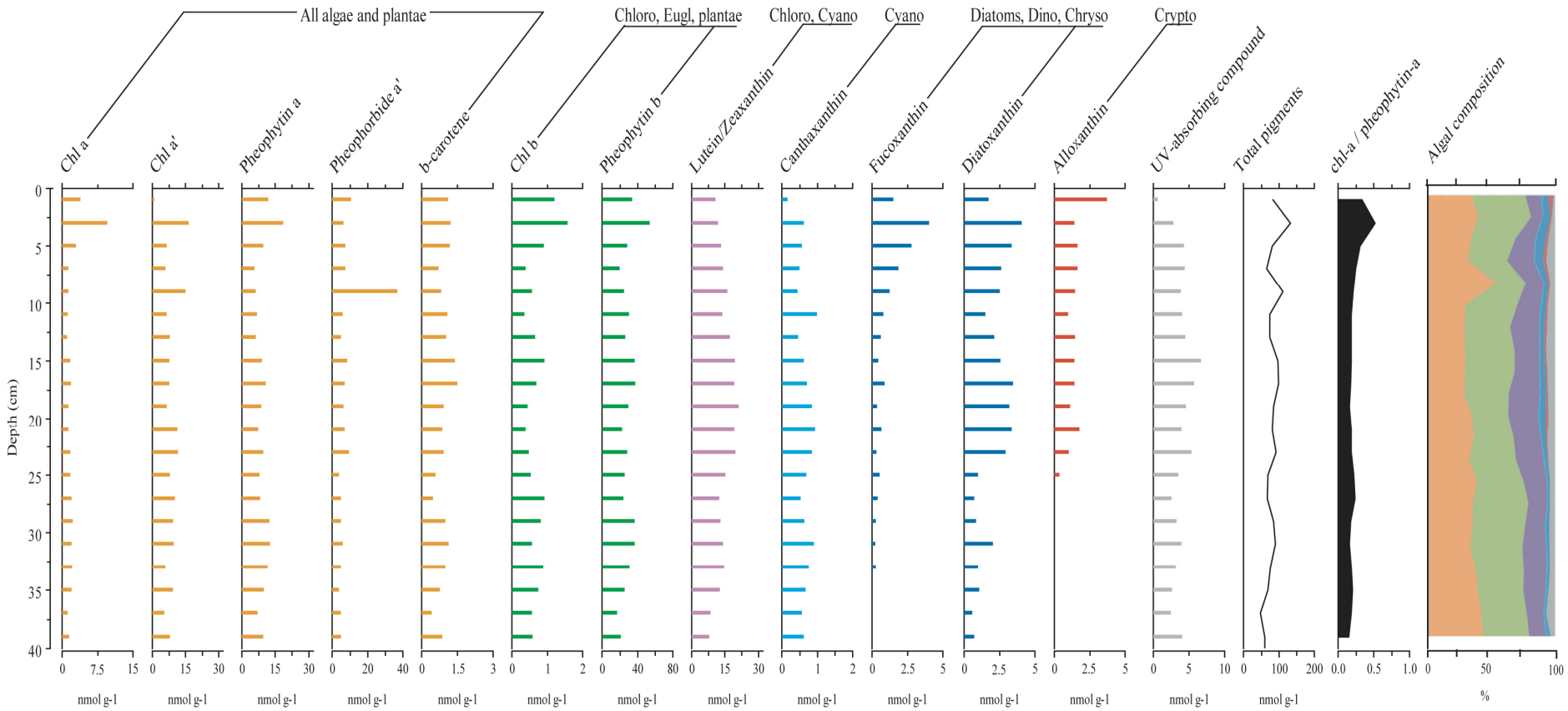

Figure 7.6 - Fossil pigments from the inflow sediment core from Feeagh. Each pigment is expressed as nmol g^{-1} and the algal community composition is estimated as %.

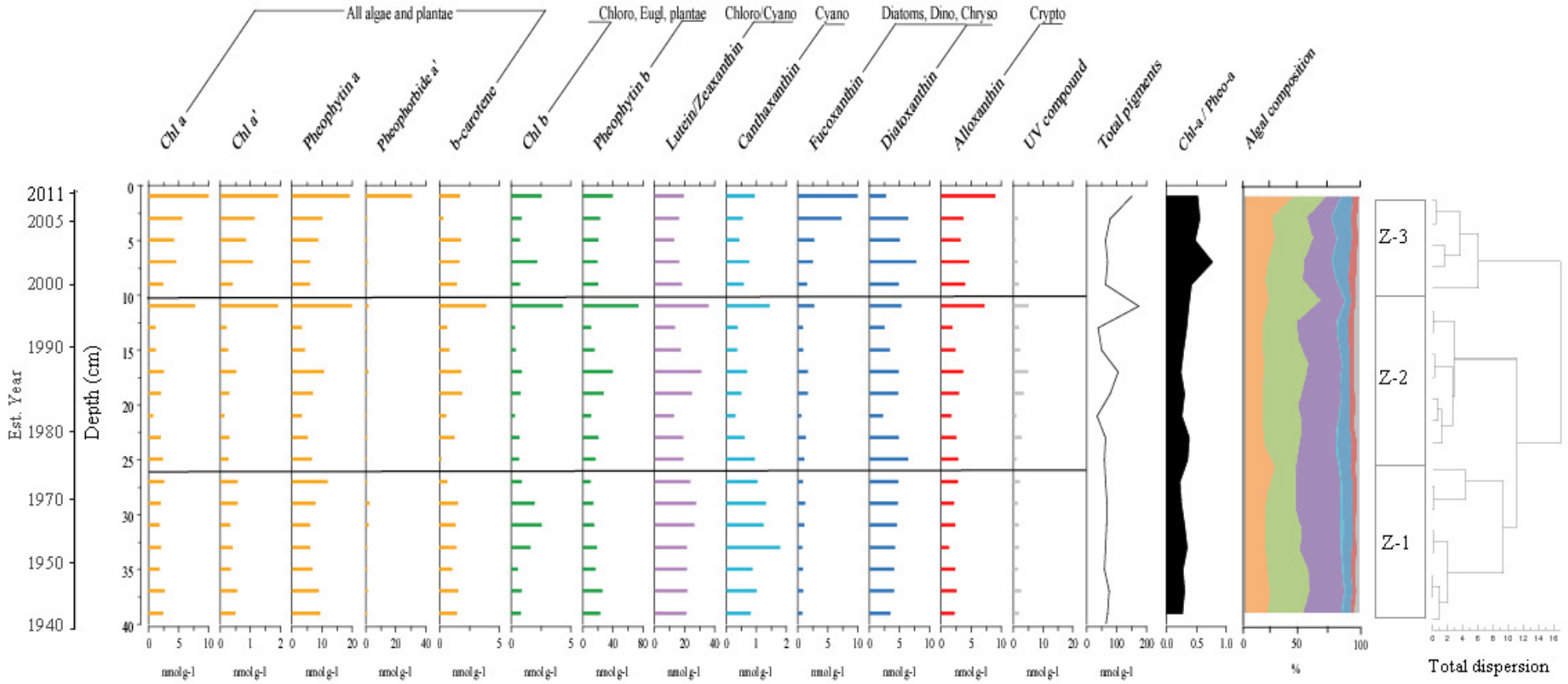

Figure 7.7 - Fossil pigments from the deepest sediment core from Feeagh. Zoning is based on CONISS constrained cluster analysis. Each pigment is expressed as nmol g^{-1} dry weight and the algal community composition is estimated as %.

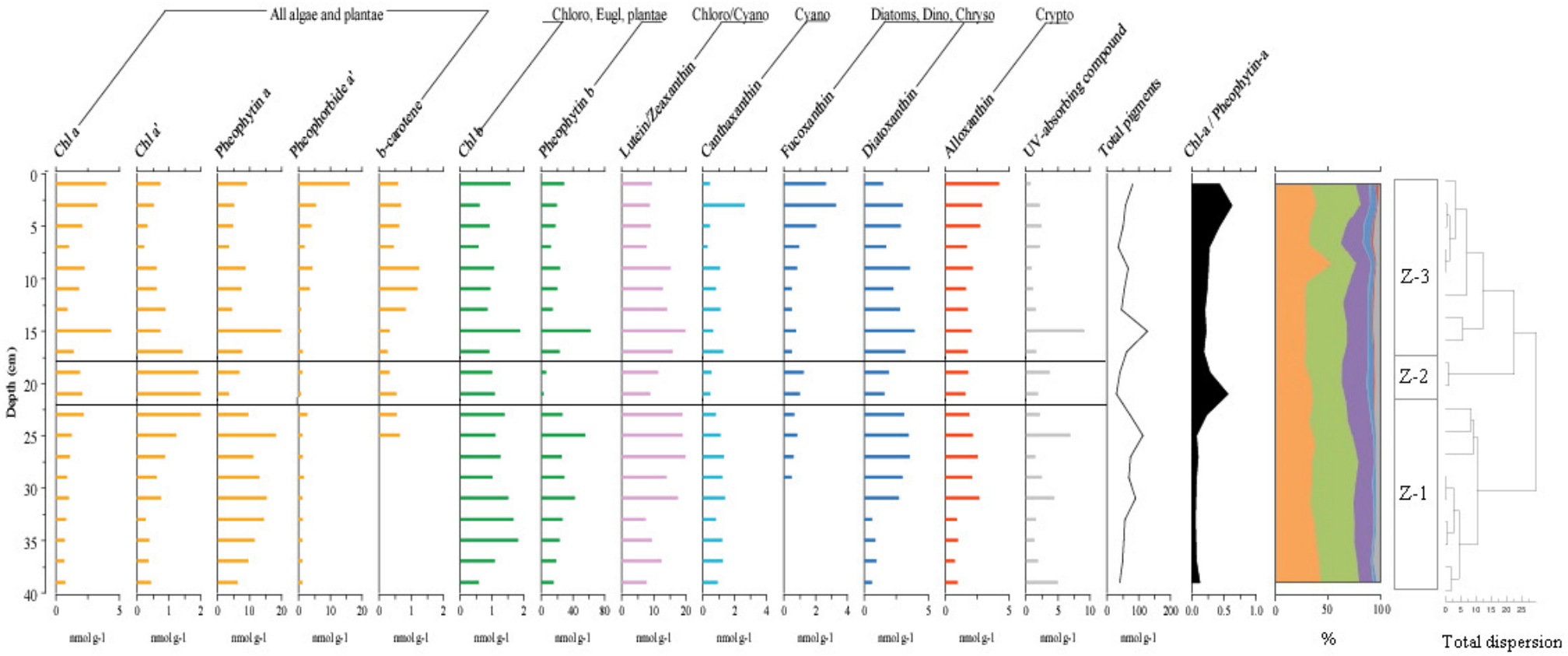

Figure 7.8 – Fossil pigments from the outflow core from Feeagh. Zoning is based on CONISS constrained cluster analysis. Each pigment is expressed as nmol g^{-1} dry weight and the algal community composition is estimated as %.

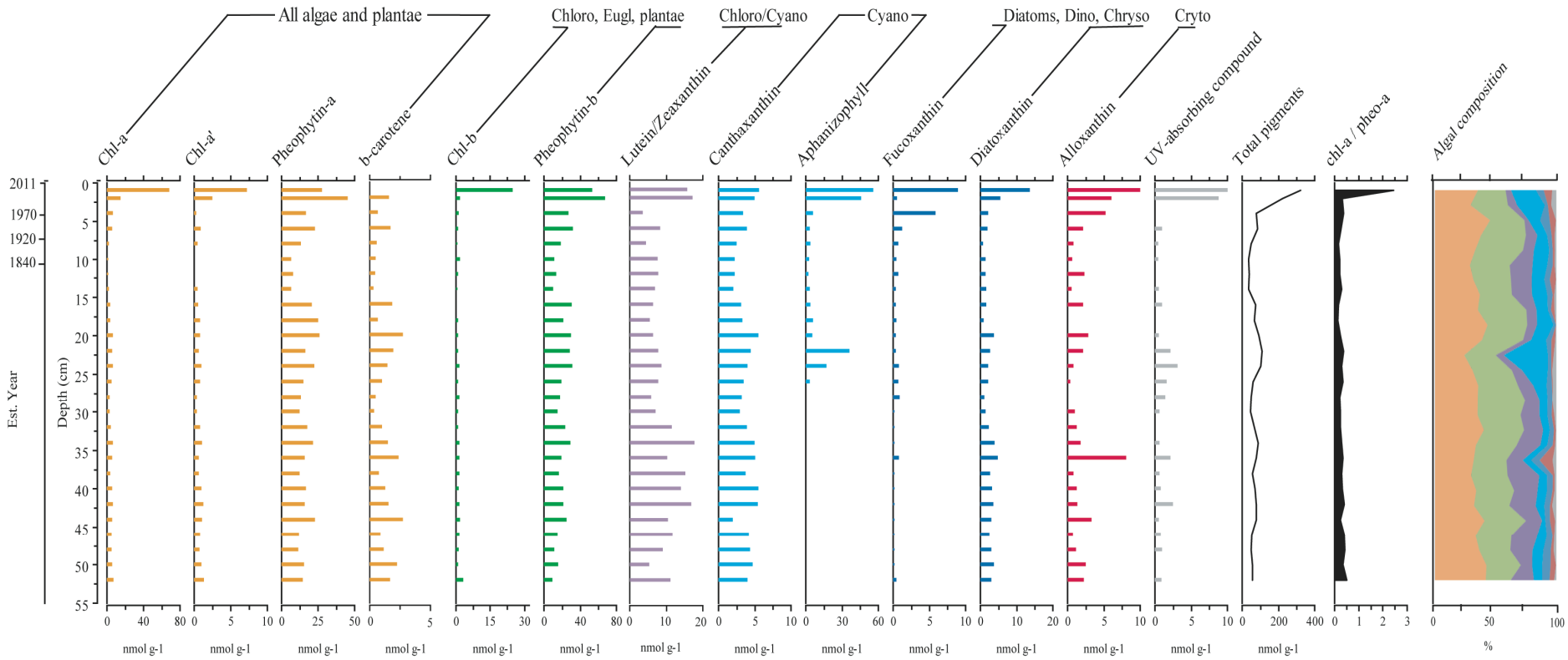

Figure 7.9 – Fossil pigments from the deepest sediment core from Guitane. Each pigment is expressed as nmol g^{-1} dry weight and the algal community composition is estimated as %.

7.6.2 Water clarity index

Water clarity or the UVR index (Leavitt *et al.*, 1997; McGowan *et al.*, 2011) was calculated based on the ratio of the UV-absorbing pigment and carotenoid pigments. A high UVR index indicates high water clarity and therefore low DOC concentrations. In Feeagh this ratio showed a decreasing trend in all cores and was more pronounced in the inflow core (Figure 7.10). The outflow core was characterized by major fluctuations. In Guitane the UVR index was generally lower relative to Feeagh. A peak is evident in the central part of the core (28-24 cm depth) and towards the core top.

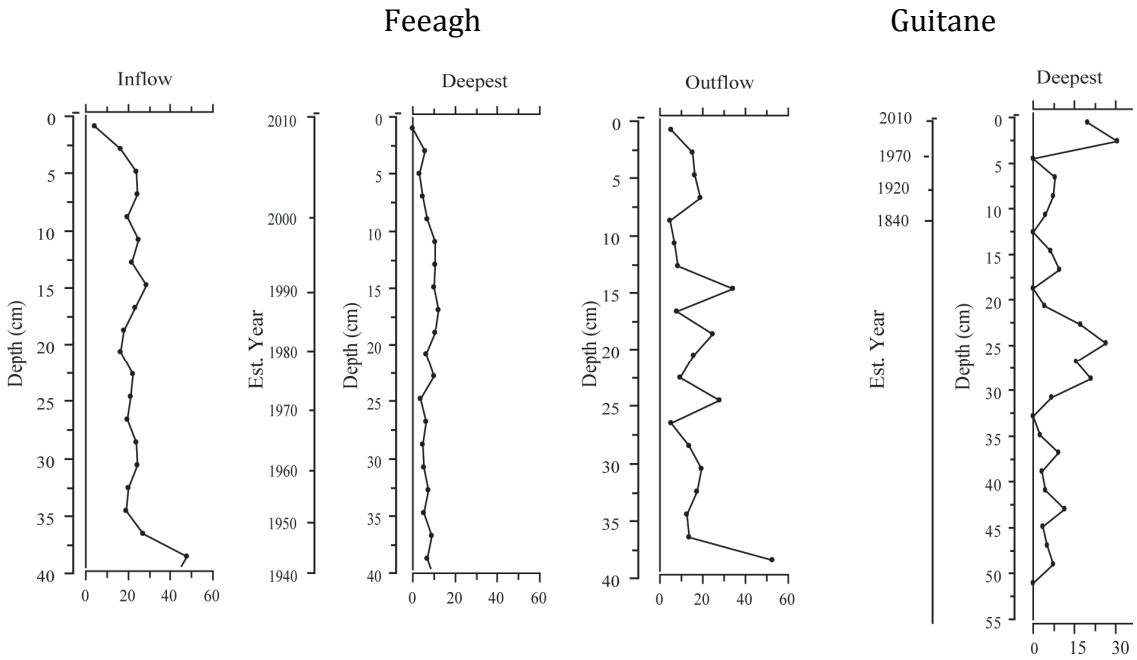

Figure 7.10 – Reconstruction of UVR index from the inflow, deepest and outflow sediment cores collected in Feeagh (on the left) (n=20 for each core) and the deepwater sediment core from Guitane (on the right) (n=27). Estimated chronologies are available for the Feeagh and Guitane deepwater core.

7.6.3 Fossil diatoms

Diatoms from Feeagh were enumerated as part of the ILLUMINATE project (Dalton *et al.*, 2010) from a 60 cm deepwater sediment core, which dated from c. 1890 to 2006. Up-core variations can be summarised into three zones of change: Zone 1 (c. 1880 – 1967) was characterized by oligotrophic species *Achnanthidium minutissimum*, *Cyclotella comensis* and *C. kuetzingiana*: Zone 2 (c. 1967 – 1987) saw increases in nutrient tolerant species *Asterionella formosa*, while *Aulacoseira granulata* and *A. subarctica* increased in Zone 3 (post c. 1987). Sampling as part of the current project

found further increases in *Asterionella formosa* from 4.5% in 2006 to 18.5% in 2010 and declines in *Aulacoseria subarctica* from 14.5% to 6.5%.

A total of 83 diatom taxa were enumerated in 10 samples from the deepest core from Guitane. Higher counting resolution was conducted for the core top as this was the main time period of interest. A full species list and their abundances (%) are given in Appendix U. In order to reduce the effect of counting errors, taxa with a maximum occurrence less than 1% and not found in more than two samples were excluded. This reduced the diatom dataset to 25 taxa. Fossil diatom assemblages throughout the core were mainly dominated by *Cyclotella kuetzingiana* (23.7%), *C. comensis* (22.7%), *Achnanthidium minutissimum* (14.0%), *Tabellaria flocculosa* (4.1%) and *C. radiosa* (3.0%). CONISS cluster analysis (Grimm, 1987) identified nine clusters and the Broken Stick model suggested that there are five statistically significant zones: Zone-1 (52 - 29 cm depth), Zone-2 (29 - 13 cm depth), Zone-3 (pre-c. 1840 to c. 1970), Zone-4 (c. 1970 – c. 1990) and Zone-5 (c. 1990 to 2010). Zone-4 and -5 made part of the same cluster and were considered for this reason within the following description as one single zone. The output of the broken stick model is shown in Appendix V and a summary diatom diagram (diatom abundance > 1%) is illustrated in Figure 7.11. Fossil assemblages at the core bottom (Zone 1 (52-29 cm depth)) are mainly dominated by *Cyclotella kuetzingiana* (25.8%), *C. comensis* (18.5%), *Achnanthidium minutissimum* (12.4%) and *Brachysira garrensis* (4.6%). This zone shows the lowest diatom concentrations (4.4-6.1 valves 10^6 g^{-1}) and sees a decline in *Brachysira garrensis* from 5.8% to 3.8% and increases in *Cyclotella rossii* and *Achnantes oblongella* from 0.7% to 4.7% and from 0.2% to 1.3%, respectively. An increase in diatom concentrations (9.8 - 10.1 valves 10^6 g^{-1}) and a clear dominance of three species, namely *Cyclotella comensis* (37.8%) *Achnanthidium minutissimum* (15.4%) and *Cyclotella kuetzingiana* (11.8%) characterise Zone 2 (29-13 cm depth). *Cyclotella* spp. decrease, while *Achnanthidium minutissimum* increases. Zone 3 (pre-c. 1840 to c. 1970) is characterised by the highest diatom concentrations (average of 9.4 valves 10^6 g^{-1} with a range from 5 to 14.1 valves 10^6 g^{-1}). The most dominant species are *Cyclotella comensis* (25.4%) and *C. kuetzingiana* (19.4%). *Achnanthidium minutissimum* decreases from 19% to 13.9%, while increases in *Tabellaria flocculosa*, *Cyclotella rossii* and *Fragilaria exigua* are evident. Zone 4 and Zone 5 (c. 1970 – c. 1990 and c. 1990 - 2010) exhibit lower diatom concentrations (6.7 valves 10^6 g^{-1} and 2.4 valves 10^6 g^{-1} respectively). The assemblages are dominated

by *Cyclotella kuetzingiana* and *C. comensis* (41.1% and 36.3% respectively). *Asterionella formosa* (10.5%) and *Cyclotella menegheniana* are now evident. The diatom concentrations reach a minimum at the core top with 2.4 valves 10^6 g^{-1}.

The diatom assemblages for Feeagh and Guitane were grouped into benthic and planktonic (including tychoplanktonic taxa such as *Aulacoseira* spp.) forms and expressed as percentages of the total number of valves in each sample (Figure 7.12) (Appendix T and U, respectively). In Feeagh the benthic taxa dominated (range 49.7 - 71.3%), while in Guitane the planktonic taxa prevailed (range 44.2 - 68.2%). In Feeagh the benthic community experienced several minor oscillations from c. 1890 onwards reaching highest percentages of 71% at c. 1942. This was followed by a decreasing trend and constant levels (c. 50%) in the 1970s. The benthic community increased again up to 60% in 2010. The planktonic taxa reached highest percentages in c. 2002 with 47.1%. In Guitane the planktonic forms experienced a progressive up-core increase, while the benthic taxa decreased. The highest contribution of pelagic taxa (68.2%) was observed in c. 1985 (3.5 cm).

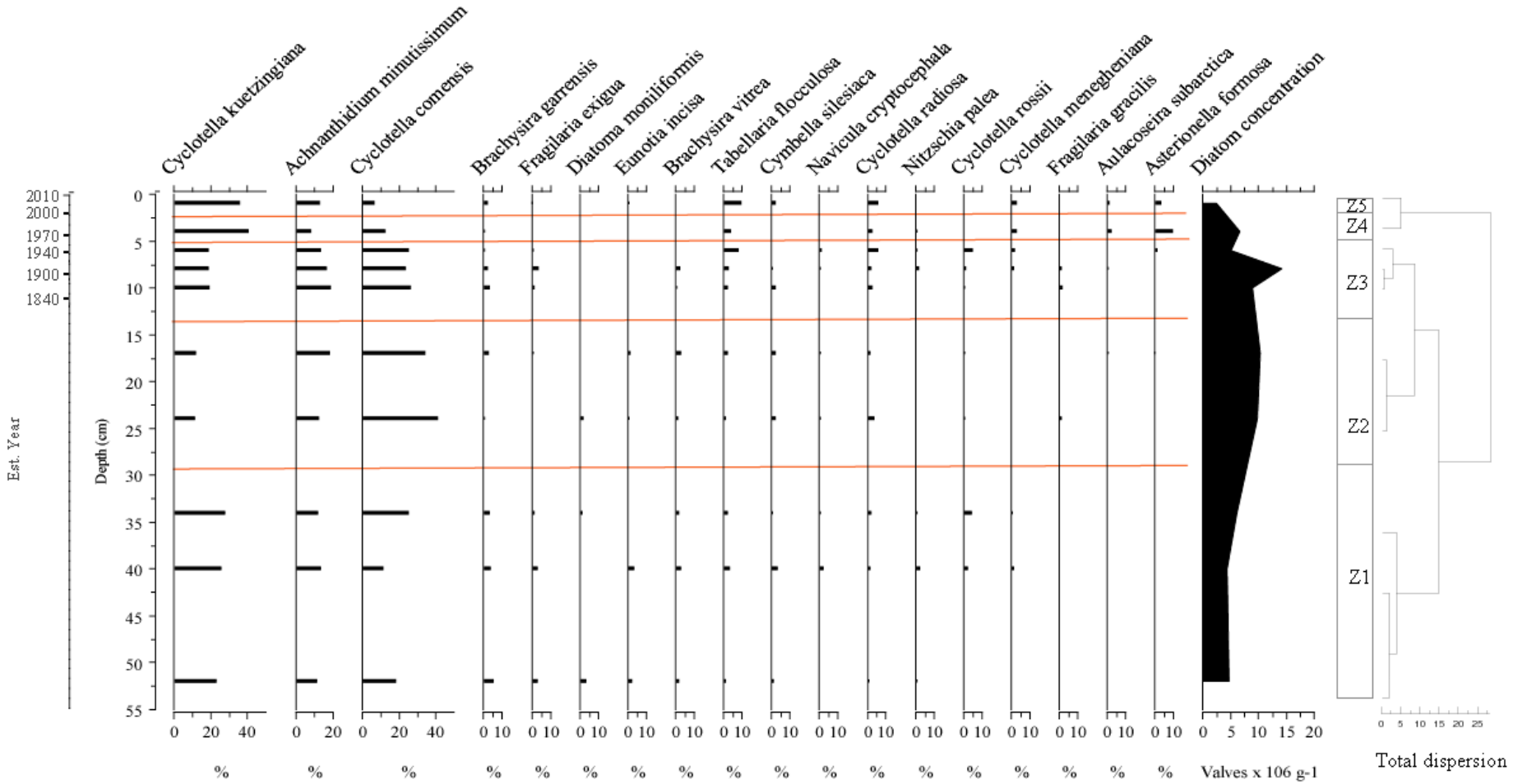

Figure 7.11 – Up-core variations in remains of dominant diatom taxa (abundance > 2%) and diatom concentrations in the sediment core samples from the deepest waters in Guitane. The red lines evidence the five statistically significant zones. CONISS zones are highlighted as red lines.

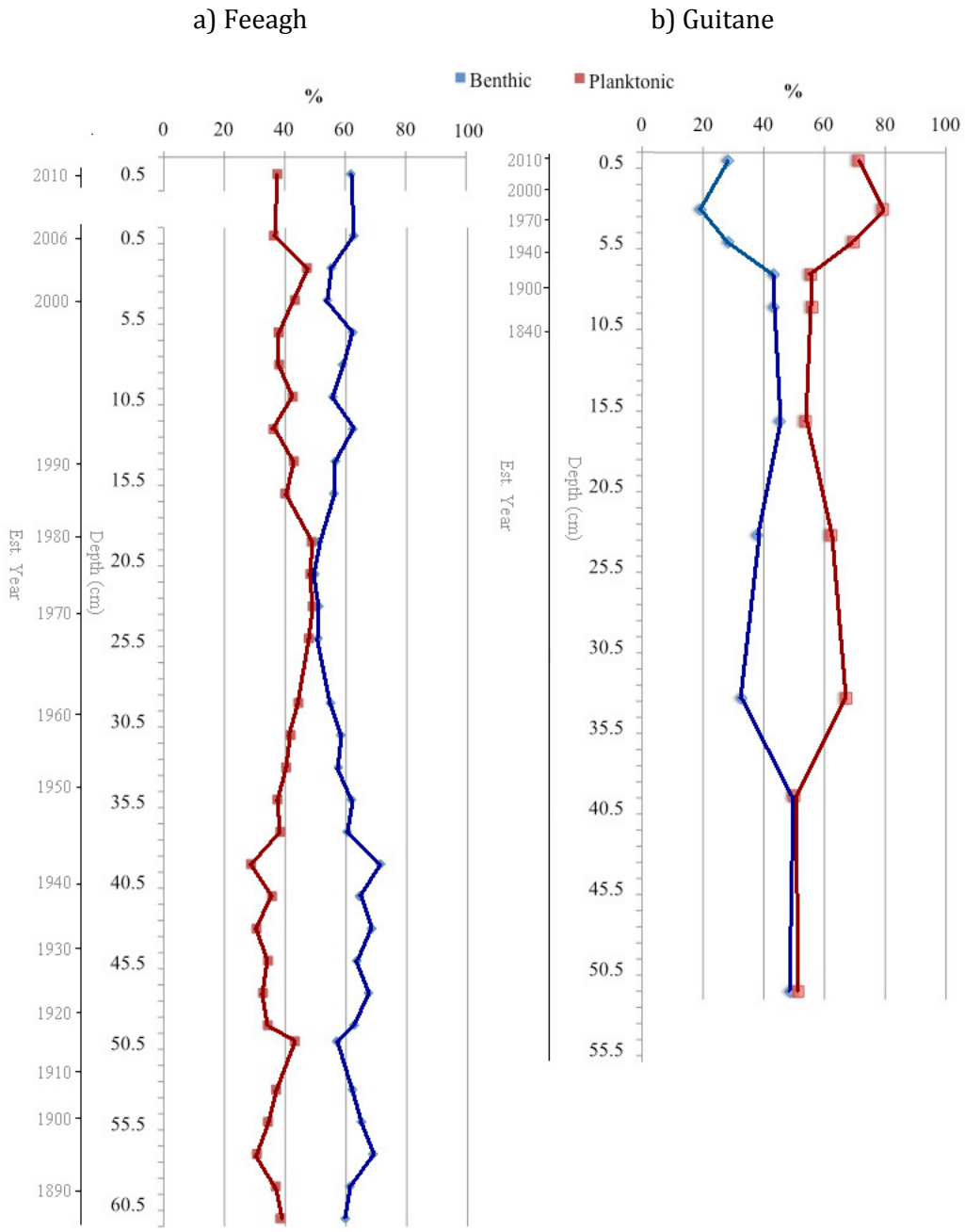

Figure 7.12 – Changes in diatom community structure of benthic (blue line) and planktonic (red line) forms in a) Feeagh (to the left) and b) Guitane (to the right).

7.7 Discussion

The results from sediment core reconstructions are considered here in the context limnological variability and historical catchment changes to explore temporal palaeoecological variations and evaluate potential drivers and stressors of limnological change in both lake basins.

7.7.1 Temporal variations in the sediment record and mechanisms of change in water clarity in Feeagh

The three sediment cores collected from Feeagh revealed spatial and temporal variations in the geochemical and biological proxies and this confirmed the importance of utilizing multiple cores in palaeolimnological studies (Wolfe, 1996; Waters *et al.*, 2005; Reavie & Baratono, 2007). While no chronology was established for the in- and outflow cores in Feeagh, it is clear that the sediment accumulation rates were highest in the inflow core and lowest in the outflow core (see Chapter 6). This has consequently let to variations between the three sediment cores. Firstly, as already observed in sediment trap samples, the organic matter content (TOC and LOI_{550}) was lowest in the inflow sediment core. This can be explained by the higher sedimentation of coarser, heavier mineral and inorganic particles (Lotter & Bigler, 2000; Vogel *et al.*, 2010). Similarly, the C/N ratios were higher in the inflow core and lower in outflow sediments. In each sediment core high C/N ratios (> 17) indicated that the organic matter was derived mainly from terrestrial sources (Meyers & Eadie, 1993; Meyers & Teranes, 2001). Ertel & Hedges (1984) reported C/N values of around 18 from peat and similar values were measured by Lamb *et al.* (2007) and Diefendorf *et al.* (2008). The total fossil pigment concentrations also showed clear spatiotemporal differences. Spatiotemporal variability in deposition processes, which can affect sedimentary pigment concentrations across lake basins, was already recognized by Leavitt & Carpenter (1989) and has been confirmed in the sediments in several lakes (Waters *et al.*, 2005; Brock *et al.*, 2006; McGowan *et al.*, 2011). Feeagh in- and outflow sediment cores were characterized by lower total pigment concentrations, compared to the deepest core. It is presumed that low light intensity and low water temperatures in the deep waters limited photo-degradation enabled arrival and preservation of pigments at the lake bottom (Carpenter *et al.*, 1986; Descy *et al.*, 2000; McGowan, 2007).

Higher levels of the UVR index, calculated as a measure of water clarity, were evident in the in- and outflow cores. More pronounced declines in reconstructed UVR in the inflow core indicated reduced water transparency over time and consequently, a reduction in penetration of UVR in the water column (Leavitt *et al.*, 1997; McGowan *et al.*, 2011). Secchi depth readings collected from the deepest point in Feeagh from 1996 to 2011 also suggest a slight decreasing trend (Marine Institute, unpublished data). While only a few Secchi readings are available between 1996 and 2002, generally

deeper and therefore more transparent waters were recorded (with maximum Secchi depths of 3 m) compared to the period between 2004 and 2010 (maximum Secchi depth of 2.5 m). A further indication of a gradual decline of water clarity is suggested by the up-core increase of alloxanthin, a pigment present in Cryptophyta, in each sediment core. Cryptophyta dominated in the open water samples (see Chapter 5) and these flagellates are known to be tolerant of low light availability and are generally assumed to prefer enriched waters (Reynolds *et al.*, 2002). The reconstruction of benthic and planktonic diatoms showed a reduction of benthic taxa between the 1940s and 1970s. Similarly, Dalton *et al.*, (2010) postulated that a shift from mainly benthic cladocera taxa in the 1960s to planktonic taxa in the 1970s could be indicative of reduced water clarity over time. This may be caused by peat silt deposition in the littoral areas that may have significantly impacted light penetration and therefore, contributing to a decrease in available aquatic macrophyte habitats in the littoral zone, which indirectly influenced also the benthic cladocera population (Duigan & Birks, 2000; Jeppesen *et al.*, 2001; Garrido *et al.*, 2003). More details on biological responses to water clarity changes in the sediment records and the combined available historical data sets together with an evaluation of potential drivers of limnological change are examined within the following paragraphs.

7.7.1.1 Land-use changes

The deepest and outflow sediment cores showed peaks in total pigment concentrations (deepest - c. 2000 and 1990 (10 and 16 cm); outflow - c. 14 and 24 cm). The increases corresponded to a rise in pigments present in all algae and plantae (chl-*a* and -*b*, pheophytin-*a* and -*b*, ß-carotene) and in lutein/zeaxanthin, and can be indicative of an influx of plant material of terrestrial origin and/or an increase in green algae and/or cyanobacteria (Leavitt, 1993; Leavitt & Hodgson, 2001a). Reconstructions of fossil pigments in remote alpine lakes, characterized by low water column chlorophyll concentrations similar to Feeagh, showed that high pigment concentrations are not necessarily representative of "high productivity" conditions (Lami *et al.*, 2000). Further confirmation of terrestrial catchment source for the pigment peaks is given by the low open water chl-*a* concentrations (< 6.9 µg L^{-1}) measured in Feeagh between 1996 and 2010 (Marine Institute, unpublished data). In the catchment the expansion of commercial conifer plantation began in the 1950s and continued until the late 1980s (Allott & Brennan, 1993) (Figure 7.13). Peaks in total pigment concentrations coincide

with clearfelling of conifers in the early 1990s, when approximately 672 hectares (29% of the total plantation area) were removed in the catchment (Whelan *et al.*, 1998). Recently, Rodgers *et al.* (2010a; 2011) confirmed an increased release of phosphorus and a significant rise in suspended solid concentrations to receiving waters in the Burrishoole catchment after harvesting operations commenced. In many other boreal areas increased turbidity (loss of suspended sediment and nutrients) and sedimentation due to harvesting operations, road construction and drainage were observed (Carignan *et al.*, 2000; Nisbet, 2001; Winkler *et al.*, 2009).

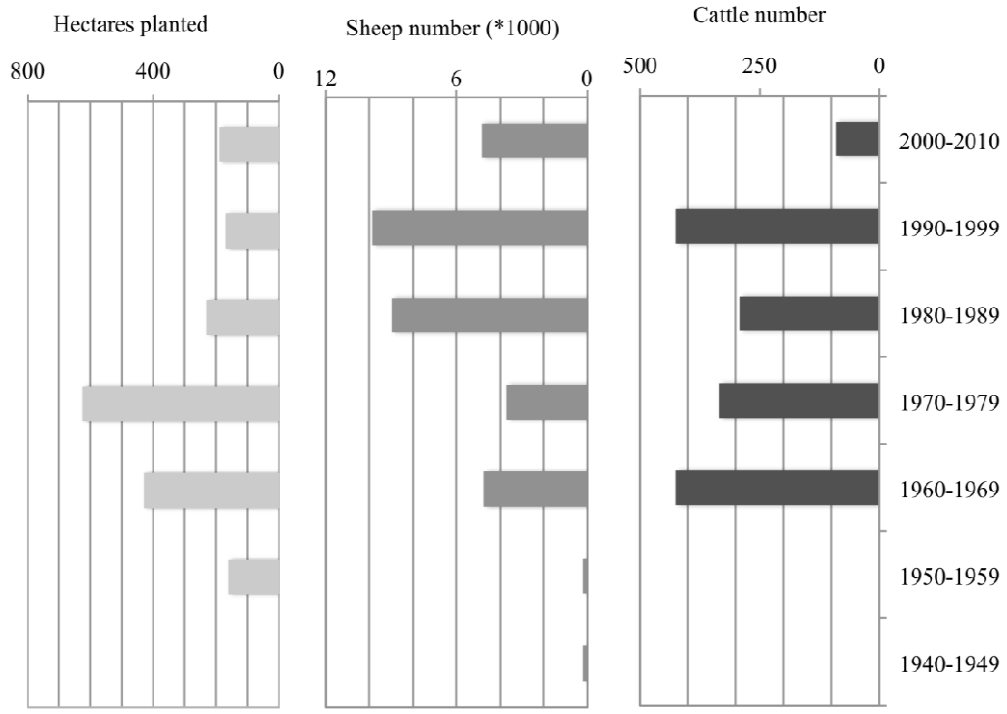

Figure 7.13 – Hectares of planted coniferous forest, numbers of sheep (*1000) and cattle and in the Burrishoole catchment between 1940 and 2010.

Reconstructions of benthic and pelagic diatom taxa were characterized by a gradual decrease of the benthic taxa between the mid 1950s and to the mid 1960s and pre-afforestation. The benthic community increased to 60% in 2010 and assemblages are also characterized by a shift to nutrient tolerant taxa and a rise in diatom-inferred TP (Dalton *et al.,* 2010). Comparable increases in the in-wash of terrestrial matter and decreases in water clarity are evident in mid-Holocene palaeolimnological reconstructions (Pienitz & Vincent, 2000), when DOC concentrations declined with decreasing forest cover in central-northwest Canada, where the highest sustained lake-water TOC concentrations coincided with the period of maximum tree-line advance

and/or highest forest cover density. The vegetation maximum corresponded to the period of abundant periphytic diatom taxa relative to planktonic diatoms. Also other studies detected a shift in the diatom assemblages and connected them with a nutrient pulse, followed by eutrophication, caused by forestry practices (ditching, fertilization, clear cutting, soil preparation by ploughing, harrowing or burning) (Turkia *et al.*, 1998; Köster *et al.*, 2005). In contrast, Rönkkö *et al.* (1988) found only mild responses in benthic diatoms in small forest streams to forest clear cutting and peat bog ditching.

The decrease in water clarity and shifts in algal assemblages could also be related to the severe soil erosion caused by increases in grazing in the uplands of the Burrishoole catchment since the 1960s (Figure 7.13), driven by the low unit return from sheep and headage payments to farmers (CSO, 1991; Weir, 1996; CSO, 2000, 2006, 2011). Cattle numbers in contrast were negligible. Such erosive forces induced the deterioration in the level of the typical vegetation cover of peaty soils, followed by the extensive exposure of the bedrock and a five-fold increase in the amount of peat lost (Salmon Research Agency, 1994). Whelan *et al.* (1998) revealed that in the 1980s and early 1990s, 21% of the Burrishoole catchment was severely overgrazed and characterized by the absence of a vegetative cover. Moreover, only 4% could be classified as intact peatland. A Commission of the European Union announced that Ireland had failed to take the necessary measures to prevent the peat bog of the Owenduff-Nephin Beg Complex Special Protection Area from being damaged by overgrazing (Edwards, 2003). Re-assessment of certain areas in 2004/2005 established that the situation had not improved in the intervening years (National Parks and Wildlife Service, 2006). In May 2011 the amendment of the Commonage Framework Plans and agri-environmental schemes expired (Marine Institute, pers. comm.) again and no particular regulations were introduced.

7.7.1.2 Climate change

Between 1960 and 2009 increases in air temperature (by 1.48°C) and in the frequency and intensity of extreme precipitation in winter (of 3.3 events) and annually (7.5 events) were found in the Burrishoole catchment (Fealy *et al.*, 2010). Similarly, the annual precipitation has increased in some areas of the northern Baltic Sea area during recent years (Arvola *et al.*, 2004). These recent climatic changes have been partly attributed to changes in the NAO index (Jennings *et al.*, 2000). Between 1970 and 1990 the

prevalence of more positive winter NAO index values was associated with increased air temperatures and rainfall amounts together with higher wind speed, increased relative humidity and cloud cover in the west of Ireland. This could have influenced surface waters and the algal community. Such responses to recent climate change have been explored over long-time scales in the UK (George & Taylor, 1995; Davies *et al.*, 1998; George *et al.*, 2004; McGowan *et al.*, 2011). Similarly, the wetter conditions in early spring were associated with lower abundances of siliceous algae, which were attributed to the NAO (Davies *et al.*, 1998).

Fealy *et al.* (2010) point out that in Burrishoole increase in hot-temperatures and decreases in cold-temperatures, together with an increase in the frequency and intensity of extreme precipitation in winter and annually was found over the last five decades. Byrne (2003) documents an extreme precipitation event in Burrishoole in June 1980 after two dry months. The palaeolimnological response in fossil pigment concentrations shows several troughs and peaks in the deepest and outflow cores during the same period. A slight increase in TOC, LOI_{550} and TN in the deepest water core around this time (20 cm depth), could potentially indicating an in-wash of terrestrial suspended solids, that reduced the natural levels of UVR and thus, resulting in declines in abundance of several algal groups. Moreover, an accumulation in Al, K and Fe was measured in the same period the deepest sediment core (Dalton *et al.*, 2010). The diatom assemblage experienced a decrease in various taxa, including *Asterionella formosa* and *Fragilaria capucina* var. *rumpens* and an increase in *Tabellaria flocculosa, Achnantidium minutissimum* and *Cyclotella kuetzingiana* (Dalton *et al.*, 2010).

7.7.2 Palaeolimnological variations in Guitane

The sediment material in Guitane differs from Feeagh in terms of organic load, pigment concentration and different composition and source of primary producers in the lake and its catchment as well as diverse diatom assemblages. The C/N ratio at the base of the deepwater core from Guitane indicated that the sedimentary organic matter was predominantly autochthonous (C/N ratio < 10) (Meyers, 1994; Meyers & Lallier-Vergès, 1999) and not allochthonous, as in Feeagh. The sediment core revealed a rise of TOC and LOI_{550} in the 18^{th} century. Consequently, the correspondent increase in the C/N ratio from c. 1880 may suggest a shift from autochthonous organic matter to a sub-equal mixture of algal and terrestrial-derived organic matter content (C/N ratio = 12 -

13) (Meyers & Lallier-Vergès, 1999; Meyers & Teranes, 2001; Meyers, 2003). This change potentially caused the decreased UVR index, indicating low water clarity and is coincident with a decline in total pigment concentrations.

Several periphytic diatom species associated with benthic substrates were present in the lowermost part of the sediment core, indicating an important component of benthic primary productivity. No major change in the diatom assemblages occurred before c. 1840 (11.5 cm depth) suggesting relatively stable conditions, although higher resolution data for this period is required to assess the degree of stability more fully. Until c. 1840 the diatom assemblages present in the sediment samples were dominated by typical acidophilous-circumneutral and oligotrophic taxa: *Cyclotella* spp., *Tabellaria flocculosa* and *Achnanthidium minutissimum.* There was then a slight change to planktonic mesotrophic taxa (*Asterionella formosa* and *Aulacoseira subarctica*) up to c. 1980 that decreased again in the surface samples. The genus *Cyclotella* has been associated with oligohumic with low DOC content (colour < 30 mg Pt L^{-1}) (Miettinen *et al.*, 2005). Additionally, deep low productivity Scottish Lochs, were dominated by *Cyclotella* spp. and *Achnanthidium minutissimum* (Bennion *et al.*, 2004). Marked species shifts to planktonic assemblages (e.g. *Asterionella formosa*, *Aulacoseira surbarctica* and *Fragilaria crotonensis*) were found to be indicative of nutrient enrichment (Jones *et al.*, 1997; Bennion *et al.*, 2004). Over the last four decades Guitane was characterized by a gradual increase in total pigment concentrations indicating a rise in algal productivity. A rise in pigments from cyanobacteria, Crypto-, Chloro-, Euglenophtya, plantae and siliceous algae was observed in the deposited sediment strata. An increase in preservation index (chl-*a* / pheophytin-*a*) at this time indicated improved pigment preservation as is often observed when algal production increases (Leavitt, 1993). Monitoring data measured between 1999 and 2007 provided from Kerry County Council (unpublished data), show stable average annual TP concentrations of 10-12 µg L^{-1} with occasional peaks of 18 µg L^{-1}, indicating oligo-mesotrophic conditions (OECD, 1982). The TP values decreased again to 9 µg L^{-1} over the last four years. Algal blooms have been observed over the summer months in the recent years (EPA, 2003; KCC, pers. comments). A detailed survey on phytoplankton carried out between 1999 and 2000 confirmed the presence of blue-green algae (dominated by *Oscillatoria agardhii* and followed by *Aphanocapsa* sp., *Aphanthece* sp., *Coelospherium kuetzingiana*, *C.*

naegeliana, *Merismopedia* sp. and *Oscillatoria limnetica*) (Allott *et al.*, 2001). Cyanobacteria contributed between 25% and 50% of the phytoplankton biomass in July 2000 and November 2000. More details on potential drivers are given within the following paragraphs putting emphasis on the more recent decades.

7.7.2.1 Land-use and lake-use changes

Key drivers of change in the Guitane catchment in the last few decades include livestock numbers and climate change. Native and commercial forestry can be elimated as influential factors for most of the period of interest. McCracken (1959) reported that much of Kerry was forested in c. 1600, however pollen records confirmed that the area was denuded of oak, birch and arbutus to fuel the ironworks during the 17th century (Mitchell, 1988, 1990). No commercial afforestation has taken place over the last century within the catchment. Moreover, agricultural activity in the catchment has been relatively restricted in the catchment in recent decades. Sheep numbers increased progressively from the 1960s to the 1980s and peaked in the 1990s in the Flesk river catchment (Figure 7.14). The cattle number was highest in the 1960s and decreased over the following decades (data from CSO, 1960; 1970; 1980; 1990; 2000; 2010). Fertilization of catchment fields potentially contributed to a rise in nutrient in-wash, and thus in an increase algal pigments and mesotrophic diatoms (*Asterionella formosa* and *Aulacoseira subarctica*) in the sediment record. Jennings & Allott (2006) described a rise in winter NO_3-N in Lough Leane (c. 10 km downstream from Guitane), from below 150 µg L^{-1} in the 1970s to levels higher than 400 µg L^{-1} in the late 1990s, tracking a parallel increase in fertilizer sales over the same period. In many other freshwater systems was observed similarly an upward trend in NO_3-N concentrations in recent decades, generally attributable to the use of nitrogen fertilizers in agricultural catchments (Vitousek *et al.*, 1997; de Klein & Monaghan, 2011). The amount of pasture in the catchment has been reduced since the mid 1990s as a result of Rural Environmental Protection Scheme (REPS) (Emerson & Gillmor, 1999) and the subsequent introduction of the Commonage Framework Plan in 1998 (Irvine *et al.*, 2007; EIS, 2009). The recovery to oligotrophic conditions could be have been favored by REPS restrictions. The precautionary principle adopted within the catchment area of the lake prohibited any form of development (EIS, 2009), enabling improvements in the water quality.

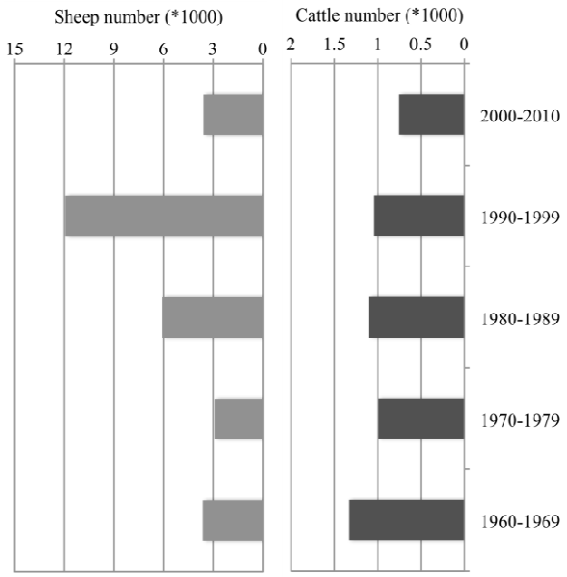

Figure 7.14 - Sheep and cattle number (both *1000) in Flesk DED between 1960 and 2010.

7.7.2.2 Climate change

A further possible driver of changes in lake productivity over time could be attributed to the NAO together with the Gulf Stream. A detailed climatic investigation of the annual precipitation between 1940 and 1993, measured on the south-west coast of Ireland, revealed an increase in mean annual precipitation since 1975 (Kiely *et al.*, 1998). Moreover, recent studies show that between 1970 and 2000 climate in southwestern Ireland was influenced not only by a positive NAO, but also to a lesser extent by the Gulf Stream, which contributes to warmer and sunnier weather, with less wind, lower cloud cover and less rainfall in late spring and summer (April-June) (Jennings & Allott, 2006). This dictates the extent of soil moisture deficit (defined as the rate which evapotranspiration exceeds the rate of rainfall (Scholefield *et al.*, 1993)) and, consequently the degree of macronutrient loss to surface waters in the following months (Betton *et al.*, 1991; Scholefield *et al.*, 1993; Reynolds & Edwards, 1995). Since 2001 the NAO has remained in a negative/neutral phase (Bates, 2011). During the latter phase the Gulf Stream shifted again southwards causing colder winters with higher levels of precipitation in Northern Europe. This indicates that over the last ten years a return to more oligotrophic conditions in Guitane could also be potentially related to a shift in NAO.

7.8 Conclusion

Both Feeagh and Guitane are characterized by contrasting water column and sediment trap responses and consequently their sediment core responses are different. Divergent levels of DOC in the two lakes contribute to different algal community structures and thus fossil assemblages. Carbon outflow from peatlands is highly dependent on the catchment morphometry, causing spatiotemporal variations in deposition in the sediment. The production / decomposition balance of the acrotelm, and thus the export of TOC from peat bogs, is linked to climate and the extent of human activity in the catchment. The aquatic ecosystem response to nutrient and carbon enrichment causes variations in the autotrophic communities and consequently, the sediment record.

Chapter 8 – Final conclusions

This neo- and palaeolimnological examination of both study sites investigated within this thesis clearly shows that allochthonous inputs from peaty catchments have major implications for biological and biogeochemical processes in oligotrophic aquatic systems. Lake trophic state and their pelagic auto- and heterotrophic assemblages through time and space were described in detail. Their response to climate variability and catchment condition has been evaluated. The purpose of this final chapter is to summarise how this thesis has: *i*) furthered knowledge of humic and clear water lakes; *ii*) contributed to lake classification; *iii*) helped refine knowledge of the consequences in the treatment of drinking water supplies. Finally, key contributions of the current research are listed.

i) Contribution to knowledge

There are 12,206 freshwater lakes in Ireland (Irvine *et al.*, 2007) and approximately one fifth (18.5%) of the land-cover is made up by peaty soils with thresholds of at least 25% organic matter (Montanarella *et al.*, 2006). Lakes in Ireland are primarily sited in the west, northwest and central lowlands, where extensive peat soils are present. This suggests that a significant number of lakes potentially have high loads of allochtonous matter in their water columns, which significantly influence lake ecology and water quality. For example, of the 197 Irish lakes monitored between 2001 and 2002 (Free *et al.*, 2006) a total of 93 and 90 were characterized as oligohumic (< 30 PtCo mg L^{-1}) and humic lakes (30-90 PtCo mg L^{-1}) respectively, if the classification formulated by Pilke *et al.* (2002) is applied. A further 14 lakes had water colour higher than 90 PtCo mg L^{-1} and seven lakes exceeded 120 PtCo mg L^{-1} and could be considered dystrophic (Lepistö *et al.*, 2006). The results from this research suggest that a significant proportion of Irish lakes have light restrictions caused by humic compounds, which limit the response of primary producers, while the supply of suspended solids stimulates mixotrophic flagellates and heterotrophic bacteria. Consequently, lakes in Ireland together with other boreal lakes can be considered heterotrophic rather than autotrophic lakes, and thus sources of carbon (Cole *et al.*, 1994; Algesten *et al.*, 2003; Sobek *et al.*, 2003). del

Giorgio & Peters (1994) describe an association between lake trophy and net metabolic balance and suggested that oligotrophic lakes are dominated by heterotrophic biomass, presumably supported by allochthonous inputs of carbon. While no measurements of community respiration in the euphotic zone were made within this study, estimations of the abundance of bacteria and mixotrophic phytoplankton taxa that have the capacity to utilise allochthonous matter gave an indirect indication of the extent of pelagic mineralization. Results from humic lakes suggest that inputs of allochthonous organic matter are crucial to the bacterial community. Bacteria are primarily stimulated by the input of terrestrial suspended solids. This suggests that community respiration can periodically exceed phytoplankton photosynthesis (e.g. after the flash-flood and over the winter months). In contrast, bacterial abundance seems to be less pronounced in the clear water lake. The clear-water lake has a lower allochthonous organic carbon loading and is characterized by a more extended photic depth and higher primary production. The lower bacterial abundance found in the clear-water lake and potentially lower rates of mineralization and atmospheric carbon emissions means that it could be considered a carbon source rather than a carbon sink. In boreal landscapes therefore, lakes play a fundamental role in carbon cycling and cannot be ignored when assessing the importance of ecosystems as sinks or sources of carbon.

IPCC (2007) states that in northern Europe the frequency and magnitude of precipitation are very likely to increase due to climate change, and thus the future scenario for those lakes will be a greater influx of terrestrial carbon especially in forested peaty catchments. This will fuel and enhance heterotrophic responses. Projected future climate data for Burrishoole catchment include an increase in air temperature in all seasons (Jennings *et al.*, 2010) and the greatest warming is expected to be experienced in the autumn and spring seasons by the 2080s (Fealy *et al.*, 2010). Both models predict distinct seasonal precipitation regimes with increased rainfall events during the winter and reductions during the summer and early autumn. Moreover, the frequency of extreme flow parameters (low and high events) will severely affect stream flow within the catchment (Fealy *et al.*, 2010). Extreme climatic events are thought to be possible drivers for the exodus of peatland carbon to surrounding rivers, lakes and oceans (Freeman *et al.*, 2001a; Milly *et al.*, 2002). Immediate lake responses were found following heavy rainfall events within the study lakes. The flash-flood event in July 2009 in Mayo and the prolonged precipitation

period in November 2009 in Kerry are examples of climate change events (Fealy *et al.*, 2010). The effects of the flash-flood event were clearly visible in the pelagic communities (Chapter 5) and in the trap accumulation of sediment and algal pigments (Chapter 6). The high sedimentation rate within the sediment traps in the humic lake confirmed that lake sediments are important carbon stores. Arvola *et al.* (2002) showed that Finnish lake sediments are the third largest carbon store after peatland and forest soils. The prolonged precipitation period in November 2009 represented an additional example of extreme precipitation event. However, this latter event did not influence the primary producers in the clear water lake as the growing season was already over. Low sediment deposition rates over the whole collecting period indicated that Guitane is a poorer sink for carbon even though it is embedded within a peaty catchment. The influx of terrestrial material, following the flood events did not show any particular increase or change in the water column traps or surface sediments. However it must be noted that the longer sediment trap sampling interval and the collection of sediment cores one year after the flood event and the relatively coarse sub-sampling interval (1 cm), may have precluded identification of the recent event in the surface sediment strata (Chapter 7).

Land use practices can be inferred from lake sediment responses or palaeolimnological reconstructions. Fossil algal pigments and diatoms were used as key indicators because of their sensitivity to water quality. The palaeolimnological investigation and from Dalton *et al*. (2010) showed that increased conversion of a blanket peat catchment to coniferous forests and overgrazing by sheep together with climatic influences induced erosion, a rise in nutrient concentrations and decreased the depth of the euphotic zone. Similar scenarios and severe erosion of upland blanket peat were observed in other parts of Britain and Ireland (Bradshaw & McGee, 1988; Evans & Warburton, 2007; McHugh, 2007). In general, forests are known to be crucial determinants of water supply, quality and quantity (Bates *et al.*, 2008; Robinson, 2008). Ireland and the United Kingdom are known to be the countries with the lowest forest cover in Europe, however huge areas were reforested over the last five decades. Ireland aims to increase national woodland cover (mainly conifers) from 8% to 17% by 2030 with an afforestation target of 20,000 ha per annum (Department of Agriculture Food and Forestry, 1996; EPA, 2006). In the future a more targeted interaction between forest management (e.g. timber harvesting and reforestation operations) and aquatic environments is essential to develop environmentally compatible and sustainable ecosystems to ensure good ecological

status. For example, Scoles *et al.* (1996) found that in forests where no specific erosion control was applied, annual soil losses were significantly higher on harvested and clear felled sites than on selectively harvested and control sites. In general, over the last three decades in Ireland and Britain the clearcutting silvicultural system has been used exclusively (Hendrick, 2004). This involves clearfelling all the stand and subsequent reforestation. The sudden change that this practice brings about in the landscape has increasingly been criticized (Hart, 1995). In many other parts of Europe continuous cover forestry (forest canopy is maintained at one or more levels without clearfelling) has been used for centuries (Forestry Commission, 2011). Only recently Coillte (the largest Irish commercial company operating in forestry, land based businesses and renewable energy) has formulated a policy of sustainable forest management and has started to maintain continuous cover forestry in approximately 1,000 ha of conifer plantations (Hendrick, 2004). This change from clear-cutting to continuous cover forestry will have implications for a wide range of issues including tree growth, harvesting, economics, amenity, landscape, recreation and consequently nutrient turnover and water quality (inflow of nutrient and allochthonous carbon) of catchment rivers and lakes. It is debatable if an increase of the forest cover will be positive for water quality and the requirements of the WFD to maintain or achieve good quality by 2015 (Solimini, 2006). (Allott *et al.*, 1997; Nisbet, 2001)

i) Lake typology

In the literature several approaches to lake classification are utilised. For example, using the classification system adopted by the Irish EPA the two study sites are classified as Typology class 4 lakes. This class groups together low alkalinity (< 20 mg L^{-1} $CaCO_3$), deep (average depth > 4 m and maximum depth > 12 m) and large (lake area > 50 ha) lakes (EPA, 2006). According to the OECD (1982) and the modified Irish EPA classification scheme (Toner *et al.*, 2005) an oligotrophic trophic status is confirmed for both Feeagh and Guitane if average and annual maximum chl-*a* and TP concentrations are considered. In contrast, the annual average and minimum Secchi disk transparency suggest eutrophic (mean 1.7 m and minimum 0.8 m) conditions for Feeagh and meso-oligotrophic (mean 5 m and minimum 4.4 m) for Guitane (OECD, 1982). Application of Nürnberg's scheme (Nürnberg, 1996) suggests both oligo- and eutrophic conditions in Feeagh (due to average summer shallow water transparency) and oligotrophic status in Guitane. Several authors have recognized that trophic classification based on Secchi

depth alone is likely to be unreliable in coloured lakes, where the lack of transparency is primarily attributable to the brown colouration (Caffrey *et al.*, 1999; Clenaghan *et al.*, 2005; George, 2010a), rather than an abundance of phytoplankton (Taylor *et al.*, 2006). If the most recently formulated lake typology classification scheme for the WFD (Poikane, 2009) is applied, Feeagh and Guitane fall into separate lake types within the Northern Geographical Intercalibration Group: *LN3a* (lowland (< 200 m), shallow (3-15m), low alkalinity (< 0.2 meq L^{-1}), humic (colour 30-90 mg Pt L^{-1}) and *LN3b* (lowland (< 200 m), mean depth (>15m), low alkalinity (< 0.2 meq L^{-1}), clear (colour < 30 mg Pt L^{-1}), respectively. This recent WFD typology system recognized and included water colour as a proxy for organic peat content (Poikane, 2009). The data shown in this thesis highlights that water colour represents a fundamental parameter that can provide a better, and in a certain sense a more adequate assessment of lakes.

iii) Drinking water

Many parts of Northern Europe have had serious difficulties in providing and treating an adequate drinking water supply over the last few decades (Rodriguez & Serodes, 2001a; Löfgren *et al.*, 2003; Sharp *et al.*, 2006). Problems have been evident with appearance, taste and smell as well as serious human health issues. Excessive abstraction from lakes and reservoirs can also impact negatively on the open water ecosystem (e.g. cyanobacterial blooms, decrease in biodiversity such as for example on fish populations (Manley *et al.*, 2008)) and on its marginal habitats (e.g. wetland and heath land) (Muñoz-Reinoso, 2001)). Drinking water providers additionally face challenges in terms of variation in DOM (or TSS and water colour) that can vary seasonally, and can also get in-washed from the surrounding catchment from diffuse or point sources.

Water quality issues have mainly centered on the presence of toxic cyanobacteria and bacterial contamination. The enumeration and estimation of algal biomass in both study sites did not reveal the presence of cyanobacteria capable of producing toxins (e.g. *Microcystis*, *Cylindrospermopsis*). In general, cyanobacteria reached higher levels in terms of abundance and biomass in the clearwater compared to the humic lake. The permanent windy conditions and relatively mild temperatures appear to preclude the formation of problematic cyanobacterial blooms (Wiedner *et al.*, 2007; Jöhnk *et al.*,

2008). However, the presence of *Aphanizomenon* in the sediment traps and recent sediment samples could represent problems for water quality management including deoxygenation of underlying waters, foul odors e.g. H_2S, undesirable tastes and fish kills (Reynodls & Walsby, 1975; Pearl, 1988). These colonial filaments were not encountered in the open water samples probably because they are known to sink down to deeper layers, and were consequently out of reach of our sampling method. No microbiological assessments were conducted as part of this thesis, however the EPA documented inadequate treatment for bacterial and protozoan pathogens in Guitane between 2008 and 2009 (EPA, 2011).

Disinfection by-products have become a focus of attention in water treatment since THMs were discovered in chlorinated water (Rook, 1974; WHO, 2005). The majority of THM problems in potable supplies are caused by either treatment systems that are incapable of removing organic matter or the complete absence of adequate treatment to remove organic matter in any form (EPA, 2011). Natural variation in DOC and potential increases under future climate change scenarios need to be traced over time and understood. In Ireland, the EPA have observed an upward trend in the number of public water supplies that failed to meet the maximum acceptable concentration THM values of 100 $\mu g\ L^{-1}$ since 2007 (Dunne, 2011). In 2009 a total of 1,851 samples were analyzed for THMs in 979 Irish water supply zones. The results showed that 15.6% failed to comply with the maximum acceptable concentration for total THMs (European Union, 1998) and that 16.1% public water supplies were non-compliant. In Guitane as in other boreal lakes used for drinking water supply, the seasonal variation of organic matter and a future rise in TOC is of concern (Muñoz-Reinoso, 2001; Manley *et al.*, 2008; EIS, 2009; EPA, 2011).

Key Contributions of the Current Research

1. This research helped establish the present ecological response in bacterioplankton and phytoplankton populations and the recent palaeoecology of two lake systems.

2. Neolimnological examination of phytoplankton communities confirmed that higher DOC levels and flashfloods have a direct effect on light attenuation,

depress primary production and promote bacterial / heterotrophic and potentially mixotrophic biomass.

3. The neolimnological examination was augmented with analysis of sediment deposition in the water column. Within and between lake variability reflected the differences in catchment, lake size and morphometry and trap samples clearly reflected seasonal algal succession and interactions with climate parameters.

4. The results of this study emphasize the interdependence of water column parameters, the downward flux of particulate matter and the balance of material arriving in the surface sediments.

5. Palaeolimnological examination of material deposited in the sediment archive extended the period of investigation and contrasting sediment core responses were evident. The divergent levels of TOC in the two lakes contribute to different algal community structures and thus fossil assemblages. These responses can be linked to climate and human activity in the catchment.

6. This three-way examination of lake system components (water column, depositing matter and sediment archives) is novel for this region.

7. This study has detailed ecological responses to natural variation in DOC and evaluated the consequences under future climate change scenarios.

8. The combination of limno- and palaeolimnological studies showed that ongoing debates about climate change and anthropogenic impacts on aquatic systems need strict management plans for aquatic environments.

9. An increase in DOC concentrations will potentially put drinking water quality at risk as allochtonous carbon contributes to excess bacterial growth, causing secondary problems such as disease, taste and smell, and contributing to high disinfection by-products (e.g. THM) levels.

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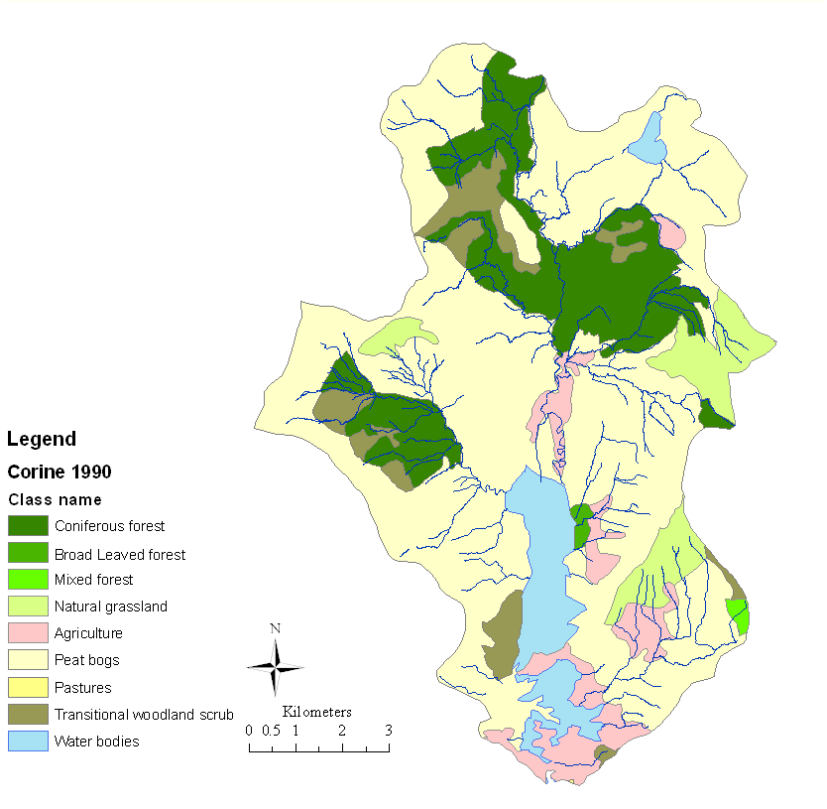
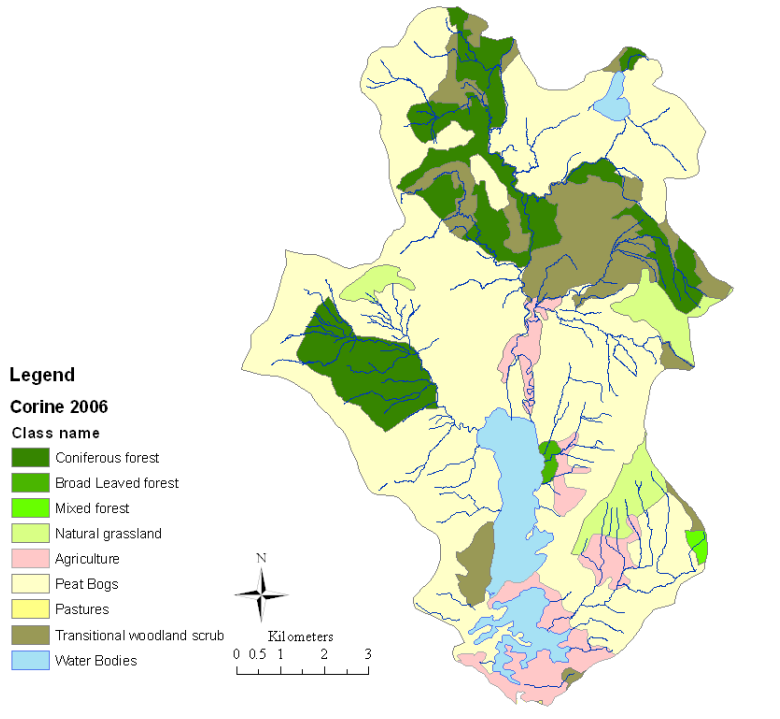
Appendixes

Appendix A - a) Corine 1990 and b) Corine 2006 for Burrishoole catchment.

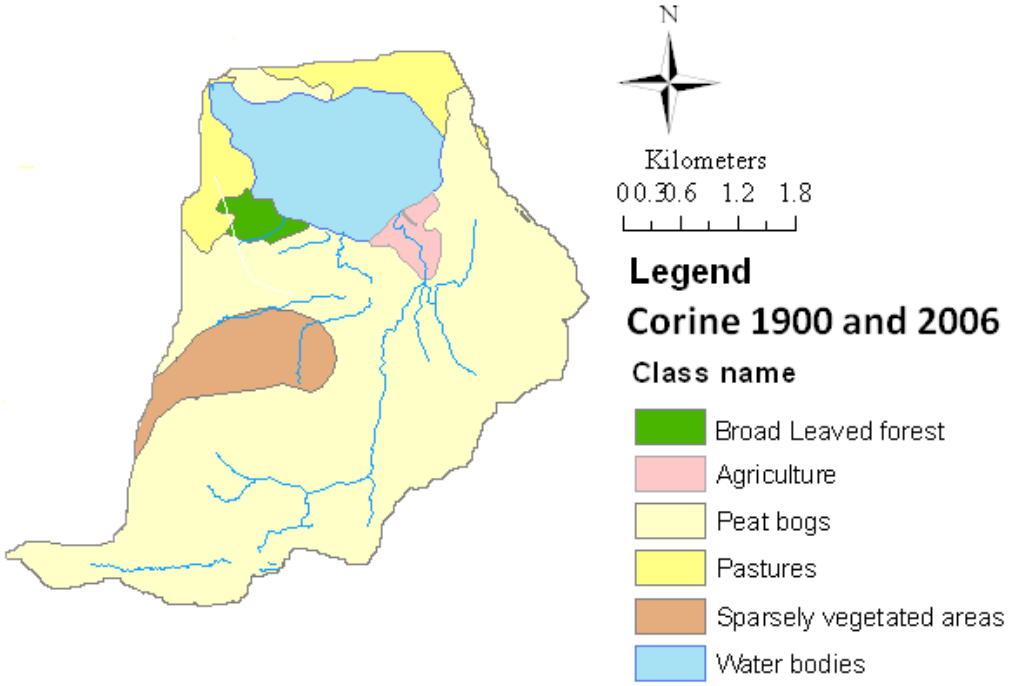
a)



b)



Appendix B - Corine 1990 and 2006 for Guitane catchment.



Appendix C - Chemical parameters measured on a monthly basis in Feeagh and Guitane between May 2009 and April 2010 (n=12).

		Conductivity	pH	Alkalinity	DMRP	TP	NO3-N	Chl a	DOC	TN	Colour	Secchi
		µS/cm		mg/L $CaCO_3$	µg/L	µg/L	µg/L	µg/L	mg/L	µg/L	mg/L PtCo	m
	26May09	91	6.9	5	1	6	79	1.7	8.9	200	64	2.1
	22June09	92	7.1	6	0.5	8	38	1.6	7.1	260	65	2.0
	22July09	90	7.0	6	2	10	54	3.0	7.5	260	73	1.8
	17Aug09	84	6.9	6	2	10	65	2.1	11.4	310	87	1.9
	01Sep09	78.9	6.8	6	1	12	70	1.0	7.4	820	110	1.6
Feeagh	01Oct09	78.6	6.8	6	1	9	71	0.8	7.6	870	99	1.8
	06Nov09	77	6.7	5	1	9	83	0.7	6.4	470	100	1.2
	04Dec09	76	6.7	4	2	7	69	0.3	8.7	370	86	1.8
	06Jan10	79	6.7	5	3	6	69	0.2	6.7	420	85	1.8
	02Feb10	81	6.9	5	2	8	76	0.4	7.7	400	78	1.8
	05Mar10	80	6.8	6	2	5	81	0.3	6.2	480	87	1.9
	07Apr10	77	6.9	6	0.5	7	83	0.9	6.5	680	83	1.8
	19May09	50	6.9	4	0.5	2	106	2.0	6.4	210	21	5.1
	11June09	53	7.0	5	0.5	5	138	2.4	3.8	400	19	4.8
	1July09	53	7.1	5	1	5	92	4.3	3.0	310	18	5.2
	24Aug09	49	7.0	6	0.5	5	75	3.4	3.5	250	19	4.4
Guitane	9Sep09	50.3	6.9	7	1	5	84	3.3	3.5	530	21	4.8
	12Oct09	49	6.9	5	0.5	4	97	2.3	3.1	310	24	4.7
	19Nov09	46	6.7	5	3	15	106	2.5	3.1	280	26	-
	02Dec09	49	6.9	5	1	8	107	1.1	2.7	280	19	5.3
	25Jan10	48	6.8	4	1	3	120	0.9	2.9	330	16	5.3
	17Feb10	47	6.8	4	0.5	3	123	0.8	3.1	390	23	5.7
	13Mar10	48	6.9	5	1	4	180	1.5	3.0	380	22	4.9
	14Apr10	47	7.0	5	0.5	5	123	1.8	1.5	410	23	4.9

Appendix D - Density (cell mL^{-1}) in Feeagh between March 2008 and April 2010 (n=39).

	26/03/2008	29/04/2008	13/05/2008	12/06/2008	08/07/2008	27/08/2008	18/09/2008	14/10/2008	25/11/2008	03/12/2008	26/01/2009	19/02/2009	01/04/2009	16/04/2009	24/04/2009	12/05/2009
Asterionella formosa	33	145	76	2	5	36	21	17	12	16	8	16	133	279	197	415
Aulacoseira alpigena	69	71	9	4	8	37	38	36	27	27	27	35	76	99	63	71
Aulacoseira subarctica	23	72	26	2	5	27	34	16	13	19	1	10	13	13	15	0
Cyclotella radiosa	2	9	11	16	4	4	1	1	7	5	1	0	0	1	1	1
Cyclotella kuetzingiana	0	0	0	11	2	0	0	0	0	1	3	0	0	0	1	1
Eunotia cfr *incisa*	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Fragilaria arcus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Fragilaria crotonensis	1	0	0	0	0	2	1	1	0	0	0	0	0	0	0	0
Fragilaria ulna	0	1	0	0	0	0	0	0	0	0	0	0	0	4	2	0
Frustulia sp.	0	0	0	0	0	0	0	0	0	0	1	1	0	0	0	0
Rhizosolenia sp.	0	1	1	2	4	1	1	1	1	3	0	1	0	0	0	1
Tabellaria flocculosa var. *asterionelloides*	0	20	7	6	9	2	5	0	7	0	2	0	6	5	19	42
Tabellaria flocculosa	0	3	0	1	13	1	0	0	6	0	56	6	3	0	0	18
Synedra sp.	0	0	0	0	0	0	0	0	0	0	0	0	1	11	13	5
Navicula spp.	0	1	0	0	0	1	0	0	0	0	3	0	0	1	0	6
Pennates	1	57	7	2	16	22	14	3	14	1	8	2	0	2	0	8
Anabaena flos aquae	0	0	0	0	5	2	6	0	0	0	0	0	0	0	0	0
Aphanocapsa	0	0	0	22	0	438	25	65	0	8	0	0	0	16	0	0
Oscillatoria agardhii	0	1	0	16	9	194	180	26	11	24	0	1	0	0	4	0
Snowella cf lacustris	0	0	0	0	0	0	3	0	0	0	0	0	0	0	0	0
Woronichinia naegeliana	0	0	24	0	0	0	3	0	18	6	0	0	0	0	0	0
Ankistrodesmus fusiformis	0	0	0	0	1	0	4	0	0	0	0	0	0	0	0	0
Bitrichia longispina	0	0	1	1	0	1	1	0	0	0	0	2	0	0	0	0
Botryococcus braunii	0	8	0	11	7	9	9	10	27	2	0	3	27	8	56	131
Carteria sp.	0	0	0	0	0	0	1	0	1	1	0	0	0	0	0	0
Chlamydomonas sp.	0	7	7	0	0	14	22	15	10	3	0	15	0	0	0	0
Closterium abruptum	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Closterium acutum var. *variabile*	17	9	10	1	19	9	16	19	20	18	10	21	9	21	3	2
Closterium gracile	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Closterium kuetzingii	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Closterium navicula	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0
Crucigeniella rectangularis	0	1	0	0	0	0	0	0	0	0	0	0	0	0	2	0
Coelastrum microporum	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Coenococcus planctonicus	1	3	2	39	2	10	7	4	1	1	0	0	0	8	2	2
Coenocuccus polycoccus	2	0	0	8	6	23	7	1	2	2	0	0	0	0	0	0
Cosmarium abbreviatum var. *planktonicum*	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Cosmarium depressum	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Cosmarium blyttii	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0
Cosmarium humile	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Chlorolobion braunii	0	0	0	1	1	0	0	0	0	0	0	0	0	0	0	0
Dictyosphaerium pulchellum	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
Kirchneriella obesa	0	3	0	5	0	5	6	4	0	0	0	0	0	0	0	0
Klebsormidium sp.	0	0	0	0	0	3	0	1	1	0	3	2	6	14	3	2
Monoraphidium contortum	0	2	3	0	5	6	0	1	0	0	6	6	5	60	61	66
Monoraphidium griffithii	1	2	0	0	2	7	2	7	1	2	0	0	0	0	0	0
Monoraphidium minutum	14	0	15	0	34	132	19	16	10	3	2	0	5	8	10	8
Mougeotia sp.	0	0	0	2	0	0	0	0	0	0	4	2	0	1	0	0
Oocystis parva	0	0	0	0	1	0	1	4	0	0	0	0	0	2	0	1
Phacus sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Scenedesmus granulatus	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0
Single round cell	59	90	94	0	68	79	23	48	79	0	0	4	0	0	0	0
Pseudosphaerocystis lacustris	0	4	0	4	3	0	4	0	0	0	0	2	1	2	5	3
Spondylosium planum	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Staurastrum anatinum	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Staurastrum arctiscon	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Staurastrum cingulum	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Staurastrum lunatum	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Staurodesmus sellatus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Tetraedron triangulare	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0
Tetraedron minimum	0	8	7	0	0	0	0	3	4	1	0	0	0	0	0	0
Rhodomonas acuta	14	35	768	467	303	51	10	26	12	136	7	9	6	34	548	467
Rhodomonas minuta	4	31	49	34	75	242	49	58	26	14	19	22	17	71	43	60
Cryptomonas marssonii	0	0	0	2	0	0	1	1	0	12	0	0	0	0	0	0
Cryptomonas sp.	0	1	3	11	43	110	1	9	2	2	0	0	0	2	1	4
Chrysochromulina parva	2	7	71	26	36	26	15	34	10	35	0	3	1	4	26	19
Dinobryon sociale	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0
Ochromonas tuberculata	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	1
Mallomonas akrokomos	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
Mallomonas caudata	0	0	0	0	0	0	0	0	0	0	0	1	1	0	0	0
Gymnodinium uberrimum	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Gymnodinium triceratium	0	0	0	0	0	1	2	1	2	0	0	0	0	1	2	2
Ceratium hirudinella	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Trachelomonas volvocina	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0
Ciliates	3	2	2	11	10	10	11	9	4	6	4	1	4	4	5	8

Appendix D continues - Density (cell mL^{-1}) in Feeagh between March 2008 and April 2010 (n=39).

	26/05/2009	11/06/2009	22/06/2009	06/07/2009	22/07/2009	04/08/2009	17/08/2009	27/08/2009	07/09/2009	01/10/2009	22/10/2009	06/11/2009	20/11/2009	04/12/2009	22/12/2009	06/01/2010	20/01/2010
Asterionella formosa	504	307	4	2	0	4	7	33	15	17	21	16	12	10	7	9	8
Aulacoseira alpigena	38	23	20	12	56	34	12	19	21	35	50	37	26	15	19	25	24
Aulacoseira subarctica	0	2	0	7	0	12	1	2	2	5	0	0	0	0	1	0	1
Cyclotella radiosa	3	1	3	2	2	15	13	11	11	6	4	1	3	0	4	2	2
Cyclotella kuetzingiana	1	0	0	0	9	2	10	16	9	2	1	2	1	0	0	0	0
Eunotia cfr *incisa*	0	0	0	0	0	0	0	0	1	0	0	0	1	0	1	0	0
Fragilaria arcus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Fragilaria crotonensis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Fragilaria ulna	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
Frustulia sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Rhizosolenia sp.	1	1	0	6	21	1	0	1	1	1	0	0	0	0	0	0	0
Tabellaria flocculosa var. *asterionelloides*	8	6	3	2	0	1	0	3	2	1	1	2	2	2	1	1	2
Tabellaria flocculosa	18	1	1	19	2	16	7	6	1	2	1	6	2	1	0	0	0
Synedra sp.	5	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Navicula spp.	6	0	1	0	0	0	1	1	0	0	0	0	0	0	0	0	0
Pennates	1	0	2	0	5	8	5	5	2	2	2	1	1	1	2	0	3
Anabaena flos aquae	0	0	0	0	0	0	0	2	1	0	0	0	0	0	0	0	0
Aphanocapsa	56	7	13	0	70	0	0	0	6	10	0	0	0	0	0	0	0
Oscillatoria agardhii	0	0	0	0	0	1	5	0	3	0	5	0	0	0	0	0	0
Snowella cf lacustris	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Woronichinia naegeliana	48	0	16	7	0	6	901	0	106	165	26	0	0	0	0	2	0
Ankistrodesmus fusiformis	0	0	0	0	6	2	1	2	0	0	0	0	0	0	0	0	0
Bitrichia longispina	0	0	0	0	0	0	2	0	0	1	0	0	0	0	0	0	0
Botryococcus braunii	2	1	0	15	34	5	17	0	21	8	1	9	7	7	24	3	0
Carteria sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	5	0	2	0
Chlamydomonas sp.	0	0	0	0	0	1	5	8	58	13	2	2	0	0	0	3	0
Closterium abruptum	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Closterium acutum var. *variabile*	3	0	0	2	26	21	28	40	46	13	9	8	5	11	2	4	9
Closterium gracile	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Closterium kuetzingii	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Closterium navicula	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Crucigeniella rectangularis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Coelastrum microporum	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Coenococcus planctonicus	3	0	1	0	25	0	0	0	0	0	2	0	3	0	0	1	1
Coenocuccus polycoccus	0	0	0	2	0	0	0	0	9	0	0	0	0	0	1	0	0
Cosmarium abbreviatum var. *planktonicum*	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Cosmarium depressum	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	1
Cosmarium blyttii	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Cosmarium humile	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Chlorolobion braunii	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Dictyosphaerium pulchellum	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Kirchneriella obesa	0	0	0	0	0	0	1	0	0	0	0	0	4	0	0	0	0
Klebsormidium sp.	0	0	9	1	0	0	0	0	1	0	0	1	1	0	0	0	0
Monoraphidium contortum	60	8	4	3	9	22	38	21	10	5	0	15	2	2	2	0	0
Monoraphidium griffithii	0	25	0	5	1	11	35	19	40	27	15	5	1	7	3	4	2
Monoraphidium minutum	10	10	47	30	71	79	66	16	25	9	7	7	10	4	1	0	5
Mougeotia sp.	0	1	6	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Oocystis parva	0	0	0	1	7	0	0	0	1	2	1	1	0	0	0	0	0
Phacus sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Scenedesmus granulatus	0	0	0	0	0	0	0	1	0	1	0	1	0	0	0	0	0
Single round cell	0	71	21	30	97	31	13	36	33	3	1	24	59	30	142	27	115
Pseudosphaerocystis lacustris	2	2	3	1	32	5	0	1	7	0	0	0	0	0	0	0	0
Spondylosium planum	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Staurastrum anatinum	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Staurastrum arctiscon	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Staurastrum cingulum	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Staurastrum lunatum	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Staurodesmus sellatus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Tetraedron triangulare	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0
Tetraedron minimum	0	0	0	0	0	0	1	0	2	0	0	0	0	2	0	4	1
Rhodomonas acuta	391	453	475	362	266	166	259	133	112	60	53	33	6	6	10	1	7
Rhodomonas minuta	24	28	33	20	17	31	79	25	7	4	3	12	21	17	18	19	11
Cryptomonas marssonii	0	0	0	17	0	0	0	0	0	0	0	3	2	2	0	0	0
Cryptomonas sp.	4	14	36	159	1	1	5	1	2	1	1	0	0	0	0	0	0
Chrysochromulina parva	24	30	4	370	9	46	16	0	2	3	4	11	0	0	23	3	10
Dinobryon sociale	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Ochromonas tuberculata	1	0	0	0	0	0	0	0	0	0	0	4	4	0	0	1	0
Mallomonas akrokomos	0	0	0	1	0	1	0	0	0	0	0	0	0	0	0	0	0
Mallomonas caudata	0	0	1	0	0	0	0	0	0	1	1	0	1	0	0	0	0
Gymnodinium uberrimum	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Gymnodinium triceratium	0	1	0	1	0	0	0	0	0	0	0	0	1	1	1	1	2
Ceratium hirudinella	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Trachelomonas volvocina	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Ciliates	6	14	11	13	9	10	8	8	7	6	4	5	5	4	4	4	3

Appendix D continues - Density (cell mL^{-1}) in Feeagh between March 2008 and April 2010 (n=39).

	02/02/2010	11/02/2010	05/03/2010	15/03/2010	07/04/2010	19/04/2010
Asterionella formosa	3	13	12	29	73	312
Aulacoseira alpigena	36	44	51	85	103	290
Aulacoseira subarctica	1	6	8	6	23	25
Cyclotella radiosa	1	0	0	1	0	0
Cyclotella kuetzingiana	0	4	0	0	0	0
Eunotia cfr *incisa*	0	0	0	0	0	0
Fragilaria arcus	0	0	0	0	0	0
Fragilaria crotonensis	0	0	0	0	0	0
Fragilaria ulna	0	0	0	0	0	0
Frustulia sp.	0	0	0	0	0	0
Rhizosolenia sp.	0	0	0	0	0	0
Tabellaria flocculosa var. *asterionelloides*	0	0	5	1	1	3
Tabellaria flocculosa	0	0	2	1	13	0
Synedra sp.	0	0	0	0	0	0
Navicula spp.	0	0	0	0	2	3
Pennates	3	4	1	1	25	23
Anabaena flos aquae	0	0	0	0	0	0
Aphanocapsa	0	0	0	0	0	0
Oscillatoria agardhii	0	0	0	0	0	0
Snowella cf lacustris	0	0	0	0	0	0
Woronichinia naegeliana	0	8	2	4	0	8
Ankistrodesmus fusiformis	0	0	0	0	0	0
Bitrichia longispina	0	0	0	0	0	0
Botryococcus braunii	0	0	5	4	0	4
Carteria sp.	1	2	0	0	0	0
Chlamydomonas sp.	4	13	1	0	8	1
Closterium abruptum	0	0	0	0	0	0
Closterium acutum var. *variabile*	7	3	9	7	8	24
Closterium gracile	0	0	0	0	0	0
Closterium kuetzingii	0	0	0	0	0	0
Closterium navicula	0	0	0	0	0	0
Crucigeniella rectangularis	0	0	0	0	0	0
Coelastrum microporum	0	0	0	0	0	0
Coenococcus planctonicus	0	0	0	1	1	2
Coenocuccus polycoccus	1	0	0	0	0	1
Cosmarium abbreviatum var. *planktonicum*	0	0	0	0	0	0
Cosmarium depressum	0	0	0	0	0	0
Cosmarium blyttii	0	0	0	0	0	0
Cosmarium humile	0	0	0	0	0	0
Chlorolobion braunii	0	0	0	0	0	0
Dictyosphaerium pulchellum	0	0	0	0	0	0
Kirchneriella obesa	0	0	0	0	0	0
Klebsormidium sp.	0	0	5	0	0	0
Monoraphidium contortum	0	0	2	3	2	2
Monoraphidium griffithii	1	2	3	4	18	9
Monoraphidium minutum	4	1	4	3	7	0
Mougeotia sp.	0	0	0	1	0	0
Oocystis parva	0	0	0	0	0	1
Phacus sp.	0	0	0	0	0	0
Scenedesmus granulatus	0	0	0	0	0	0
Single round cell	181	269	7	18	25	33
Pseudosphaerocystis lacustris	0	0	0	0	0	0
Spondylosium planum	0	0	0	0	0	0
Staurastrum anatinum	0	0	0	0	0	0
Staurastrum arctiscon	0	0	0	0	0	0
Staurastrum cingulum	0	0	0	0	0	0
Staurastrum lunatum	0	0	0	0	0	0
Staurodesmus sellatus	0	0	0	0	0	0
Tetraedron triangulare	0	0	0	0	0	0
Tetraedron minimum	4	2	2	1	0	1
Rhodomonas acuta	5	4	8	10	44	143
Rhodomonas minuta	15	27	6	4	12	14
Cryptomonas marssonii	0	0	0	0	0	0
Cryptomonas sp.	0	0	0	0	0	0
Chrysochromulina parva	25	0	0	0	8	12
Dinobryon sociale	0	0	0	0	0	0
Ochromonas tuberculata	0	0	0	0	0	0
Mallomonas akrokomos	0	0	0	0	0	0
Mallomonas caudata	0	0	0	0	1	0
Gymnodinium uberrimum	0	0	0	0	0	0
Gymnodinium triceratium	1	3	0	1	1	1
Ceratium hirudinella	0	0	0	0	0	0
Trachelomonas volvocina	0	0	0	0	0	0
Ciliates	5	4	3	3	4	5

Appendix E - Algal and Ciliates biovolume (μm^3) and biomass ($mm^3\ m^{-3}$) in Feeagh between March '08 and Apr '10 (n=39)

	Biovolume	26/03/2008	29/04/2008	13/05/2008	12/06/2008	08/07/2008	27/08/2008	18/09/2008	14/10/2008	25/11/2008	03/12/2008	26/01/2009	19/02/2009	01/04/2009	16/04/2009	24/04/2009	12/05/2009
Asterionella formosa	402.0	13.	58.	30.	0.9	2.1	14.3	8.5	6.7	4.7	6.6	3.3	6.6	53.	112.	79.	166.
Aulacoseira alpigena	154.0	10.	10.	1.3	0.6	1.2	5.8	5.8	5.5	4.2	4.1	4.2	5.4	11.	15.2	9.7	10.9
Aulacoseira subarctica	1342.0	31.	96.	35.	2.0	7.0	36.7	45.	21.	17.	25.	1.1	12.	16.	17.7	19.	0.0
Cyclotella radiosa	2132.0	4.5	20.	24.	33.3	8.9	8.9	2.2	2.1	15.	11.	2.1	0.0	0.0	2.1	2.1	2.1
Cyclotella kuetzingiana	475.0	0.0	0.0	0.0	5.4	1.0	0.0	0.0	0.0	0.1	0.5	1.5	0.0	0.2	0.0	0.4	0.3
Eunotia cfr *incisa*	948.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.0	0.0	0.0
Fragilaria arcus	850.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.2	0.0	0.0
Fragilaria crotonensis	1072.0	1.5	0.0	0.0	0.0	0.0	2.4	1.5	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Fragilaria ulna	5346.4	0.0	3.4	1.7	0.0	0.9	0.0	0.0	0.2	0.0	0.4	0.0	0.0	2.1	20.3	12.	0.0
Frustulia sp.	970.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.8	0.8	0.1	0.4	0.0	0.0
Rhizosolenia sp.	964.0	0.1	0.8	0.9	1.8	4.0	0.8	1.1	0.6	1.3	3.0	0.0	1.0	0.2	0.4	0.0	1.3
Tabellaria flocculosa var. *asterionelloides*	242.0	0.1	4.8	1.6	1.5	2.1	0.4	1.1	0.0	1.8	0.0	0.5	0.0	1.5	1.2	4.6	10.2
Tabellaria flocculosa	126.0	0.0	0.4	0.0	0.1	1.6	0.2	0.0	0.0	0.8	0.0	7.1	0.7	0.3	0.0	0.0	2.3
Synedra sp.	327.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.2	3.7	4.2	1.6
Navicula spp.	970.0	0.0	0.8	0.1	0.0	0.0	0.5	0.0	0.0	0.0	0.0	2.8	0.0	0.4	0.6	0.0	5.8
Pennates	243.0	0.2	13.	1.8	0.4	3.8	5.2	3.3	0.7	3.4	0.2	1.9	0.5	0.1	0.4	0.1	1.9
Anabaena flos aquae	120.5	0.0	0.0	0.0	0.0	0.6	0.2	0.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Aphanocapsa	1.2	0.0	0.0	0.0	0.0	0.0	0.5	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Oscillatoria agardhii	213.0	0.0	0.2	0.0	3.5	2.0	41.3	38.	5.6	2.3	5.1	0.0	0.1	0.0	0.0	0.8	0.0
Snowella cf lacustris	7.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Woronichinia naegeliana	22.0	0.0	0.0	0.5	0.0	0.0	0.0	0.1	0.0	0.4	0.1	0.0	0.0	0.0	0.0	0.0	0.0
Ankistrodesmus fusiformis	40.3	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Bitrichia longispina	90.4	0.0	0.0	0.1	0.1	0.0	0.1	0.1	0.0	0.0	0.0	0.0	0.2	0.0	0.0	0.0	0.0
Botryococcus braunii	1.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.2
Carteria sp.	561.0	0.0	0.0	0.0	0.0	0.0	0.0	0.6	0.0	0.6	0.6	0.0	0.0	0.0	0.0	0.0	0.0
Chlamydomonas sp.	374.0	0.0	2.7	2.7	0.0	0.0	5.1	8.2	5.4	3.9	1.2	0.0	5.4	0.0	0.0	0.0	0.0
Closterium abruptum	48143.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	9.6	4.8	0.0	0.0	0.0
Closterium acutum var. *variabile*	186.0	3.1	1.7	1.9	0.2	3.5	1.7	2.9	3.5	3.7	3.3	1.9	3.9	1.7	3.9	0.6	0.4
Closterium gracile	5801.0	0.0	0.0	0.5	0.9	0.2	0.7	1.2	0.9	0.0	1.4	0.0	0.0	0.0	1.7	2.3	0.0
Closterium kuetzingii	31586.	0.0	2.5	0.0	1.3	1.3	0.0	0.0	0.0	0.0	0.0	5.1	0.0	0.0	6.3	3.8	0.0
Closterium navicula	2308.2	0.0	0.2	0.0	0.0	0.0	0.0	0.4	0.4	0.0	0.0	0.0	1.8	0.0	0.2	0.0	0.0
Crucigeniella rectangularis	36.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0
Coelastrum microporum	180.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Coenococcus planctonicus	30.7	0.0	0.1	0.1	1.2	0.1	0.3	0.2	0.1	0.0	0.0	0.0	0.0	0.0	0.3	0.1	0.1
Coenocuccus polycoccus	491.0	0.9	0.2	0.0	3.8	2.7	11.4	3.4	0.4	1.1	1.1	0.0	0.0	0.0	0.0	0.0	0.0
Cosmarium abbreviatum var. *planktonicum*	1663.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.0	0.0	0.0
Cosmarium depressum	1170.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0
Cosmarium blyttii	2241.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0	2.3	0.0	0.0	0.0	0.0
Cosmarium humile	782.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Chlorolobion braunii	60.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Dictyosphaerium pulchellum	35.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Kirchneriella obesa	20.2	0.0	0.1	0.0	0.1	0.0	0.1	0.1	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Klebsormidium sp.	3933.7	0.0	0.0	0.0	0.0	1.9	13.5	0.0	5.0	3.8	0.0	10.	8.2	23.	54.3	11.	7.9
Monoraphidium contortum	33.6	0.0	0.1	0.1	0.0	0.2	0.2	0.0	0.0	0.0	0.0	0.2	0.2	0.2	2.0	2.1	2.2
Monoraphidium griffithii	57.0	0.1	0.1	0.0	0.0	0.1	0.4	0.1	0.4	0.1	0.1	0.0	0.0	0.0	0.0	0.0	0.0
Monoraphidium minutum	85.0	1.1	0.0	1.2	0.0	2.9	11.2	1.6	1.3	0.9	0.3	0.2	0.0	0.4	0.7	0.9	0.7
Mougeotia sp.	7620.4	0.0	0.0	0.0	16.5	0.9	0.6	0.0	0.6	0.0	0.0	34.	15.	1.2	6.1	0.0	0.0
Oocystis parva	93.0	0.0	0.0	0.0	0.0	0.1	0.0	0.1	0.4	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.1
Phacus sp.	2748.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Scenedesmus granulatus	27.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Single round cell	14.1	0.8	1.3	1.3	0.0	1.0	1.1	0.3	0.7	1.1	0.0	0.0	0.1	0.0	0.0	0.0	0.0
Pseudosphaerocystis lacustris	246.0	0.0	0.9	0.0	0.9	0.7	0.0	1.0	0.0	0.0	0.0	0.1	0.4	0.2	0.4	1.2	0.8
Spondylosium planum	235.9	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Staurastrum anatinum	11874.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	3.6
Staurastrum arctiscon	25344.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Staurastrum cingulum	8817.3	0.0	0.0	0.0	0.7	0.0	0.4	0.4	0.4	1.4	0.0	0.0	0.0	0.0	0.0	0.9	0.0
Staurastrum lunatum	16005.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Staurodesmus sellatus	12052.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.5	3.6
Tetraedron triangulare	288.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0
Tetraedron minimum	941.2	0.0	7.8	6.9	0.0	0.0	0.0	0.0	2.9	3.9	1.0	0.0	0.0	0.0	0.0	0.0	0.0
Rhodomonas acuta	98.0	0.6	1.5	32.	19.7	12.7	2.1	0.4	1.1	0.5	5.7	0.3	0.9	0.6	3.4	53.	45.8
Rhodomonas minuta	45.0	0.1	1.1	1.7	1.2	2.6	8.5	1.7	2.0	0.9	0.5	0.7	1.0	0.7	3.2	1.9	2.7
Cryptomonas marssonii	308.0	0.0	0.0	0.0	0.5	0.0	0.0	0.2	0.2	0.0	2.9	0.0	0.0	0.0	0.0	0.0	0.0
Cryptomonas sp.	3270.0	0.0	3.3	9.5	34.9	136.	348.	3.3	29.	6.6	6.6	0.0	0.0	0.0	6.8	3.4	13.6
Chrysochromulina parva	67.9	0.1	0.5	4.8	1.8	2.5	1.8	1.0	2.3	0.7	2.4	0.0	0.2	0.1	0.3	1.8	1.3
Dinobryon sociale	349.0	0.0	0.0	0.0	0.1	0.7	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Ochromonas tuberculata	1349.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.8	0.0	0.0	0.0	1.3
Mallomonas akrokomos	153.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Mallomonas caudata	21696.	0.0	0.0	0.0	6.9	6.9	6.5	2.2	0.0	0.0	0.0	0.0	22.	21.	6.5	2.2	0.0
Gymnodinium uberrimum	27332.	0.0	0.0	0.0	0.0	0.0	1.1	0.0	0.0	1.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Gymnodinium triceratium	1187.6	0.0	0.0	0.0	0.0	0.2	1.2	2.5	1.2	2.5	0.0	0.5	0.0	0.0	1.2	1.9	2.5
Ceratium hirudinella	61348.	4.9	0.0	0.0	4.9	12.3	2.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	4.9	0.0
Trachelomonas volvocina	571.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.6	0.0	0.0	0.0	0.0	0.0
Ciliates	5631.4	12.	7.1	9.2	166.	83.0	79.8	28.	21.	9.2	22.	22.	5.9	17.	13.5	14.	37.7

Appendix E continues - Algal and Ciliates biovolume (μm^3) and biomass ($mm^3\ m^{-3}$) in Feeagh between March '08 and Apr '10 (n=39)

	26/05/2009	11/06/2009	22/06/2009	06/07/2009	22/07/2009	04/08/2009	17/08/2009	27/08/2009	07/09/2009	01/10/2009	22/10/2009	06/11/2009	20/11/2009	04/12/2009	22/12/2009	06/01/2010	20/01/2010
Asterionella formosa	202.4	123.6	1.6	0.6	0.2	1.5	2.8	13.3	6.2	6.9	8.3	6.3	4.9	4.1	3.0	3.7	3.2
Aulacoseira alpigena	5.9	3.6	3.1	1.8	8.6	5.3	1.8	2.9	3.3	5.4	7.6	5.7	3.9	2.4	3.0	3.8	3.7
Aulacoseira subarctica	0.3	2.8	0.4	8.9	0.0	15.7	1.8	2.7	2.3	6.7	0.0	0.0	0.4	0.5	1.9	0.1	0.7
Cyclotella radiosa	6.6	2.1	6.6	4.4	4.5	31.1	26.7	24.3	24.4	13.3	8.9	2.2	6.7	0.0	9.0	4.5	4.5
Cyclotella kuetzingiana	0.4	0.0	0.0	0.0	4.2	0.9	4.9	7.7	4.1	0.9	0.4	1.0	0.5	0.2	0.0	0.0	0.0
Eunotia cfr *incisa*	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.8	0.4	0.0	0.0	1.1	0.0	1.1	0.0	0.0
Fragilaria arcus	0.0	0.0	0.0	0.0	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Fragilaria crotonensis	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Fragilaria ulna	1.8	1.1	0.4	0.9	6.4	1.7	1.1	0.0	1.1	0.0	0.4	0.0	0.0	0.0	0.5	0.0	0.0
Frustulia sp.	0.0	0.0	0.1	0.0	0.0	0.1	0.1	0.2	0.2	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0
Rhizosolenia sp.	1.3	1.0	0.3	6.0	20.1	0.9	0.2	0.6	0.5	0.6	0.3	0.1	0.1	0.1	0.0	0.0	0.0
Tabellaria flocculosa var. *asterionelloides*	1.9	1.4	0.8	0.6	0.0	0.2	0.0	0.8	0.5	0.2	0.3	0.5	0.5	0.4	0.2	0.1	0.4
Tabellaria flocculosa	2.3	0.2	0.2	2.4	0.3	2.0	0.9	0.8	0.1	0.2	0.1	0.8	0.3	0.2	0.0	0.1	0.0
Synedra sp.	1.7	0.1	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Navicula spp.	5.8	0.0	0.8	0.0	0.0	0.0	0.5	0.6	0.4	0.0	0.2	0.0	0.0	0.0	0.1	0.0	0.0
Pennates	0.2	0.0	0.5	0.0	1.2	2.0	1.3	1.2	0.5	0.6	0.6	0.3	0.3	0.1	0.5	0.1	0.8
Anabaena flos aquae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Aphanocapsa	0.1	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Oscillatoria agardhii	0.0	0.0	0.0	0.0	0.0	0.3	1.1	0.0	0.7	0.0	1.1	0.0	0.0	0.0	0.0	0.0	0.0
Snowella cf lacustris	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Woronichinia naegeliana	1.1	0.0	0.4	0.1	0.0	0.1	19.8	0.0	2.3	3.6	0.6	0.0	0.0	0.0	0.0	0.0	0.0
Ankistrodesmus fusiformis	0.0	0.0	0.0	0.0	0.2	0.1	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Bitrichia longispina	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Botryococcus braunii	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Carteria sp.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.9	0.0	1.2	0.0
Chlamydomonas sp.	0.0	0.0	0.0	0.0	0.0	0.4	1.9	3.1	21.8	4.7	0.8	0.8	0.0	0.0	0.0	1.2	0.0
Closterium abruptum	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Closterium acutum var. *variabile*	0.6	0.0	0.0	0.4	4.8	3.9	5.2	7.3	8.5	2.3	1.7	1.5	1.0	2.1	0.4	0.8	1.7
Closterium gracile	0.6	0.0	1.0	1.4	0.6	0.9	1.4	1.7	0.6	0.0	1.2	0.2	0.0	0.0	1.2	0.0	0.0
Closterium kuetzingii	2.5	0.0	0.0	0.0	3.2	0.0	0.0	0.0	0.0	0.0	3.2	1.3	0.0	0.0	0.0	0.0	0.0
Closterium navicula	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.0	0.0	0.0	0.1	0.9	0.0	0.0	0.0	0.0
Crucigeniella rectangularis	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Coelastrum microporum	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Coenococcus planctonicus	0.1	0.0	0.0	0.0	0.8	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.1	0.0	0.0	0.0	0.0
Coenococcus polycoccus	0.0	0.1	0.0	0.9	0.0	0.0	0.0	0.0	4.4	0.0	0.0	0.0	0.2	0.1	0.5	0.0	0.0
Cosmarium abbreviatum var. *planktonicum*	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Cosmarium depressum	0.0	0.0	0.1	0.1	0.2	0.0	0.0	0.0	0.0	1.1	0.0	0.0	0.0	0.0	0.0	0.0	1.2
Cosmarium blyttii	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Cosmarium humile	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Chlorolobion braunii	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Dictyosphaerium pulchellum	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Kirchneriella obesa	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0
Klebsormidium sp.	0.0	0.0	34.6	2.7	0.0	0.0	0.6	0.0	3.5	0.0	0.0	2.0	4.4	0.0	0.0	0.0	0.0
Monoraphidium contortum	2.0	0.3	0.1	0.1	0.3	0.7	1.3	0.7	0.3	0.2	0.0	0.5	0.1	0.1	0.1	0.0	0.0
Monoraphidium griffithii	0.0	1.4	0.0	0.3	0.1	0.7	2.0	1.1	2.3	1.5	0.8	0.3	0.1	0.4	0.2	0.2	0.1
Monoraphidium minutum	0.9	0.9	4.0	2.6	6.0	6.7	5.6	1.3	2.1	0.8	0.6	0.6	0.9	0.4	0.1	0.0	0.4
Mougeotia sp.	3.0	8.4	45.6	0.0	0.0	0.0	0.0	0.0	1.8	0.0	2.4	0.0	0.0	0.0	0.0	0.0	1.5
Oocystis parva	0.0	0.0	0.0	0.1	0.7	0.0	0.0	0.0	0.1	0.1	0.1	0.1	0.0	0.0	0.0	0.0	0.0
Phacus sp.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0
Scenedesmus granulatus	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Single round cell	0.0	1.0	0.3	0.4	1.4	0.4	0.2	0.5	0.5	0.0	0.0	0.3	0.8	0.4	2.0	0.4	1.6
Pseudosphaerocystis lacustris	0.4	0.5	0.7	0.3	7.8	1.2	0.0	0.2	1.8	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Spondylosium planum	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Staurastrum anatinum	0.0	0.0	0.0	0.0	2.4	0.9	0.9	2.4	0.9	0.0	0.0	0.5	0.9	0.0	0.0	0.0	0.0
Staurastrum arctiscon	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Staurastrum cingulum	0.9	0.9	0.0	1.8	0.0	0.0	0.9	0.0	0.9	0.9	0.7	0.0	0.0	0.0	0.0	0.0	0.0
Staurastrum lunatum	0.0	0.0	1.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Staurodesmus sellatus	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Tetraedron triangulare	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Tetraedron minimum	0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0	2.0	0.0	0.0	0.0	0.0	2.0	0.0	4.0	0.9
Rhodomonas acuta	38.3	44.4	46.6	35.5	26.1	16.3	25.4	13.0	11.0	5.9	5.2	3.3	0.6	0.6	1.0	0.1	0.7
Rhodomonas minuta	1.1	1.3	1.5	0.9	0.7	1.4	3.6	1.1	0.3	0.2	0.1	0.6	0.9	0.7	0.8	0.8	0.5
Cryptomonas marssonii	0.0	0.0	0.0	5.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.6	0.6	0.0	0.0	0.0
Cryptomonas sp.	13.6	44.1	119.0	520.2	3.3	3.4	17.0	3.4	6.8	3.4	3.4	0.0	0.0	0.0	0.0	0.0	0.0
Chrysochromulina parva	1.6	2.1	0.3	25.1	0.6	3.1	1.1	0.0	0.1	0.2	0.3	0.8	0.0	0.0	1.6	0.2	0.7
Dinobryon sociale	0.1	0.0	0.1	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Ochromonas tuberculata	1.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	5.6	5.6	0.0	0.0	1.3	0.0
Mallomonas akrokomos	0.0	0.0	0.0	0.2	0.0	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Mallomonas caudata	2.2	0.0	18.4	2.6	2.2	0.0	4.3	4.3	6.5	21.7	21.7	1.7	22.6	0.0	0.0	0.0	0.0
Gymnodinium uberrimum	0.0	2.2	2.7	0.0	0.0	2.7	2.7	0.0	5.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Gymnodinium triceratium	0.0	0.8	0.0	1.2	0.0	0.2	0.1	0.1	0.1	0.1	0.2	0.1	1.2	1.2	1.2	1.2	2.5
Ceratium hirudinella	0.0	0.0	4.9	4.9	18.4	6.1	4.9	0.0	4.9	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Trachelomonas volvocina	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0
Ciliates	30.7	150.6	107.5	57.5	18.6	84.5	85.6	93.1	17.7	17.1	8.2	9.6	10.3	16.8	16.4	13.0	28.4

Appendix E continues - Algal and Ciliates biovolume (μm^3) and biomass ($mm^3\ m^{-3}$) in Feeagh between March '08 and Apr '10 (n=39)

	02/02/2010	11/02/2010	05/03/2010	15/03/2010	07/04/2010	19/04/2010
Asterionella formosa	1.3	5.3	4.9	11.5	29.5	125.6
Aulacoseira alpigena	5.5	6.8	7.8	13.1	15.9	44.6
Aulacoseira subarctica	1.1	8.6	10.9	7.8	31.5	33.1
Cyclotella radiosa	2.1	0.0	0.0	2.1	0.0	0.0
Cyclotella kuetzingiana	0.0	2.0	0.0	0.0	0.0	0.0
Eunotia cfr *incisa*	0.0	0.0	0.0	0.0	0.0	0.0
Fragilaria arcus	0.0	0.0	0.0	0.0	0.0	0.0
Fragilaria crotonensis	0.0	0.0	0.0	0.0	0.0	0.2
Fragilaria ulna	0.0	1.6	0.0	0.0	2.1	1.7
Frustulia sp.	0.0	0.0	0.0	0.0	0.0	0.0
Rhizosolenia sp.	0.0	0.4	0.0	0.1	0.0	0.1
Tabellaria flocculosa var. *asterionelloides*	0.0	0.0	1.3	0.2	0.2	0.7
Tabellaria flocculosa	0.0	0.0	0.2	0.2	1.6	0.0
Synedra sp.	0.0	0.0	0.0	0.0	0.0	0.0
Navicula spp.	0.0	0.0	0.0	0.0	2.1	2.5
Pennates	0.7	1.0	0.1	0.2	6.1	5.6
Anabaena flos aquae	0.0	0.0	0.0	0.0	0.0	0.0
Aphanocapsa	0.0	0.0	0.0	0.0	0.0	0.0
Oscillatoria agardhii	0.0	0.0	0.0	0.0	0.0	0.0
Snowella cf lacustris	0.0	0.0	0.0	0.0	0.0	0.0
Woronichinia naegeliana	0.0	0.2	0.0	0.1	0.0	0.2
Ankistrodesmus fusiformis	0.0	0.0	0.0	0.0	0.0	0.0
Bitrichia longispina	0.0	0.0	0.0	0.0	0.0	0.0
Botryococcus braunii	0.0	0.0	0.0	0.0	0.0	0.0
Carteria sp.	0.3	1.2	0.0	0.1	0.0	0.0
Chlamydomonas sp.	1.6	4.7	0.4	0.0	3.1	0.4
Closterium abruptum	0.0	0.0	0.0	0.0	0.0	0.0
Closterium acutum var. *variabile*	1.4	0.6	1.7	1.4	1.5	4.4
Closterium gracile	0.6	0.0	0.0	0.0	0.0	0.0
Closterium kuetzingii	0.0	0.0	0.0	6.3	0.0	0.0
Closterium navicula	0.0	0.0	0.0	0.0	0.2	0.0
Crucigeniella rectangularis	0.0	0.0	0.0	0.0	0.0	0.0
Coelastrum microporum	0.0	0.0	0.0	0.0	0.0	0.0
Coenococcus planctonicus	0.0	0.0	0.0	0.0	0.0	0.1
Coenocuccus polycoccus	0.3	0.0	0.0	0.0	0.0	0.5
Cosmarium abbreviatum var. *planktonicum*	0.0	0.0	0.0	0.0	0.0	0.0
Cosmarium depressum	0.0	0.0	0.0	0.0	0.0	0.0
Cosmarium blyttii	0.0	0.0	0.0	0.0	0.0	0.0
Cosmarium humile	0.0	0.0	0.0	0.0	0.0	0.0
Chlorolobion braunii	0.0	0.0	0.0	0.0	0.0	0.0
Dictyosphaerium pulchellum	0.0	0.0	0.0	0.0	0.0	0.0
Kirchneriella obesa	0.0	0.0	0.0	0.0	0.0	0.0
Klebsormidium sp.	0.0	0.0	18.9	0.0	0.0	0.0
Monoraphidium contortum	0.0	0.0	0.1	0.1	0.1	0.1
Monoraphidium griffithii	0.1	0.1	0.2	0.2	1.0	0.5
Monoraphidium minutum	0.4	0.1	0.4	0.3	0.6	0.0
Mougeotia sp.	0.0	0.0	0.0	3.8	0.0	0.0
Oocystis parva	0.0	0.0	0.0	0.0	0.0	0.1
Phacus sp.	0.0	0.0	0.0	0.0	0.0	0.0
Scenedesmus granulatus	0.0	0.0	0.0	0.0	0.0	0.0
Single round cell	2.6	3.8	0.1	0.2	0.4	0.5
Pseudosphaerocystis lacustris	0.0	0.0	0.0	0.0	0.0	0.0
Spondylosium planum	0.0	0.0	0.0	0.0	0.0	0.0
Staurastrum anatinum	0.0	0.0	0.0	0.0	0.0	0.0
Staurastrum arctiscon	0.0	0.0	0.0	0.0	0.0	0.0
Staurastrum cingulum	0.0	0.0	0.0	0.9	0.0	0.0
Staurastrum lunatum	0.0	0.0	0.0	0.0	0.0	0.0
Staurodesmus sellatus	0.0	0.0	0.0	0.0	0.0	0.0
Tetraedron triangulare	0.0	0.0	0.0	0.0	0.0	0.0
Tetraedron minimum	4.0	2.0	2.0	0.9	0.0	1.0
Rhodomonas acuta	0.5	0.4	0.8	1.0	4.3	14.1
Rhodomonas minuta	0.7	1.2	0.3	0.2	0.6	0.6
Cryptomonas marssonii	0.0	0.0	0.0	0.0	0.0	0.0
Cryptomonas sp.	0.0	0.0	0.0	0.0	0.0	0.0
Chrysochromulina parva	1.7	0.0	0.0	0.0	0.6	0.8
Dinobryon sociale	0.0	0.0	0.0	0.0	0.0	0.0
Ochromonas tuberculata	0.0	0.0	0.0	0.0	0.0	0.0
Mallomonas akrokomos	0.0	0.0	0.0	0.0	0.0	0.0
Mallomonas caudata	4.3	0.0	0.0	0.0	22.6	0.0
Gymnodinium uberrimum	0.0	0.0	0.0	0.0	0.0	0.0
Gymnodinium triceratium	1.2	3.7	0.4	0.7	1.2	1.2
Ceratium hirudinella	0.0	0.0	0.0	0.0	0.0	0.0
Trachelomonas volvocina	0.0	0.0	0.0	0.1	0.0	0.0
Ciliates	34.3	28.6	44.0	65.9	14.3	18.5

Appendix F - Algal and ciliates density (cells mL^{-1}) for Guitane between May 2008 and April 2010 (n=12).

	19/05/09	11/06/09	1J/07/09	24/09/09	09/09/09	12/10/09	19/11/09	02/12/09	25/01/10	17/02/10	13/03/10	14/04/10
Asterionella formosa	68.8	98.3	16.8	4.0	0.0	0.0	1.3	36.0	8.3	0.4	24.8	2.4
Aulacoseira subarctica	23.0	0.0	0.0	0.0	0.0	0.0	2.0	0.0	0.0	9.7	9.8	0.0
Cyclotella spp.	33.8	31.3	171.6	248.4	305.2	118.8	0.0	2.5	7.8	14.5	61.7	69.0
Eunotia sp.	1.0	0.0	2.0	3.0	0.0	0.0	0.0	2.0	1.3	4.0	5.0	1.1
Rhizosolenia sp.	2.0	2.5	1.0	0.0	0.2	0.2	0.0	0.0	0.2	0.1	2.6	0.0
Tabellaria flocculosa	169.0	40.0	58.4	60.0	123.8	14.0	80.8	422.0	59.6	16.9	56.6	132.8
Fragilaria ulna	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Navicula sp.	0.0	0.0	0.0	23.0	3.2	0.8	0.0	0.5	0.0	0.0	0.0	0.0
Pennates	6.0	43.3	16.0	0.0	3.3	1.4	22.3	2.3	5.8	3.2	2.0	8.4
Anabaena flos aquae	12.0	110.0	114.0	102.0	65.2	25.4	9.5	0.0	0.0	0.4	0.0	4.0
Aphanocapsa	0.0	58016.7	4523.1	63168.1	0.0	0.0	964.0	1280.0	166.0	400.0	286.0	304.0
Aphanocapsa elastica	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	100.0	0.0	0.0
Aphanothece	940.0	16550.0	46479.2	13517.5	6758.7	558.0	0.0	0.0	0.0	0.0	8.0	0.0
Snowella lacustris	492.5	1183.3	7570.0	4020.0	1156.0	594.2	11.8	350.0	146.0	55.0	82.0	124.0
Merismopedia tenuissima	18.0	2550.0	6779.5	9831.3	10697.0	1632.5	29.6	410.0	11.6	5.5	14.4	6.8
Oscillatoria agardhii	371.0	6.1	120.0	271.8	88.3	147.5	160.0	315.0	195.3	500.0	201.3	707.4
Ankistrodesmus fusiformis	0.0	0.0	0.0	0.0	2.8	0.8	0.8	2.0	0.0	0.0	0.0	0.8
Bitrichia sp.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.5
Botryococcus braunii	53.2	130.0	233.5	114.0	141.2	126.6	35.2	2.3	0.0	25.9	124.6	33.2
Botryosphaerella sudetica	0.0	0.0	0.0	0.0	60.0	42.0	0.0	0.0	0.0	0.0	0.0	0.0
Closteriopsis aciculare	0.0	25.0	0.0	5.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	10.4
Closterium kuetzingii	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.0	0.2	0.1
Closterium navicula	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.0	0.0	0.0
Chlamydomonas sp.	0.0	0.0	0.0	0.0	0.0	13.0	0.0	2.6	23.4	0.0	0.0	0.0
Coelastrum microporum	10.8	0.0	0.0	0.0	2.0	2.5	0.0	0.0	3.2	0.0	0.0	0.0
Cosmarium cf *tinctum*	1.2	6.7	4.0	3.0	2.8	1.2	1.7	0.8	0.0	0.6	0.6	0.0
Cosmarium quadrifarium	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.0	0.0	0.0	0.0
Cosmarium contractum	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.2	0.0	4.8
Cosmarium depressum	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.2	0.0
Crucigeniella crucifera	0.0	6.7	7.0	28.0	0.0	7.0	0.0	2.4	0.0	0.0	0.0	0.0
Crucigenia tetrapedia	84.0	1118.3	481.0	439.0	1403.7	203.8	85.2	70.2	75.8	118.7	96.0	24.8
Crucigenia rectangularis	0.0	0.0	0.0	0.0	8.0	0.0	6.5	2.5	0.0	2.4	6.4	1.6
Dictyosphaerium pulchellum	0.8	5.6	8.0	14.0	29.6	3.6	0.0	7.2	0.0	0.0	0.0	3.7
Euastrum binale	0.0	0.0	0.0	1.0	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Euastrum dubium	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.0	0.0	0.0	0.0
Euastrum pinnatum	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Eudorina sp.	8.0	0.0	20.0	1.0	13.0	0.4	1.0	0.0	0.4	0.0	0.0	0.0
Monoraphidium contortum	0.0	10.4	41.6	5.2	5.2	7.8	33.8	5.2	0.0	20.8	5.2	18.2
Monoraphidium griffithii	0.0	0.0	0.0	0.0	2.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Monoraphidium minutum	67.5	130.0	106.6	223.6	119.6	70.2	2.5	10.4	18.5	2.5	7.8	80.6
cf *Mougeotia*	5.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Oocystis parva/lacustris	6.0	25.0	18.0	8.0	1.6	1.6	2.8	0.2	4.0	1.6	1.6	0.8
Quadrigula closterioides	27.0	170.0	67.0	20.0	10.6	10.8	2.3	7.3	2.8	2.0	13.0	5.8
Radiococcus planktonicus	11.0	711.7	140.0	14.0	0.0	4.8	2.5	4.0	0.0	0.0	3.2	32.8
Scenedesmus dimorphus	0.0	6.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Scenedesmus ecornis	0.0	0.0	28.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Scenedesmus granulatus	4.0	6.7	0.0	25.0	19.2	15.8	8.3	13.5	8.8	3.7	8.4	2.4
Scenedesmus acutus	0.0	0.0	0.0	0.0	0.0	0.0	0.0	7.2	0.0	0.0	3.4	0.0
Scenedesmus subspicata	0.0	0.0	0.0	0.0	0.0	0.0	1.5	0.0	0.0	2.9	0.0	0.0
Pseudosphaerocystis lacustris	28.0	216.7	366.0	25.0	10.4	8.0	0.0	1.6	0.0	0.0	0.0	7.2
Spondylosium planum	8.0	21.7	13.0	24.0	16.4	9.6	5.3	0.0	0.0	0.0	0.0	0.2
Staurastrum anatinum	0.3	0.0	1.0	0.0	0.0	0.0	0.2	0.0	0.0	0.0	0.0	0.3
Staurastrum arctiscon	0.3	0.1	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Staurastrum cingulum	0.4	0.0	2.0	0.0	0.0	0.4	0.3	0.0	0.0	0.1	0.0	0.0
Staurodesmus incus	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.0	0.0	0.0	0.0
Staurodesmus subulatus	0.0	1.0	2.6	2.0	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0
Staurodesmus triangularis	0.0	0.0	1.0	0.0	0.6	0.6	0.0	0.2	0.0	0.0	0.0	0.0
Tetraedron minimum	7.8	0.0	0.0	0.0	5.2	10.4	0.0	0.0	18.2	0.0	2.6	5.2
Round single cells (smal)	0.0	126.7	735.7	4.0	5.4	4.8	0.0	3.2	2.8	0.0	0.0	3.8
Round single cell (bigger size)	0.0	0.0	0.0	645.0	111.8	104.0	0.8	26.0	44.2	49.3	330.1	109.2
Chroomonas/Rhodomonas minuta	70.3	376.9	98.8	179.4	39.0	15.6	0.0	36.4	0.0	2.5	522.5	130.0
Chroomonas/Rhodomonas acuta	177.5	223.6	759.1	104.0	179.4	434.1	361.3	275.5	265.2	532.5	210.6	447.1
Cryptomonas marsonii	0.0	0.0	59.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Cryptomonas sp.	2.5	26.0	41.6	5.2	15.6	26.0	0.0	5.2	5.2	0.0	2.6	0.0
Chrysochromulina parva	102.5	280.7	85.8	278.0	7.8	98.8	33.8	41.6	2.5	101.5	0.0	109.2
Dinobryon bavaricum	0.0	0.0	0.0	0.0	1.2	0.2	0.0	0.6	0.0	0.0	0.0	0.0
Dinobryon sertularia	7.0	23.0	15.0	4.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0
Mallomonas akrokomos	7.3	10.5	0.0	1.0	1.0	5.8	1.2	0.6	0.4	0.0	0.0	0.2
Mallomonas caudata	1.0	0.0	0.0	1.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	2.6
Ceratium hirudinella	0.5	0.1	0.3	0.3	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.1
Gymnodinium triceratium	0.4	2.5	3.0	4.0	13.0	2.6	2.5	1.0	3.2	2.6	10.4	2.6
Gymnodinium uberrimum	0.3	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.0	0.0	0.4	0.2
Trachelomonas	0.0	0.0	0.0	2.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Phacus striatus	0.0	0.0	0.0	0.3	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0
Ciliates	8.0	9.0	11.5	12.0	6.8	9.0	11.8	8.5	5.8	7.3	4.9	5.0

Appendix G - Algal and ciliates biovolume (μm^3) and biomass ($mm^3\ m^{-3}$) for Guitane between May '08 and April '10 (n=12).

	Biovolume	19/05/09	11/06/09	11/07/09	24/09/09	09/09/09	12/10/09	19/11/09	02/12/09	25/01/10	17/02/10	13/03/10	14/04/10
Asterionella formosa	342.6	23.6	33.7	5.8	1.4	0.0	0.0	0.4	12.3	2.8	0.1	8.5	0.8
Aulacoseira subarctica	616.6	16.5	0.0	0.0	0.0	0.0	0.0	1.5	0.0	0.0	7.3	7.4	0.0
Cyclotella spp.	541.2	34.9	32.3	87.3	108.5	132.0	43.1	0.0	2.6	8.1	8.8	43.9	26.2
Eunotia sp.	950.0	1.0	0.0	1.9	2.9	0.0	0.0	0.0	1.9	1.2	3.8	4.8	1.0
Rhizosolenia sp.	1041.6	2.1	2.6	1.0	0.0	0.2	0.2	0.0	0.0	0.2	0.1	2.7	0.0
Tabellaria flocculosa	628.9	140.8	35.6	42.6	32.4	42.9	12.5	29.3	222.7	46.6	11.8	48.1	100.6
Fragilaria ulna	5402.9	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Navicula sp.	471.4	0.0	0.0	0.0	10.8	1.5	0.4	0.0	0.2	0.0	0.0	0.0	0.0
Pennates	382.5	2.3	16.6	6.1	0.0	1.2	0.5	8.5	0.9	2.2	1.2	0.8	3.2
Anabaena flos aquae	174.7	2.1	19.2	19.9	17.8	11.4	4.4	1.7	0.0	0.0	0.1	0.0	0.7
Aphanocapsa	0.4	0.0	23.2	1.8	25.3	0.0	0.0	0.4	0.5	0.1	0.2	0.1	0.1
Aphanocapsa elastica	4.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0
Aphanothece	0.6	0.6	9.9	27.9	8.1	4.1	0.3	0.0	0.0	0.0	0.0	0.0	0.0
Snowella lacustris	7.7	3.8	9.1	58.3	31.0	8.9	4.6	0.1	2.7	1.1	0.4	0.6	1.0
Merismopedia tenuissima	7.0	0.1	17.9	47.5	68.8	74.9	11.4	0.2	2.9	0.1	0.0	0.1	0.0
Oscillatoria agardhii	79.5	29.5	0.5	9.5	21.6	7.0	11.7	12.7	25.0	15.5	39.8	16.0	56.2
Ankistrodesmus fusiformis	19.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Bitrichia sp.	328.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.8
Botryococcus braunii	1.8	0.1	0.2	0.4	0.2	0.3	0.2	0.1	0.0	0.0	0.0	0.2	0.1
Botryosphaerella sudetica	25.9	0.0	0.0	0.0	0.0	1.6	1.1	0.0	0.0	0.0	0.0	0.0	0.0
Closteriopsis aciculare	34.0	0.0	0.9	0.0	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.4
Closterium kuetzingii	31586.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	6.3	0.0	0.0	6.3	3.2
Closterium navicula	2308.2	0.0	0.0	0.0	0.0	0.0	0.0	0.9	0.0	0.0	0.0	0.0	0.0
Chlamydomonas sp.	377.0	0.0	0.0	0.0	0.0	0.0	4.9	0.0	1.0	8.8	0.0	0.0	0.0
Coelastrum microporum	258.0	2.8	0.0	0.0	0.0	0.5	0.6	0.0	0.0	0.8	0.0	0.0	0.0
Cosmarium cf *tinctum*	3971.0	4.8	26.5	15.9	11.9	11.1	4.8	6.8	3.2	0.0	2.4	2.4	0.0
Cosmarium quadrifarium	33208.0	0.0	0.0	0.0	0.0	0.0	0.0	6.6	0.0	0.0	0.0	0.0	0.0
Cosmarium contractum	14978.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	3.0	3.0	0.0	71.9
Cosmarium depressum	184.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Crucigeniella crucifera	34.2	0.0	0.2	0.2	1.0	0.0	0.2	0.0	0.1	0.0	0.0	0.0	0.0
Crucigenia tetrapedia	16.4	1.4	18.3	7.9	7.2	23.0	3.3	1.4	1.2	1.2	1.9	1.6	0.4
Crucigenia rectangularis	34.2	0.0	0.0	0.0	0.0	0.3	0.0	0.2	0.1	0.0	0.1	0.2	0.1
Dictyosphaerium pulchellum	114.6	0.1	0.6	0.9	1.6	3.4	0.4	0.0	0.8	0.0	0.0	0.0	0.4
Euastrum binale	14.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Euastrum dubium	209.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Euastrum pinnatum	209.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Eudorina sp.	384.0	3.1	0.0	7.7	0.4	5.0	0.2	0.4	0.0	0.2	0.0	0.0	0.0
Monoraphidium contortum	13.0	0.0	0.1	0.5	0.1	0.1	0.1	0.4	0.1	0.0	0.3	0.1	0.2
Monoraphidium griffithii	21.1	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Monoraphidium minutum	109.9	7.4	14.3	11.7	24.6	13.1	7.7	0.3	1.1	2.0	0.3	0.9	8.9
cf *Mougeotia*	7620.4	38.1	0.0	0.0	1.9	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Oocystis parva/lacustris	192.9	1.2	4.8	3.5	1.5	0.3	0.3	0.5	0.0	0.8	0.3	0.3	0.2
Quadrigula closterioides	30.4	0.8	5.2	2.0	0.6	0.3	0.3	0.1	0.2	0.1	0.1	0.4	0.2
Radiococcus planktonicus	20.7	0.2	14.7	2.9	0.3	0.0	0.1	0.1	0.1	0.0	0.0	0.1	0.7
Scenedesmus dimorphus	11.7	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Scenedesmus ecornis	58.9	0.0	0.0	1.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Scenedesmus granulatus	38.1	0.2	0.3	0.0	1.0	0.7	0.6	0.3	0.5	0.3	0.1	0.3	0.1
Scenedesmus acutus	54.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.2	0.0
Scenedesmus subspicata	59.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.2	0.0	0.0
Pseudosphaerocystis lacustris	209.8	5.9	45.5	76.8	5.2	2.2	1.7	0.0	0.3	0.0	0.0	0.0	1.5
Spondylosium planum	1184.6	9.5	25.7	15.4	28.4	19.4	11.4	6.2	0.0	0.0	0.0	0.0	0.2
Staurastrum anatinum	9505.7	2.9	0.0	9.5	0.0	0.0	0.0	1.9	0.0	0.0	0.0	0.0	2.9
Staurastrum arctiscon	25344.0	7.6	2.5	0.0	6.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Staurastrum cingulum	10977.1	4.4	0.0	22.0	0.0	0.0	4.4	2.7	0.0	0.0	1.1	0.0	0.0
Staurodesmus incus	15500.0	0.0	0.0	0.0	0.0	0.0	0.0	3.1	0.0	0.0	0.0	0.0	0.0
Staurodesmus subulatus	5531.1	0.0	5.5	14.4	11.1	0.0	0.0	0.0	0.0	0.0	0.0	2.2	0.0
Staurodesmus triangularis	8686.8	0.0	0.0	8.7	0.0	5.2	5.2	0.0	1.7	0.0	0.0	0.0	0.0
Tetraedron minimum	86.3	0.7	0.0	0.0	0.0	0.4	0.9	0.0	0.0	1.6	0.0	0.2	0.4
Round single cells (smal)	61.3	0.0	7.8	45.1	0.2	0.3	0.3	0.0	0.2	0.2	0.0	0.0	0.2
Round single cell (bigger size)	142.8	0.0	0.0	0.0	92.1	16.0	14.8	0.1	3.7	6.3	7.0	47.1	15.6
Chroomonas/Rhodomonas minuta	45.0	3.2	17.0	4.4	8.1	1.8	0.7	0.0	1.6	0.0	0.1	23.5	5.8
Chroomonas/Rhodomonas acuta	98.0	17.4	21.9	74.4	10.2	17.6	42.5	35.4	27.0	26.0	52.2	20.6	43.8
Cryptomonas marsonii	1410.0	0.0	0.0	84.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Cryptomonas sp.	1939.9	4.8	50.4	80.7	10.1	30.3	50.4	0.0	10.1	10.1	0.0	5.0	0.0
Chrysochromulina parva	66.4	6.8	18.6	5.7	18.5	0.5	6.6	2.2	2.8	0.2	6.7	0.0	7.2
Dinobryon bavaricum	218.5	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.1	0.0	0.0	0.0	0.0
Dinobryon sertularia	146.4	1.0	3.4	2.2	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Mallomonas akrokomos	153.0	1.1	1.6	0.0	0.2	0.2	0.9	0.2	0.1	0.1	0.0	0.0	0.0
Mallomonas caudata	1195.0	1.2	0.0	0.0	1.2	0.0	0.3	0.0	0.0	0.0	0.0	0.0	3.1
Ceratium hirudinella	61348.9	30.7	6.1	15.3	15.3	12.3	0.0	0.0	0.0	0.0	0.0	0.0	6.1
Gymnodinium triceratium	1187.6	0.5	3.0	3.6	4.8	15.4	3.1	3.0	1.2	3.8	3.1	12.3	3.1
Gymnodinium uberrimum	57808.0	14.5	0.0	0.0	0.0	0.0	0.0	11.6	0.0	0.0	0.0	23.1	11.6
Trachelomonas	571.2	0.0	0.0	0.0	1.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Phacus striatus	2746.0	0.0	0.0	0.0	0.7	0.0	0.7	0.0	0.0	0.0	0.0	0.0	0.0
Ciliates	4486.1	26.6	70.7	75.7	63.8	11.5	14.0	25.9	21.7	23.8	34.2	55.7	12.6

Appendix H - Picoplankton density (cell 10^3 mL^{-1}), mean cell biovolume for each sample (μm^3) and total biomass (mm^3 m^{-3}) for Feeagh and Guitane between May 2009 and April 2010 (n=24 and 12, respectively).

	Feeagh		
	Density cell 10^3 mL^{-1}	Biovolume μm^3	Biomass mm^3 m^{-3}
12May2009	28.4	0.8	23.0
25May2009	35.5	0.8	28.8
11June2009	34.0	0.8	27.6
22June2009	138.6	0.8	110.9
06July2009	56.2	0.8	45.6
22July2009	45.1	0.8	36.6
04Aug2009	64.7	0.8	51.7
17Aug2009	114.4	0.8	92.7
27Aug2009	85.3	0.8	68.2
07Sep2009	93.5	0.8	76.7
01Oct2009	74.0	0.8	59.2
22Oct2009	84.0	0.8	67.2
06Nov2009	63.3	0.8	48.1
20Nov2009	92.0	0.7	63.5
04Dec2009	47.6	0.7	33.3
22Dec2009	21.3	0.7	13.9
06Jan2010	25.6	0.6	16.4
20Jan2010	54.0	0.8	42.1
02Feb2010	60.4	0.8	46.2
11Feb2010	43.4	0.7	28.6
05Mar2010	61.8	0.8	47.0
15Mar2010	30.6	0.8	22.9
07Apr2010	36.0	0.8	29.2
19Apr2010	42.7	0.8	34.6

	Guitane		
	Density cell 10^3 mL^{-1}	Biovolume μm^3	Biomass mm^3 m^{-3}
	Density cell 10^3 mL^{-1}	Biovolume μm^3	Biomass mm^3 m^{-3}
19May09	67.7	0.9	60.9
11June09	75.0	0.9	67.5
1July09	230.0	0.9	207.0
24Aug09	113.0	0.9	101.7
9Sep09	70.4	0.9	63.3
12Oct09	134.0	0.9	120.6
19Nov09	29.9	0.8	23.3
02Dec09	19.2	0.9	16.7
25Jan10	27.7	0.8	22.5
17Feb10	19.2	0.8	14.6
13Mar10	24.5	0.9	22.1
14Apr10	103.9	0.9	93.5

Appendix I - Bacterioplankton density (cell 10^3 mL^{-1}), mean cell biovolume for each sample (μm^3) and total biomass (mm^3 m^{-3}) for Feeagh and Guitane between May 2009 and April 2010 (n=24 and 12, respectively).

	Feeagh		
	Density cell 10^3 mL^{-1}	Biovolume μm^3	Biomass mm^3 m^{-3}
12May2009	1323.6	0.12	152.9
25May2009	1210.1	0.09	113.2
11June2009	1388.7	0.09	118.9
22June2009	2110.3	0.10	218.0
06July2009	2113.8	0.10	211.1
22July2009	4014.8	0.08	323.9
04Aug2009	1711.1	0.08	130.2
17Aug2009	2318.5	0.10	222.8
27Aug2009	2593.2	0.06	167.4
07Sep2009	2315.1	0.08	196.0
01Oct2009	1497.9	0.08	117.1
22Oct2009	1716.3	0.12	213.9
06Nov2009	1762.3	0.09	167.4
20Nov2009	1777.7	0.10	171.0
04Dec2009	1295.5	0.09	122.9
22Dec2009	1784.2	0.09	166.7
06Jan2010	1535.4	0.07	106.2
20Jan2010	1435.5	0.07	106.1
02Feb2010	1258.2	0.08	99.0
11Feb2010	1279.5	0.07	93.3
05Mar2010	1354.2	0.07	91.0
15Mar2010	1579.4	0.07	111.4
07Apr2010	858.3	0.07	63.4
19Apr2010	939.5	0.07	66.2

	Guitane		
	Density cell 10^3 mL^{-1}	Biovolume μm^3	Biomass mm^3 m^{-3}
	Density cell 10^3 mL^{-1}	Biovolume μm^3	Biomass mm^3 m^{-3}
19May09	1440.4	0.08	114.1
11June09	995.6	0.13	133.4
1July09	931.9	0.10	92.7
24Aug09	1110.0	0.10	106.4
9Sep09	1224.7	0.10	117.1
12Oct09	1045.5	0.08	87.9
19Nov09	1423.5	0.05	71.2
02Dec09	1666.0	0.07	116.8
25Jan10	929.8	0.08	76.4
17Feb10	1099.4	0.08	89.5
13Mar10	1330.2	0.11	146.0
14Apr10	928.8	0.10	88.8

Appendix J - Daily (g DW m^{-2} d^{-1}) and total (g DW m^{-2}) sediment deposition, LOI_{550} (%); TOC (%), TN (%) and C/N ratio at inflow, deepest and outflow traps and in Feeagh between 1st April 2009 and 8th February 2011 and in Guitane between 9th May 2009 and 19th January 2011 and at surface sediments (0 – 1 cm) (* = 27Aug-01Oct for Inflow; n.d. = no data).

	Feeagh										Guitane			
	01/04-26/05/09	26/05-22/07/09	22/07-01/10/09 (*)	01/10-20/11/09	20/11/09-20/01/10	20/01-19/03/10	19/03-02/06/10	02/06-22/07/10	22/07/10-02/02/11	**Surface sediment**	19/05/09-25/01/10	25/01-14/07/10	14/01/10-19/01/11	**Surface sediment**
Daily sediment deposition (g DW m^{-2} d^{-1})														
Inflow	2.3	7.9	3.2	5.6	6.1	1.6	4.5	2.0	3.9		1.5	0.5	1.0	
Deepest	2.6	6.9	4.6	4.9	5.4	0.8	4.3	1.8	3.6		n.d.	0.5	0.6	
Outflow	1.6	5.0	2.9	2.7	4.4	0.6	2.5	1.6	2.3		1.5	0.3	0.7	
Total sediment deposition rate (g DW m^{-2})														
Inflow	123.8	452.3	111.7	278.9	372.3	95.5	337.2	101.0	783.8		380.8	77.2	189.5	
Deepest	141.8	395.0	328.5	244.5	331.0	48.1	318.8	90.7	728.3		n.d.	83.5	115.1	
Outflow	87.9	287.4	204.2	136.3	269.9	37.2	188.7	78.9	459.7		388.9	57.3	137.5	
LOI_{550} (%)														
Inflow	37.7	12.6	22.4	20.5	27.5	26.2	19.3	28.5	20.7	32.0	29.9	28.3	28.2	
Deepest	42.3	15.1	19.4	26.0	28.8	36.1	23.4	30.8	24.2	32.0	n.d	29.1	22.9	21.
Outflow	40.2	16.3	24.7	32.4	34.6	40.0	26.4	30.5	31.5	45.5	29.1	30.6	30.6	
TOC (%)														
Inflow	9.2	4.9	6.9	9.3	11.3	12.9	10.4	9.4		14.3	15.3	12.4		
Deepest	18.2	5.6	8.9	12.7	12.7	12.5	10.4	11.2		13.4	n.d	13.4		7
Outflow	17.1	5.6	15.2	14.3	13.8	17.4	12.3	8.8		19.1	14.1	13.2		
TN (%)														
Inflow	0.5	0.3	0.4	0.5	0.7	0.7	0.6	0.6		0.7	1.1	1		
Deepest	0.9	0.4	0.5	0.8	0.7	0.7	0.6	0.8		0.8	n.d	1.1		0.5
Outflow	0.9	0.4	1.3	1.2	0.8	0.9	0.8	0.6		0.9	1.1	1.1		
C/N ratio														
Inflow	16.7	15.9	17.5	19.7	16.0	18.8	17.1	15.6		20.4	13.4	12.0		
Deepest	20.2	14.8	16.3	15.9	18.3	18.3	16.4	14.4		16.1	n.d	12.0		13.
Outflow	18.2	14.3	11.8	11.5	18.1	18.6	15.7	15.2		21.1	13.3	11.9		

Appendix K - Pigment concentrations (nmol g^{-1}) of the identified taxonomic groups and respective pigment types measured in sediment traps and surface sediment from inflow, deepest and outflow stations in Feeagh between November 2009 and July 2010 (n=5) and in Guitane between May 2009 and July 2010 (n=2).

		Feeagh															**Guitane**						
		Inflow					**Deepest**					**Outflow**					**Inflow**		**Deepest**			**Outflow**	
		20Nov09-20Jan10	20Jan10-19Mar10	19Mar10-02Jun10	02Jun10-22Jul10	**Surface sediment**	20Nov09-20Jan10	20Jan10-19Mar10	19Mar10-02Jun10	02Jun10-22Jul10	**Surface sediment**	20Nov09-20Jan10	20Jan10-19Mar10	19Mar10-02Jun10	02Jun10-22Jul10	**Surface sediment**	19May2009-25Jan10	25Jan10-14July2010	19May2009-25Jan10	25Jan10-14July2010	**Surface sediment**	19May2009-25Jan10	25Jan10-14July2010
All algae and plantae	Chl-*a*	6.1	32.3	38.6	46.6	123.7	5.9	37.3	44.0	50.4	137.6	1.9	6.9	8.6	35.3	52.7	19.8	14.1	n.d.	206.0	9.0	21.2	123.2
	Chl-*a'*	1.1	6.0	5.2	8.3	20.7	0.7	6.4	2.2	7.8	17.0	0.6	1.2	7.8	6.2	15.9	1.1	7.4	n.d.	21.5	13.6	2.0	9.6
	Pheophytin *a*	9.8	19.8	22.6	30.1	82.3	15.3	16.6	14.5	41.3	87.7	16.8	44.2	7.8	30.0	98.9	5.4	29.7	n.d.	39.5	24.0	9.4	40.5
	Pheophorbide *a'*	5.3	21.5	33.2	37.2	97.2	4.0	23.5	43.5	174.0	245.0	7.5	25.3	30.4	34.0	97.1	3.5	13.6	n.d.	17.6	24.6	7.6	32.6
	Chl *c2*	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.7	24.3	27.0	0.0	0.0	0.0	0.0	0.0	43.1	55.4	n.d.	41.5	53.4	8.2	324.0
Chloro, Eugl,	Chl-*b*	2.9	5.2	9.5	6.2	23.7	4.1	6.1	3.9	13.0	27.1	1.9	3.4	1.5	6.9	13.7	1.4	25.0	n.d.	36.1	15.8	5.3	22.9
	Pheophytin *b*	26.2	26.4	45.2	34.9	132.7	36.1	28.2	14.7	26.8	105.7	38.6	35.9	14.9	32.1	121.4	2.8	5.1	n.d.	4.2	5.5	5.0	21.5
Chloro/Cyano	Lutein/Zeaxanthin	2.3	8.9	8.6	19.6	39.3	1.9	9.0	8.9	23.8	43.6	1.0	6.0	2.0	9.6	18.6	7.3	0.0	n.d.	0.0	55.7	11.1	0.0
Cyanobacteria	Canthaxanthin	0.0	0.0	0.0	1.8	1.8	0.0	0.0	0.0	3.1	3.1	0.0	0.0	0.0	1.8	1.8	0.0	0.0	n.d.	0.0	8.4	0.0	0.0
Diatoms, Dino-, Chrysophyta	Fucoxanthin	5.5	22.0	38.5	29.0	94.9	9.4	23.4	61.7	32.9	127.4	6.8	12.3	44.4	27.1	90.5	9.4	62.4	n.d.	193.8	68.0	30.1	9.6
	Diatoxanthin	0.4	1.3	2.2	7.8	11.7	0.6	1.5	3.0	25.7	30.7	0.9	2.2	0.9	11.5	15.5	0.8	9.2	n.d.	17.8	7.2	1.2	6.7
Cryptophyta	Alloxanthin	0.4	0.0	3.1	11.5	15.1	1.3	0.0	6.2	45.7	53.1	2.0	2.4	4.0	12.5	20.8	19.8	57.0	n.d.	55.5	27.8	8.1	172.9
	UV-abs. comp.	0.0	0.0	0.0	0.0	0.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.8	57.5	117.5	n.d.	385.7	18.2	79.9	133.1

Appendix L - Relative abundance (%) of planktonic and benthic diatom taxa, relative abundance of fossil diatoms (≥ 1% abundance), diatom accumulation (valves 10^3 d^{-1} cm^{-2}) in trap samples (grey columns) and diatom concentration (valves 10^3 g^{-1}) of surface sediment in sediment traps sediment trap samples (1st April 2009 – 22nd July 2010; n=8) and surface sediment (0-1 cm; n=3) from the inflow, deepest and outflow stations in Feeagh.

		Planktonic	**Benthic**	*Achnanthes lanceolata* var. *frequentissima*	*Achnanthes oblongella*	*Achnanthes subatomoides*	*Achnanthidium minutissimum*	*Asterionella formosa*	*Aulacoseira alpigena*	*Aulacoseira subarctica*	*Brachysira garrensis*	*Brachysira vitrea*	*Cavinula cocconeiformis*	*Cocconeis neothumensis*	*Cocconeis placentula*	*Cyclotella radiosa*	*Cyclotella kuetzingiana*	*Cymbella silesiaca*	*Denticula tenuis*
Inflow	1Apr-26May09	59.2	35.3	0.0	3.6	1.4	9.2	20.3	18.6	7.2	0.0	1.9	0.0	0.0	1.0	0.6	0.0	1.9	0.0
	26May-22Jul09	73.6	22.7	0.3	3.2	2.5	3.9	49.6	4.1	5.2	0.0	0.3	0.0	0.0	0.9	0.8	0.4	0.9	0.0
	27Aug-01Oct09	36.6	59.5	0.9	12.1	1.8	9.7	7.3	12.1	2.4	1.8	0.6	1.2	1.8	0.3	1.2	0.0	1.8	0.6
	01Oct-20Nov09	41.0	50.2	1.4	5.3	0.0	8.0	12.9	9.4	3.7	1.4	0.0	0.5	1.2	0.7	0.5	0.0	0.5	0.0
	20Nov-20Jan10	37.3	57.8	0.3	9.7	2.7	10.5	8.9	7.8	7.8	1.9	1.2	0.5	0.0	2.6	0.5	1.9	2.7	0.0
	20Jan-19Mar10	38.4	53.7	0.3	8.1	3.0	11.0	4.0	17.0	11.5	0.0	0.3	1.5	0.0	0.0	0.5	1.3	2.0	0.0
	19Mar-02Jun10	61.4	35.1	0.0	2.5	0.2	5.3	30.9	12.2	12.0	0.0	0.4	0.0	0.0	0.1	0.0	0.0	2.1	0.0
	02Jun-22Jul10	69.5	27.5	0.5	3.9	2.3	6.8	58.0	1.8	1.1	0.0	0.0	0.2	0.0	1.1	3.9	0.0	0.8	0.0
	Surface sediment	52.4	41.6	0.5	8.2	1.0	8.9	33.0	4.0	9.7	0.0	1.5	0.5	0.0	0.0	0.0	0.5	1.5	0.0
Deepest	1Apr-26May09	63.5	34.6	0.5	4.1	2.2	4.8	28.4	14.0	12.7	0.0	1.7	1.0	1.0	0.7	0.0	0.0	1.0	0.0
	26May-22Jul09	87.2	11.9	0.0	2.1	0.6	2.6	71.9	7.2	1.7	0.0	0.9	0.0	0.0	0.4	0.0	0.0	0.4	0.0
	22Jul-01Oct09	40.9	55.1	0.8	10.6	0.5	7.0	9.9	7.0	6.2	0.0	2.9	0.3	3.2	0.3	3.4	0.0	1.3	0.5
	01Oct-20Nov09	38.6	57.9	0.3	9.3	3.0	10.2	8.9	13.7	5.8	1.3	1.3	2.5	1.3	0.5	0.5	0.0	0.5	0.0
	20Nov-20Jan10	30.3	63.8	0.5	7.4	3.2	12.2	8.1	4.1	8.8	0.2	0.5	0.7	1.8	1.6	0.6	1.6	4.3	0.0
	20Jan-19Mar10	38.3	56.1	0.0	9.6	2.5	14.7	7.6	17.6	8.6	0.0	0.8	0.4	2.5	0.6	0.4	0.4	1.4	0.0
	19Mar-02Jun10	66.0	30.0	0.0	3.4	0.6	9.1	34.5	10.9	13.5	0.0	0.4	0.0	0.0	0.6	0.0	0.0	1.7	3.5
	02Jun-22Jul10	82.6	16.9	0.0	1.9	1.0	6.8	65.3	8.7	4.8	0.2	0.5	0.0	0.2	0.2	1.6	0.0	0.0	0.7
	Surface sediment	36.7	56.9	0.0	9.3	0.8	15.0	18.5	4.1	6.5	1.2	0.6	0.6	0.4	1.2	0.6	0.2	2.4	0.0
Outflow	1Apr-26May09	59.2	30.9	0.0	4.3	1.8	5.5	25.7	13.8	14.0	0.0	0.0	0.0	0.0	0.9	0.0	0.0	1.5	0.0
	26May-22Jul09	73.6	15.7	0.0	4.0	0.0	5.8	71.7	2.7	0.4	0.0	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.4
	22Jul-01Oct09	73.6	58.5	0.0	10.3	2.6	11.2	9.0	5.6	5.6	1.5	0.0	0.7	1.1	1.9	2.4	0.0	0.7	0.0
	01Oct-20Nov09	73.6	48.0	0.5	5.7	1.0	15.4	13.4	18.2	8.3	0.3	1.0	0.5	1.8	0.8	0.0	0.0	2.1	0.5
	20Nov-20Jan10	73.6	64.2	1.0	10.5	3.1	17.3	4.8	7.8	13.0	0.3	2.3	0.5	0.5	3.3	0.9	0.3	0.6	0.0
	20Jan-19Mar10	73.6	63.1	0.3	11.6	0.8	18.0	4.5	6.3	13.2	0.5	1.1	0.8	1.1	1.1	2.4	0.0	2.9	0.0
	19Mar-02Jun10	73.6	29.0	0.2	2.6	0.9	5.9	26.6	22.5	13.5	0.0	1.6	0.3	0.3	0.9	0.0	0.0	0.7	2.8
	02Jun-22Jul10	73.6	27.0	0.2	2.5	0.5	12.1	66.3	0.5	1.0	0.5	0.2	0.0	1.2	0.2	0.6	0.0	1.7	0.7
	Surface sediment	16.9	82.6	1.2	8.9	1.6	13.1	17.5	0.5	6.5	1.2	1.2	1.6	0.7	0.2	1.4	1.4	0.0	4.2

Appendix L continues - Relative abundance (%) of planktonic and benthic diatom taxa, relative abundance of fossil diatoms (≥ 1% abundance), diatom accumulation (valves 10^3 d^{-1} cm^{-2}) in trap samples (grey columns) and diatom concentration (valves 10^3 g^{-1}) of surface sediment in sediment traps sediment trap samples (1st April 2009 – 22nd July 2010; n=8) and surface sediment (0-1 cm; n=3) from the inflow, deepest and outflow stations in Feeagh.

		Diatoma moniliformis	*Diatoma tenue*	*Eunotia bilunaris*	*Eunotia exigua*	*Eunotia implicata*	*Eunotia incisa*	*Eunotia pectinalis*	*Eunotia paludosa*	*Eunotia rhomboidea*	*Fragilaria capucina*	*Fragilaria capucina* var. *rumpens*	*Fragilaria capucina* var. *vaucheriae*	*Fragilaria exigua*	*Fragilaria gracilis*	*Frustulia rhomboides* var. *saxonica*	*Frustulia rhomboides* var. *viridula*	*Gomphonema gracile*	*Gomphonema minutum*	*Gomphonema olivaceoides*
Inflow	1Apr-26May09	0.0	0.0	0.0	0.0	0.0	1.5	0.5	1.0	0.0	5.2	0.5	1.2	0.0	6.7	0.7	0.0	0.0	2.9	0.0
	26May-22Jul09	0.0	0.3	0.0	0.0	0.0	0.5	0.3	0.6	0.0	0.0	1.9	3.2	0.6	0.3	0.0	0.0	0.0	0.0	0.9
	27Aug-01Oct09	0.0	0.0	1.8	0.6	0.0	0.9	1.2	1.8	0.0	4.8	0.6	0.0	4.8	4.5	0.2	1.2	0.6	0.6	0.6
	01Oct-20Nov09	1.4	0.0	0.0	2.3	2.3	0.5	2.3	2.8	0.5	4.7	0.0	0.9	1.4	5.4	0.0	1.6	1.2	0.5	0.0
	20Nov-20Jan10	0.0	0.0	0.0	0.5	0.5	3.2	1.3	1.1	0.5	3.2	0.0	1.6	4.1	1.1	1.3	0.0	0.8	0.5	0.5
	20Jan-19Mar10	0.0	0.8	0.0	1.0	0.8	4.4	0.8	0.3	1.5	1.0	1.8	1.3	5.8	0.0	1.4	0.0	0.5	0.5	2.5
	19Mar-02Jun10	0.0	1.6	0.0	0.4	0.0	1.3	0.0	0.0	0.2	1.7	4.4	5.1	0.8	0.0	0.6	0.0	0.8	0.4	2.7
	02Jun-22Jul10	0.2	1.6	0.0	0.0	0.0	0.0	0.2	0.7	0.5	0.0	3.0	0.5	0.5	1.8	0.0	0.0	0.9	0.0	0.5
	Surface sediment	0.0	2.5	0.0	0.7	0.0	1.0	0.5	1.5	0.4	1.0	1.7	2.5	2.2	1.0	0.6	0.0	0.0	0.5	1.0
Deepest	1Apr-26May09	0.0	0.5	0.0	1.7	0.0	0.6	0.0	0.0	0.0	3.1	0.0	0.0	2.4	1.7	0.0	0.0	0.0	4.6	0.5
	26May-22Jul09	0.0	0.0	0.0	0.4	0.0	0.0	0.0	0.0	0.0	2.1	0.0	0.9	0.0	2.6	0.0	0.0	0.0	0.0	0.4
	22Jul-01Oct09	1.1	0.0	0.0	1.9	0.5	7.2	0.0	0.5	0.0	5.6	0.0	0.0	1.6	7.0	0.0	0.0	0.5	1.6	0.0
	01Oct-20Nov09	1.0	0.0	0.0	0.0	1.0	1.1	2.0	2.0	0.0	1.5	0.0	1.5	3.6	3.6	0.0	0.5	1.5	1.0	2.0
	20Nov-20Jan10	0.0	0.0	1.0	0.2	0.5	4.6	0.5	0.7	0.7	3.0	1.4	0.5	4.6	0.6	2.3	0.0	0.5	0.0	0.5
	20Jan-19Mar10	0.0	1.4	0.6	0.8	0.4	0.9	0.4	0.0	1.9	1.0	1.2	0.6	2.0	1.2	1.4	0.0	1.0	0.8	1.2
	19Mar-02Jun10	0.0	0.0	0.0	0.6	0.0	0.4	0.0	0.0	0.6	0.0	2.9	1.7	0.0	0.6	1.0	0.0	0.0	0.8	1.7
	02Jun-22Jul10	0.0	0.0	0.0	0.0	0.5	0.0	0.0	0.0	0.0	0.0	2.4	0.5	1.0	1.0	0.5	0.0	0.0	0.0	0.0
	Surface sediment	0.0	1.0	0.8	0.6	0.4	2.8	0.4	0.4	1.2	0.8	4.5	2.6	1.6	0.0	1.0	0.0	0.0	0.4	1.2
Outflow	1Apr-26May09	0.4	0.4	0.0	1.3	0.0	1.1	0.7	0.7	0.0	6.1	0.9	0.4	0.9	2.9	0.4	0.4	0.0	0.9	0.4
	26May-22Jul09	0.0	0.0	0.0	0.4	0.0	0.0	0.0	1.6	0.0	2.7	0.0	0.4	0.4	0.4	0.7	0.0	0.0	0.0	0.0
	22Jul-01Oct09	0.7	0.7	0.0	0.7	0.0	2.6	1.5	1.5	1.5	5.2	0.0	0.0	2.2	7.8	0.5	0.4	2.2	0.0	3.7
	01Oct-20Nov09	0.0	0.0	0.0	0.5	0.0	2.1	1.5	1.5	0.0	3.5	0.0	0.0	1.8	3.0	1.4	0.0	0.0	1.0	0.0
	20Nov-20Jan10	0.0	0.0	0.0	0.4	0.0	1.3	0.8	1.3	2.0	0.5	1.6	1.3	3.8	0.0	0.7	0.0	0.3	0.5	1.0
	20Jan-19Mar10	1.2	0.0	0.3	0.3	0.5	1.3	0.0	0.0	3.2	0.0	1.5	0.5	5.0	0.3	2.4	0.0	0.8	0.5	2.4
	19Mar-02Jun10	0.0	0.0	0.0	1.2	0.3	0.0	0.2	0.2	0.5	0.0	1.9	2.4	3.5	0.0	0.1	0.0	0.0	1.4	0.3
	02Jun-22Jul10	0.0	0.0	0.0	0.5	0.0	0.0	0.0	0.0	0.0	0.0	1.5	0.0	0.0	1.0	0.0	0.0	1.0	0.0	0.5
	Surface sediment	0.0	1.2	0.5	0.2	0.2	0.0	0.0	0.9	0.5	0.7	0.9	4.0	1.9	0.7	2.3	0.7	0.0	0.9	1.9

Appendix L continues - Relative abundance (%) of planktonic and benthic diatom taxa, relative abundance of fossil diatoms (≥ 1% abundance), diatom accumulation (valves 10^3 d^{-1} cm^{-2}) in trap samples (grey columns) and diatom concentration (valves 10^3 g^{-1}) of surface sediment in sediment traps sediment trap samples (1st April 2009 – 22nd July 2010; n=8) and surface sediment (0-1 cm; n=3) from the inflow, deepest and outflow stations in Feeagh.

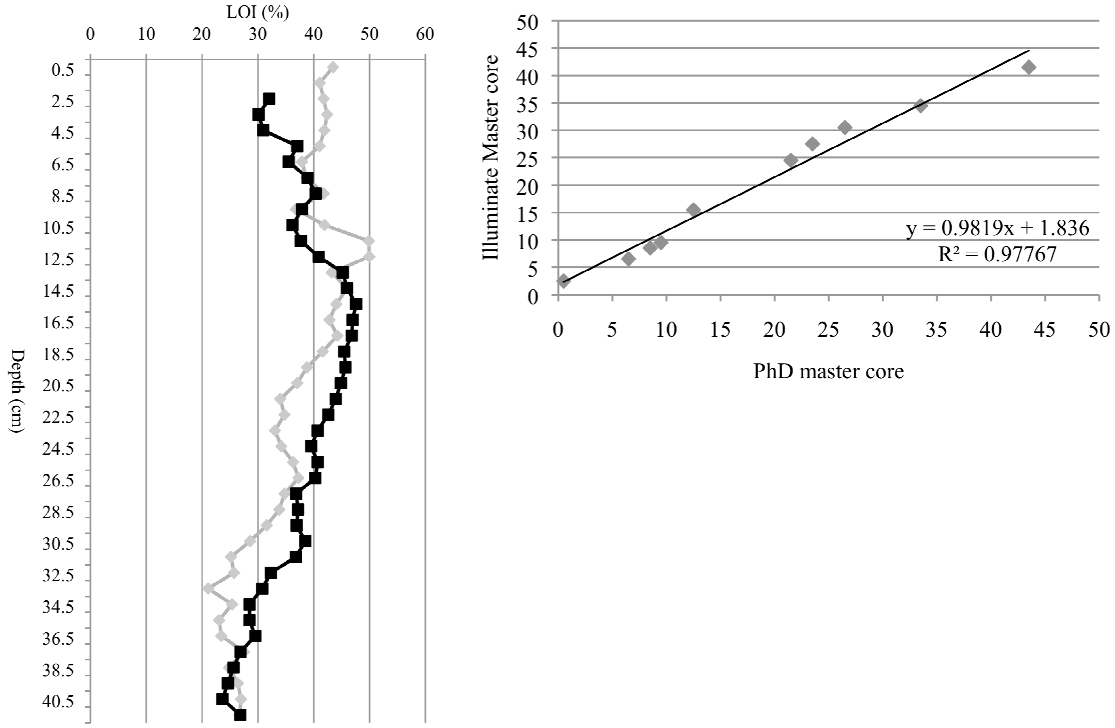
		Gomphonema parvulum	*Gomphonema pumilum*	*Gompho* sp.	*Navicula porifera* var. *opportuna*	*Navicula pseudocutiformis*	*Pinnularia appendiculata*	*Reimera sinuata*	*Synedra ulna*	*Tabellaria flocculosa*	unkown	Diatom accumulation/ concentration
Inflow	1Apr-26May09	0.5	0.0	0.0	0.0	0.0	0.5	0.5	1.9	5.5	0.0	25.8
	26May-22Jul09	0.3	0.0	0.3	0.0	0.3	0.3	0.0	0.4	13.1	0.6	0.5
	27Aug-01Oct09	1.8	0.0	0.6	0.9	1.5	1.2	1.2	0.0	8.2	0.6	10.4
	01Oct-20Nov09	3.3	0.0	2.3	0.0	1.2	1.4	0.0	1.2	8.7	0.0	6.7
	20Nov-20Jan10	3.5	0.0	0.8	1.1	0.0	1.1	0.0	0.5	6.5	1.9	6.4
	20Jan-19Mar10	1.8	0.0	1.3	0.0	0.0	0.8	0.3	0.1	3.0	0.5	0.7
	19Mar-02Jun10	1.7	1.3	1.7	0.0	0.4	0.4	0.0	0.4	4.1	0.6	0.6
	02Jun-22Jul10	0.5	0.0	0.5	0.2	0.0	0.0	0.0	1.3	3.4	0.5	4.5
	Surface sediment	2.0	0.0	0.0	0.0	0.0	0.5	0.5	0.1	4.2	0.5	8.4
Deepest	1Apr-26May09	0.5	0.0	3.9	0.0	0.7	0.7	0.0	1.9	3.4	0.0	21.0
	26May-22Jul09	0.0	0.0	0.0	0.0	0.2	0.4	0.0	0.0	4.3	0.0	1.1
	22Jul-01Oct09	0.5	0.0	3.2	1.1	1.3	0.0	0.0	0.3	8.0	0.0	7.2
	01Oct-20Nov09	3.0	0.0	2.5	0.5	0.0	0.3	0.5	0.5	7.6	0.0	12.8
	20Nov-20Jan10	3.7	0.0	2.3	0.5	2.3	1.3	1.2	0.7	3.5	1.8	0.6
	20Jan-19Mar10	1.8	0.4	2.5	0.0	0.0	2.4	0.4	0.3	2.5	0.0	0.9
	19Mar-02Jun10	1.7	0.4	0.0	0.4	0.6	0.4	0.0	0.8	2.7	0.2	2.5
	02Jun-22Jul10	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2	1.2	0.2	11.0
	Surface sediment	1.2	1.2	0.4	0.4	0.0	2.2	0.4	0.6	5.5	0.4	4.9
Outflow	1Apr-26May09	0.4	0.0	1.3	0.4	1.8	0.0	0.0	0.7	2.6	1.3	24.2
	26May-22Jul09	0.4	0.0	0.0	0.0	0.0	0.9	0.0	0.4	5.4	0.0	0.6
	22Jul-01Oct09	0.0	0.0	1.5	0.0	0.7	0.0	0.0	0.0	11.6	0.0	10.4
	01Oct-20Nov09	1.3	0.0	3.5	0.3	0.3	0.0	0.8	0.8	5.5	0.0	6.8
	20Nov-20Jan10	2.8	0.0	1.0	0.0	2.5	0.9	0.5	0.6	3.6	2.5	5.4
	20Jan-19Mar10	2.4	0.5	0.5	0.0	0.0	0.0	0.0	0.7	2.4	1.6	0.6
	19Mar-02Jun10	1.9	0.7	0.3	0.0	0.2	0.0	0.3	0.4	1.9	0.2	1.6
	02Jun-22Jul10	1.5	0.0	0.5	0.0	0.7	0.0	0.0	0.1	0.7	0.5	8.3
	Surface sediment	0.5	2.3	0.5	0.5	0.0	0.0	2.3	0.0	1.1	4.0	3.2

Appendix M - Relative abundance (%) of planktonic and benthic diatom taxa and relative abundance of fossil diatoms (≥ 1% abundance), diatom accumulation (valves 10^3 d^{-1} cm^{-2}) in trap samples (grey columns) and diatom concentration (valves 10^3 g^{-1}) in sediment traps sediment trap samples at inflow, deepest and outflow stations (19th May 2009 – 14th July 2010; n=2) and surface sediment (0-1 cm; n=1) from the deepest point in Guitane (n.d. = no data available).

		Planktonic	**Benthic**	*Achnanthes subatomoides*	*Achnanthidium minutissimum*	*Asterionella formosa*	*Aulacoseira subarctica*	*Brachysira garrensis*	*Brachysira vitrea*	*Cyclotella comensis*	*Cyclotella kuetzingiana*	*Cyclotella meneghenia*	*Cyclotella radiosa*	*Cymbella silesiaca*	*Eunotia implicata*	*Eunotia rhomboidea*	*Fragilaria capucina* var. *rumpens*	*Nitzschia palea*	*Tabellaria flocculosa*	Diatom accumulation/ concentration
	19May09-25Jan10	75.7	14.4	0.0	7.9	0.5	1.2	2.2	1.0	12.0	44.7	0.0	6.3	1.4	0.7	0.2	0.5	0.5	11.1	1.2
Inflow	25Jan10-14July10	60.7	26.5	0.6	13.0	1.5	5.0	2.5	2.5	8.8	27.3	1.1	2.2	2.5	1.3	2.3	1.5	0.2	14.7	1.3
	19May09-25Jan10	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d
Deepest	25Jan10-14July10	72.7	20.0	0.7	10.6	0.2	7.3	1.8	1.6	12.2	32.6	2.5	0.7	0.5	0.9	1.1	1.1	1.6	17.2	1.7
	Surface sediment	71.1	28.2	0.0	13.3	3.9	1.8	2.4	0.0	7.0	36.3	3.3	6.1	2.6	0.0	0.0	2.0	0.4	11.2	2.4
	19May09-25Jan10	68.0	23.3	1.1	11.6	3.0	1.9	1.7	2.4	11.4	35.7	0.0	6.2	3.0	1.1	1.1	0.9	0.4	9.7	1.7
Outflow	25Jan10-14July10	58.4	30.9	1.2	17.6	1.7	1.4	3.6	1.7	10.9	29.2	1.0	0.5	3.0	0.7	0.7	1.2	1.3	13.8	1.2

Appendix N a) Feeagh LOI_{550} versus depth (grey line corresponds to Illuminate master core with ^{210}Pb chronology, while black line corresponds to PhD master core) b) Correlation scatter plot between Illuminate Master core and PhD master core (n=10, R^2=0.97767).

a) b)



Appendix O - ^{210}Pb, Cs^{137} and Am^{241} concentrations in core Guitane in the sediment core collected from Guitane.

Depth	Dry Mass	Total		Pb^{210} Supported		Unsupported		Cum supported Pb^{210}		Cs^{137}		Am^{241}	
cm	g cm^{-2}	Bq kg^{-1}	±	Bq k g^{-1}	±	Bq kg^{-1}	±	Bq m^{-2}	±	Bq kg^{-1}	±	Bq kg^{-1}	±
0.5	0.02	1094.85	64.95	106.04	13.77	988.81	66.39	184.10	12.70	113.23	9.76	0.00	0.00
1.5	0.09	881.47	35.00	58.18	6.90	823.29	35.67	821.10	46.60	197.13	6.70	0.00	0.00
2.5	0.21	947.50	24.79	62.63	3.82	884.87	25.08	1815.70	76.30	262.00	5.24	0.00	0.00
3.5	0.34	825.91	33.38	82.52	6.27	743.39	33.96	2879.50	99.50	329.56	7.94	0.00	0.00
4.5	0.47	672.06	19.88	64.31	3.32	607.75	20.16	3771.60	116.60	260.64	4.61	1.82	1.36
5.5	0.59	408.83	20.17	54.12	3.89	354.71	20.54	4359.00	123.10	142.78	4.23	1.65	1.50
6.5	0.74	284.55	16.48	54.42	3.42	230.13	16.83	4773.70	127.90	69.52	2.89	2.39	1.26
7.5	0.92	147.94	14.85	56.49	3.47	91.45	15.25	5052.40	131.90	30.99	2.27	0.00	0.00
8.5	1.13	103.69	12.89	56.27	3.16	47.42	13.27	5187.90	135.30	24.00	1.92	0.00	0.00
9.5	1.33	82.24	15.10	53.33	3.74	28.91	15.56	5264.70	138.30	19.12	2.15	0.00	0.00
10.5	1.53	75.18	8.80	55.97	2.12	19.21	9.05	5311.80	141.20	17.62	1.25	0.00	0.00
11.5	1.71	57.40	9.12	62.75	2.44	-5.35	9.44	5324.50	142.20	9.67	1.22	0.00	0.00
12.5	1.89	72.59	15.02	49.78	3.85	22.81	15.51	5340.10	143.70	7.67	2.01	0.00	0.00
14.5	2.28	56.90	7.29	56.90	2.10	0.00	7.59	5384.00	150.20	5.75	0.93	0.00	0.00
16.5	2.72	56.28	6.54	53.28	1.68	3.00	6.75	5390.60	153.50	3.13	0.74	0.00	0.00
18.5	3.17	63.22	7.83	61.96	2.06	1.26	8.10	5399.80	156.80	2.39	0.82	0.00	0.00
20.5	3.57	58.18	8.23	64.86	2.20	-6.68	8.52	5389.10	160.40	1.83	0.97	0.00	0.00
22.5	3.93	66.53	7.71	68.65	2.26	-2.12	8.03	5373.00	163.40	2.85	0.87	0.00	0.00
24.5	4.35	65.08	10.62	64.77	2.74	0.31	10.97	5369.20	167.40	0.00	0.00	0.00	0.00
26.5	4.72	64.40	8.40	61.07	2.11	3.33	8.66	5373.80	172.10	2.33	0.91	0.00	0.00

Appendix P - LOI_{550} (%), TN (%), TOC (%) and C/N ratio for Feeagh (inflow, deepest and outflow) and Guitane (deepest sediment core).

	Feeagh												Guitane			
	Inflow				Deepest				Outflow				Deepest			
Depth	**LOI**	**TN**	**TOC**	**C/N**	**LOI**	**TN**	**TOC**	**C/N**	**LOI**	**TN**	**TOC**	**C/N**	**LOI**	**TN**	**TOC**	**C/N**
0-1	32.0	0.7	14.3	20.4	32.0	0.8	13.4	16.8	45.5	0.9	19.1	21.2	21.6	0.5	7	13.4
1-2	24.6				30.1				31.7				20.2			
2-3					30.9								18.6			
3-4	28.9	0.9	17.9	19.6	37.0	1	18.5	19.2	36.3	1	19.6	19.1	18.0	0.6	7.8	13.9
4-5					35.5								15.9			
5-6	26.0	0.7	14.3	21.7	38.9				41.1				13.6	0.6	6.1	13.4
6-7					40.4								12.4			
7-8	31.3				37.9	1.1	21.9	19.9	39.1	0.9	19.2	21.5	11.4	0.4	5.1	13.9
8-9					36.2								12.8			
9-10	35.3				37.7	1.1	21.3	20.3	39.8				17.3	0.5	8.7	16.2
10-11					40.9								18.0			
11-12	31.8	0.8	17.2	21.8	45.2	1.2	24.1	20.2	43.7	1.2	25	20.6	18.6			
12-13					45.9								18.8			
13-14	28.5				47.6	1.3	27.1	20.4	38.3	1	20.1	20.9	18.4			
14-15					47.0								16.0			
15-16	27.0	0.8	16.1	21.5	46.8	1.2	24.4	20.3	34.6				12.9			
16-17					45.5								10.8	0.6	4.9	7.7
17-18	23.0				45.7				33.6				11.4			
18-19					44.9								13.1			
19-20	23.8				43.9	1.3	25.8	19.7	27.7	0.8	17	20.7	13.8			
20-21					42.6								12.8			
21-22	21.0	0.6	13.1	21.4	40.7				26.2				14.7			
22-23					39.5								14.3			
23-24	17.0				40.7				25.5	0.7	13.6	19.7	15.2	1.0	8.5	8.6
24-25					40.3								15.1			
25-26	16.2	0.4	8.4	21	36.8				24.1				15.4			
26-27					37.2								14.1			
27-28	16.9				36.9	1.1	20	19.2	22.4				12.8			
28-29					38.5								13.0			
29-30	17.1	0.4	8.8	20.7	36.8				21.0	0.6	10.4	18.5	15.3			
30-31					32.3								12.9			
31-32	15.0				30.8				20.6				13.8			
32-33					28.5								16.7			
33-34	17.5				28.5				20.9	0.5	9.6	19.4	17.3	1.2	10.1	8.2
34-35					29.5								16.6			
35-36	16.6	0.4	7.7	21.1	26.9				20.0				18.5			
36-37					25.6								16.8			
37-38	19.9				24.6				19.8				16.1			
38-39					23.6								15.8			
39-40	18.1	0.4	8	23.1	26.8	0.8	15.5	20.1		0.6	11.6	20.8	16.0	1.4	10.7	7.9
40-41													15.7			
41-42													15.9			
42-43													14.0			
43-44													14.9			
44-45													15.2			
45-46													15.5			
46-47													13.3			
47-48													14.5			
48-49													12.9			
49-50													16.3			
50-51													15.2			
51-52													18.1	1.0	11.3	11
52-53													17.9			

Appendix Q - Fossil pigment concentrations (nmol g^{-1} DW) in Feeagh inflow sediment core.

Depth (cm)	All plantae and algae					Chlorophyta, Euglenophyta, plantae		Chlorophyta/ Cyanophyta	Cyanobacteria			Siliceous algae		Crytpo-phyta	
	Chl-*a*	Chl-*a*'	Pheophytin *a*	b-caro-tene	Pheophor-bide *a*'	Chl-*b*	Pheo-phytin *b*	Lutein/Zea-xanthin	Cantha-xanthin	Echine-none	Myxo-xantho-phyll *a*	Fuco-xanthin	Diato-xanthin	Allo-xanthin	UV-abs comp
0-1	3.91	0.89	11.66	1.11	10.64	1.21	33.99	10.68	0.16	0.39	0.53	1.52	1.74	3.73	0.65
1-2	9.58	16.41	18.46	1.22	6.40	1.57	54.00	11.73	0.62	0.00	0.00	4.02	4.08	1.41	2.81
3-4	2.93	6.42	9.49	1.17	7.40	0.90	28.15	13.14	0.57	0.00	0.00	2.78	3.37	1.64	4.29
5-6	1.36	5.90	5.53	0.70	7.50	0.39	19.93	13.91	0.50	0.00	0.00	1.87	2.63	1.66	4.40
7-8	1.28	15.06	6.07	0.82	36.82	0.56	24.99	16.03	0.44	0.00	0.00	1.23	2.51	1.45	3.89
9-10	1.26	6.60	6.84	1.07	6.00	0.35	30.20	13.68	0.99	0.00	0.00	0.79	1.51	0.99	4.07
11-12	1.12	7.80	6.14	1.03	5.10	0.65	26.11	17.15	0.46	0.00	0.00	0.60	2.14	1.46	4.47
13-14	1.68	7.70	8.99	1.41	8.40	0.91	37.10	19.43	0.61	0.00	0.00	0.46	2.57	1.43	6.66
15-16	1.89	7.72	10.72	1.51	7.00	0.70	37.68	19.20	0.71	0.00	0.00	0.88	3.43	1.41	5.67
17-18	1.38	6.55	8.74	0.91	6.30	0.44	29.63	20.97	0.86	0.00	0.00	0.33	3.19	1.09	4.55
19-20	1.30	11.30	7.25	0.86	7.00	0.40	22.81	19.04	0.95	0.00	0.00	0.65	3.36	1.79	3.94
21-22	1.74	11.60	9.53	0.92	9.40	0.47	28.62	19.71	0.85	0.00	0.00	0.30	2.91	1.01	5.31
23-24	1.72	7.90	7.96	0.57	4.00	0.53	25.16	15.25	0.70	0.00	0.00	0.54	0.97	0.36	3.50
25-26	1.97	10.10	8.21	0.48	5.00	0.92	24.22	12.45	0.53	0.00	0.00	0.41	0.72	0.00	2.57
27-28	2.20	9.30	12.27	1.00	5.00	0.82	36.83	12.79	0.64	0.00	0.00	0.25	0.83	0.00	3.25
29-30	2.03	9.50	12.62	1.13	6.00	0.56	36.88	14.02	0.91	0.00	0.00	0.24	2.06	0.00	3.92
31-32	2.13	5.90	11.41	0.99	5.00	0.88	30.80	14.54	0.77	0.00	0.00	0.26	0.97	0.00	3.15
33-34	2.02	9.30	9.94	0.76	4.00	0.75	25.62	12.69	0.67	0.00	0.00	0.00	1.05	0.00	2.64
35-36	1.26	5.30	6.94	0.43	5.00	0.57	17.07	8.49	0.57	0.00	0.00	0.00	0.56	0.00	2.47
37-38	1.45	8.07	9.54	0.86	5.00	0.59	20.91	7.77	0.61	0.00	0.00	0.00	0.72	0.00	4.05
39-40	1.94	7.60	9.66	0.94	5.00	0.44	21.25	7.82	0.74	0.00	0.00	0.00	1.40	0.00	1.92

Appendix Q continues - Fossil pigment concentrations (nmol g^{-1} DW) in Feeagh deepest sediment core.

Depth (cm)	All plantae and algae					Chlorophyta-, Euglenophyta, plantae		Chlorophyta/ Cyanophyta	Cyanobacteria			Siliceous algae		Crytpo-phyta	
	Chl-*a*	Chl-*a*'	Pheo-phytin *a*	b-caro-tene	Pheophor-bide *a*'	Chl-*b*	Pheo-phytin *b*	Lutein/Zea-xanthin	Cantha-xanthin	Echine-none	Myxo-xantho-phyll *a*	Fuco-xanthin	Diato-xanthin	Allo-xanthin	UV-abs comp
0-1	10.04	1.94	19.29	1.74	30.92	2.55	40.22	19.66	0.97	0.83	0.00	10.11	2.84	9.00	0.00
1-2	5.65	1.14	10.16	0.36	1.14	0.89	23.41	16.62	0.56	0.00	0.00	7.27	6.46	3.81	1.55
3-4	4.23	0.86	8.83	1.83	1.00	0.77	20.82	13.30	0.44	0.00	0.00	2.83	5.08	3.28	0.73
5-6	4.61	1.08	5.97	1.66	1.52	2.23	19.73	16.33	0.78	0.00	0.00	2.58	7.77	4.67	1.44
7-8	2.48	0.43	6.03	1.44	0.92	0.74	20.23	18.36	0.60	0.00	0.00	1.54	4.93	4.04	1.95
9-10	7.74	1.92	20.99	3.95	2.19	4.34	74.08	36.02	1.46	0.00	0.00	2.83	5.33	7.30	5.04
11-12	1.11	0.21	3.30	0.65	0.89	0.31	11.15	13.63	0.39	0.00	0.00	0.90	2.51	1.92	1.92
13-14	1.24	0.26	4.36	0.86	0.93	0.40	16.34	17.51	0.36	0.00	0.00	0.98	3.43	2.42	2.35
15-16	2.56	0.52	10.72	1.85	1.63	0.89	40.10	31.06	0.70	0.00	0.00	1.65	4.90	3.65	4.88
17-18	2.12	0.30	6.96	1.92	1.16	0.80	28.09	24.75	0.51	0.00	0.00	1.67	4.86	3.02	3.48
19-20	0.82	0.14	3.25	0.56	0.66	0.31	11.04	13.12	0.32	0.00	0.00	0.58	2.31	1.75	1.10
21-22	1.98	0.29	5.23	1.29	0.93	0.71	21.02	19.45	0.61	0.00	0.00	1.29	4.91	2.59	2.78
23-24	2.36	0.29	6.67	0.19	0.86	0.68	17.58	19.14	0.97	0.00	0.00	1.11	6.46	2.85	1.03
25-26	2.63	0.58	11.97	0.67	1.07	0.89	10.54	23.71	1.05	0.00	0.00	0.87	4.79	2.85	2.07
27-28	1.98	0.57	7.89	1.56	2.37	1.99	13.98	27.64	1.33	0.00	0.00	1.20	4.77	2.16	1.57
29-30	1.81	0.33	6.02	1.37	2.08	2.57	15.48	26.59	1.25	0.00	0.00	1.17	4.59	2.35	1.79
31-32	2.09	0.42	6.09	1.43	1.08	1.63	18.72	21.62	1.80	0.00	0.00	0.75	4.31	1.33	2.01
33-34	1.86	0.35	6.79	1.05	0.94	0.56	17.45	21.64	0.87	0.00	0.00	0.85	4.15	2.38	1.53
35-36	2.69	0.55	8.90	1.60	1.32	0.87	26.53	22.17	1.02	0.00	0.00	0.98	4.10	2.66	2.60
37-38	2.49	0.52	9.50	1.48	1.21	0.82	23.32	21.33	0.83	0.00	0.00	0.75	3.55	2.25	1.92
39-40	1.84	0.36	7.23	1.00	1.84	0.63	17.61	19.87	0.74	0.00	0.00	0.64	3.42	1.99	2.50

Appendix Q continues - Fossil pigment concentrations (nmol g^{-1} DW) in Feeagh outflow sediment core.

Depth (cm)	All plantae and algae					Chlorophyta, Euglenophyta, plantae		Chlorophyta/ Cyanophyta	Cyanobacteria			Siliceous algae		Crytpo-phyta	
	Chl-*a*	Chl-*a*'	Pheo-phytin *a*	b-caro-tene	Pheo-phorbide *a*'	Chl-*b*	Pheo-phytin *b*	Lutein/Zea-xanthin	Cantha-xanthin	Echine-none	Myxo-xantho-phyll *a*	Fuco-xanthin	Diato-xanthin	Allo-xanthin	UV-abs comp
0-1	4.00	0.75	9.37	0.59	16.12	1.59	29.37	9.42	0.46	0.37	0.51	2.66	1.51	4.24	0.80
1-2	3.28	0.53	5.30	0.70	5.43	0.63	19.68	8.69	2.66	0.00	0.00	3.31	3.05	2.94	2.20
3-4	2.07	0.33	4.86	0.64	4.04	0.95	18.43	9.12	0.45	0.00	0.00	2.06	2.84	2.73	2.42
5-6	1.01	0.22	3.71	0.46	1.93	0.58	12.60	7.98	0.31	0.00	0.00	0.98	1.71	1.73	2.15
7-8	2.24	0.62	8.75	1.26	4.50	1.08	24.07	15.22	1.08	0.00	0.00	0.85	3.62	2.18	0.97
9-10	1.79	0.62	7.59	1.19	3.50	0.95	20.58	12.76	0.85	0.00	0.00	0.51	2.28	1.63	1.12
11-12	0.91	0.91	4.53	0.84	0.91	0.86	15.06	14.17	1.14	0.00	0.00	0.53	2.82	1.76	1.57
13-14	4.35	0.74	19.87	0.33	0.93	1.89	62.45	20.76	0.66	0.00	0.00	0.78	3.93	2.08	9.09
15-16	1.43	1.43	7.65	0.27	1.34	0.93	23.51	15.99	1.30	0.00	0.00	0.55	3.20	1.78	1.70
17-18	1.92	1.92	6.84	0.34	1.17	1.01	6.55	11.47	0.57	0.00	0.00	1.29	1.92	1.83	3.80
19-20	2.08	2.08	3.72	0.55	0.93	1.10	3.72	9.02	0.49	0.00	0.00	1.02	1.60	1.61	1.94
21-22	2.23	2.23	9.65	0.55	2.86	1.40	26.58	18.90	0.86	0.00	0.00	0.67	3.12	1.89	2.24
23-24	1.25	1.25	18.33	0.64	1.17	1.11	55.64	19.10	1.14	0.00	0.00	0.87	3.46	2.18	6.92
25-26	1.09	0.88	11.21	0.00	1.39	1.27	26.51	21.56	1.35	0.00	0.00	0.65	3.54	2.57	1.48
27-28	0.88	0.63	13.03	0.00	1.74	1.04	28.89	13.95	1.25	0.00	0.00	0.48	2.98	2.12	2.57
29-30	1.01	0.76	15.47	0.00	1.21	1.52	42.22	17.56	1.42	0.00	0.00	0.00	2.73	2.68	4.45
31-32	0.81	0.27	14.50	0.00	1.39	1.68	26.86	7.55	0.85	0.00	0.00	0.00	0.60	0.94	1.58
33-34	0.71	0.40	11.54	0.00	1.16	1.83	23.19	9.49	1.24	0.00	0.00	0.00	0.87	1.00	1.45
35-36	0.66	0.38	9.61	0.00	1.20	1.10	19.42	12.42	1.29	0.00	0.00	0.00	0.97	0.75	1.95
37-38	0.76	0.45	6.39	0.00	1.28	0.60	16.53	7.87	0.95	0.00	0.00	0.00	0.59	0.98	4.99
39-40	4.00	0.75	9.37	0.59	16.12	1.59	29.37	9.42	0.46	0.37	0.51	2.66	1.51	4.24	0.80

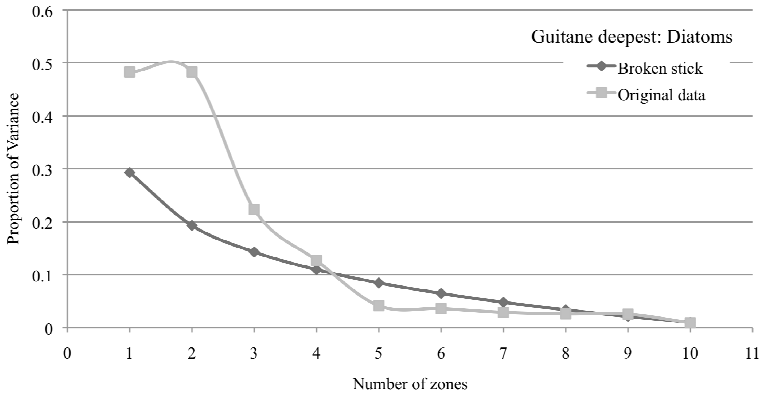
Appendix T – Relative abundance (%) of planktonic and benthic diatom taxa in surface sediment from the deepest waters and the sediment core collected for *Illuminate* Project (0-61.5 cm).

	Depth (cm)	Planktonic	Benthic
	0-1	37.5	61.8
	0-1	62.5	36.3
	2-3	55.1	47.1
	4-5	53.8	43.1
	6-7	62.2	37.8
	8-9	58.9	38
	10-11	55.6	42.5
	12-13	62.2	36
	14-15	56.5	42.9
	16-17	56	40
	19-20	51	48.7
	21-22	49.7	48.3
	23-24	51	49
	25-26	50.8	47.7
	29-30	54.8	44.3
	31-32	58.2	41.8
	33-34	57.4	40.3
	35-36	61.7	37.3
	37-38	60.4	38.4
	39-40	71.3	28.7
	41-42	64.4	35.6
	43-44	68.4	30.6
	45-46	63.6	34.3
	47-48	67.3	32.7
	49-50	62.7	34.3
	50-51	56.9	43.1
	53-54	62.1	37
	55-56	65.1	34.3
	57-58	68.8	30.4
	59-60	61.3	36.8
	61-62	59.7	38.4

Appendix U – Relative abundance (%) of planktonic and benthic diatom taxa and relative abundance of fossil diatoms (• 1 % abundance) in deepest sediment core in Guitane (n=10).

Depth (cm)	Planktonic	Benthic	*Achnanthes oblongella*	*Achnanthes petersenii*	*Achnanthidium minutissimum*	*Asterionella formosa*	*Aulacoseira subarctica*	*Brachysira garrensis*	*Brachysira vitrea*	*Cyclotella comensis*	*Cyclotella kuetzingiana*	*Cyclotella meneghenia na*	*Cyclotella radiosa*	*Cyclotella rossii*	*Cymbella silesiaca*	*Diatoma moniliformis*	*Eunotia incisa*	*Fragilaria brevistriata*	*Fragilaria capucina* var. *rumpens*	*Fragilaria exigua*	*Fragilaria gracilis*	*Fragilaria tenera*	*Gomphonema minutum*	*Navicula cryptocephala*	*Nitzschia linearis*	*Nitzschia palea*	*Tabellaria flocculosa*	Diatom concentration
0.5	71.1	28.2	0.2	0.9	13.1	3.9	1.8	2.4	0.0	7.0	36.3	3.3	6.1	0.0	2.6	0.0	0.9	0.4	2.0	0.2	0.0	0.7	0.4	0.0	0.0	0.4	11.2	2.4
3.5	79.2	19.2	0.0	0.0	8.6	10.5	2.6	0.9	0.7	12.6	41.1	3.0	2.6	0.0	0.0	0.7	0.0	0.0	0.7	0.0	0.0	1.4	0.0	0.5	0.9	1.2	4.2	6.7
5.5	69.5	28.3	0.5	0.0	13.9	1.3	0.5	0.8	0.3	25.7	19.0	1.9	6.1	5.3	0.0	0.0	0.0	0.5	0.0	1.1	0.3	0.8	0.5	1.3	1.3	1.1	8.6	5.1
7.5	55.4	43.3	1.0	0.0	17.1	0.0	1.2	2.5	2.5	24.0	19.3	1.7	2.5	1.7	1.2	0.5	0.0	1.5	0.0	3.7	1.5	0.2	0.2	1.2	1.2	2.0	3.0	14.2
9.5	55.9	43.3	0.0	1.2	19.1	0.4	0.0	3.5	0.8	26.6	19.9	0.0	3.0	0.8	2.6	0.4	0.0	0.8	0.0	1.4	2.2	0.6	0.0	0.4	1.0	0.2	2.8	9.0
16.5	53.7	45.3	0.7	0.7	18.3	0.2	1.0	3.2	3.2	34.4	12.1	0.0	1.5	1.0	2.7	0.7	1.5	1.2	0.0	1.2	0.2	0.2	0.0	1.2	1.5	0.0	2.7	10.4
23.5	62.2	37.8	1.2	0.0	12.5	0.0	0.0	1.0	1.7	41.3	11.5	0.5	3.7	1.2	2.5	2.0	1.2	0.0	1.0	0.5	1.7	1.0	1.5	1.0	0.0	1.5	1.5	9.8
33.5	67.1	32.6	1.4	0.0	11.9	0.0	0.0	3.9	2.3	25.6	28.3	0.9	2.1	4.8	0.9	1.6	0.5	0.0	0.2	1.4	0.5	1.4	0.2	1.1	0.0	0.9	2.5	6.2
39.5	50.0	50.0	0.0	0.0	13.9	0.0	0.0	4.0	3.2	11.7	25.9	1.7	1.7	2.5	3.7	0.5	3.7	0.0	1.7	3.2	0.7	0.2	0.0	2.7	0.0	2.7	3.5	4.5
51.5	51.3	48.7	0.2	2.1	11.5	0.0	0.0	5.9	2.3	18.5	23.4	0.5	1.2	0.7	1.4	3.5	2.8	0.0	0.9	3.0	0.7	1.6	1.6	0.0	0.0	1.2	1.4	4.7

Appendix V – Broken stick model (dark grey) and original data (clear grey) for diatom assemblages from the deepest core in Guitane.



Appendix W - Taxon name and authorities for diatoms identified in Feeagh (traps and surface sediment) and Guitane (traps and sediment core).

Taxon name	Authority
Achnanthes amoena	Hustedt, 1952
Achnanthes daui	Foged, 1962
Achnanthes didyma	Hustedt, 1933
Achnanthes impexiformis	Lange-Bertalot in Lange-Bertalot & Krammer, 1989
Achnanthes flexella	(Kützing) Brun, 1880
Achnanthes helvetica	(Hustedt) Lange-Bertalot in Lange-Bertalot & Krammer, 1989
Achnanthes laterostrata	Hustedt, 1933
Achnanthes laevis	Oestrup, 1910
Achnanthes lanceolata	(Breb. ex. Küting.) Gruen in Cleve & Grunow, 1880
Achnanthes oblongella	Oestrup, 1902
Achnanthes petersenii	Hustedt, 1937
Achnanthes pusilla	(Grunow) De Toni, 1891
Achnanthes pseudoswazi	Carther, 1963
Achnanthes saccula	Carter in Carter & Bailey-Watts, 1981
Achnanthes subatomoides	(Hustedt) Lange-Bertalot & Archibald in Krammer & Lange-Bertalot, 1985
Achnanthes ventralis	(Krasske) Lange-Bertalot in Lange-Bertalot & Krammer, 1989
Achnanthidium minutissimum	(Kützing) Czarnecki (sensu latu)
Amphora libyca	Ehrenberg, 1840
Amphora veneta	Kuetzing, 1844
Asterionella formosa	Hassll, 1850
Aulacoseira alpigena	(Grunow) Krammer, 1990
Aulacoseira subarctica	(O. Mueller) Haworth, 1988
Amphora veneta	Kützing, 1844
Brachysira garrensis	Lange-Bertalot & Krammer, 1985
Brachysira neoexilis	Lange-Bertalot, 1994
Brachysira styriaca	(Grunow) Hudstedt, 1930
Brachysira vitrea	(Grunow) Ross, 1966
Caloneis molaris	(Grunow) Krammer, 1985
Cavinula cocconeiformis	(Gregory ex Greville) Mann & Stickle in Round, Crawford & Mann 1990
Cocconeis placentula	Ehrenberg, 1838
Cyclotella comensis	Grunow in Van Heurck, 1882
Cyclotella kuetzingiana	(Grunow) Hakansson, 1990
Cyclotella menegheniana	Kützing, 1844
Cyclotella ocellata	Patocsek, 1901
Cyclotella radiosa	(Grunow) Lemmermann, 1900
Cyclotella rossii	Hakonsson, 1990

Cyclotella stelligera	Cleve & Grunow in Cleve, 1881

Appendix Y continues - Taxon name and authorities for diatoms identified in Feeagh (traps and surface sediment) and Guitane (traps and sediment core)

Cymbella cistula	(Ehrenberg) Kirchner, 1878
Cymbella gracilis	(Ehrenberg 1843) Kuetzing, 1844
Cymbella helvetica	Kützing, 1844
Cymbella microcephala	Grunow in Van Heurck, 1880
Cymbella silesiaca	Bleisch in Rabenhorst, 1864
Diatoma moniliformis	Kuetzing, 1833
Diatoma tenue	Agardh, 1812
Diploneis oblongella	(Naegeli) Cleve-Euler, 1922
Diploneis parma	Cleve, 1891
Epithemia adnata	(Kützing) Rabenhorst, 1853
Eunotia bilunaris	(Ehrenberg) Mills, 1934
Eunotia exigua	(Brèbisson ex Kuetzing) Rabenhorst, 1864
Eunotia implicata	Noerpel, Lange-Bertalot & Alles 1991
Eunotia incisa	Gregory, 1854
Eunotia fallax	A. Cleve, 1895
Eunotia glacialis	Meister, 1912
Eunotia hexaglypha	Ehrenberg, 1954
Eunotia paludosa	Grunow, 1862
Eunotia pectinalis	Rabenhorst, 1864
Eunotia praerupta	Ehrenberg, 1843
Eunotia rhomboidea	Hustedt, 1850
Fragilaria arcus	(Ehrenberg) Cleve, 1898
Fragilaria brevistriata	Grunow in Van Heurck, 1885
Fragilaria capucina var. *rumpens*	(Kützing) Lange-Bertalot, 1991
Fragilaria capucina var. *vaucheriae*	(Kützing) Lange-Bertalot, 1980
Fragilaria construens	(Ehrenberg) Hustedt, 1957
Fragilaria exigua	Grunow in Cleve & Moeller, 1878
Fragilaria gracilis	(Oestrup) Hustedt, 1950
Fragilaris leptostauron var. *martyi*	(Héribaud) Lange-Bertalot, 1991
Fragilaria pinnata	Ehrenberg, 1843
Fragilaria tenera	(W. Smith) Lange-Bertalot, 1980
Fragilaria virescens	Ralfs, 1843
Frustulia rhomboides	(Ehrenberg) De Toni, 1891
Gomphonema acuminatum	Ehrenberg, 1832
Gomphonema angustum	Agardh, 1831
Gomphonema clavatum	Ehrenberg, 1832
Gomphonema gracile	Ehrenberg, 1838
Gomphonema hebridense	Gregory, 1854
Gomphonema minutum	Agardgh, 1831
Gomphonema olivaceoides	Hustedt,
Gomphonema truncatum	Ehrenberg, 1832
Gomphonema parvulum	(Kützing) Kützing, 1849
Gomphonema pumilum	(Grunow) Reichardt & Lange-Bertalot, 1991
Meridion circulare	(Greville) Agardgh, 1831
Navicula capitata	Ehrenberg, 1838
Navicula cari	Ehrenberg, 1836
Navicula cryptocephala	Kuetzing, 1844
Navicula cryptotenella	Lange-Bertalot, 1985
Navicula jarnefeltii	Hutstedt, 1942
Navicula leptostriata	Joergensen, 1948
Navicula placentula	(Ehrenberg) Kuetzing, 1844
Navicula pupula	Kuetzing, 1844
Navicula pusilla	W. Smith, 1853
Navicula radiosa	Kuetzing, 1844
Navicula rhynchocephala	(Grunow) Grunow in Cleve & Moeller, 1877
Navicula minima	Grunow in Van Heurck, 1880
Navicula reinhardtii	(Grunow) Grunow in Cleve & Moeller, 1877

Appendix Y continues - Taxon name and authorities for diatoms identified in Feeagh (traps and surface sediment) and Guitane (traps and sediment core)

Navicula porifera var. *opportuna*	(Hustedt) Lange-Bertalot 1985
Navicula subtilissima	Cleve, 1891
Navicula tripunctata	(O.F. Mueller) Bory, 1822
Navicula viridula	(Kuetzing) Ehrenberg, 1838
Neidium ampliatum	(Ehrenberg) Krammer, 1985
Neidium ladegensis	(Cleve) Foged, 1952
Nitzschia angustata	(Schmith) Grunow in Cleve & Grunow, 1880
Nitzschia gracilis	Hantzsch, 1860
Nitzschia linearis	(Agardh) W. Smith, 1853
Nitzschia palea	(Kützing) W. Smith, 1856
Pinnularia appendiculata	(Agardh) Cleve 1895
Pinnularia divergens var. *linearis*	Ehrenberg, 1841
Pinnularia gibba var. *linearis*	Ehrenberg, 1841
Pinnularia intermedia	(Lagerstedt) Cleve, 1895
Pinnularia microstauron	(Ehrenberg) Cleve, 1891
Pinnularia silvatica	Hantzsch in Rabenhorst, 1861
Pinnularia viridis	(Nitzsch) Ehrenberg, 1843
Reimeria sinuata	(Gregory) Kociolek & Störmer, 1987
Stauroneis anceps	Ehrenberg, 1843
Surirella brebissonii	Krammer & Lange-Bertalot, 1985
Surirella linearis	Smith, W. 1853
Surirella brebissoni	Krammer & Lange-Bertalot, 1987
Synedra ulna	(Nitzsch) Ehrenberg, 1836
Tabellaria flocculosa	(Roth) Kützing, 1844
Tetracyclus glans	(Ehrenberg) Mills, 1935