Biotic response to forest harvesting in acidic blanket peat fed streams: a case study from Ireland

Connie O'Driscoll[1], Elvira de Eyto[2], Michael Rodgers[1], Mark O'Connor[1], Zaki-ul-Zaman Asam[1] and Liwen Xiao[1*]

[1] Civil Engineering, National University of Ireland, Galway, Republic of Ireland

[2] Marine Institute, Newport, Co. Mayo, Republic of Ireland

* Corresponding author: Liwen Xiao; Email: liwen.xiao@nuigalway.ie; Tel: +353 91 494188

Abstract

Blanket peat catchments are important biodiversity refugia and are increasingly recognised for their role in regional carbon and water balances. A key pressure on these catchments is forest clearfelling which increases stream phosphorus potentially leading to eutrophication. However, these unique systems are underrepresented in the development of bioassessment monitoring programmes and so are at risk to impacts. In this study, a multiple before- after-control-impact (MBACI) study was designed in three neighbouring peatland catchments and provided a unique opportunity to assess the impact of forest clearfelling events on macroinvertebrate and phytobenthic assemblages. Statistical analysis revealed that substantial differences in the macroinvertebrate assemblages after clearfelling with higher abundances of chironomids. Macroinvertebrate derived indices EPT, diversity and species richness were significantly reduced. This was accompanied with a consistent shift in functional feeding group representation away shredders and collector-filterers to a dominance of collector-gatherers after clearfelling. In contrast, forest clearfelling did not significantly impact the diatom assemblages and diatom derived indices remained static for the duration of the study period.

Keywords: Blanket peat; diatoms; macroinvertebrates; forest clearfelling; nutrients; Water Framework Directive (WFD)

1. Introduction

Peatland afforestation was practiced extensively in the UK and Ireland, Fennoscandia, and North America, during the late 20th century (Paavilainen and Paivanen, 1995). Many of these blanket peat forests are now reaching harvestable age and concerns have been raised about the potential impact of forest clearfelling to the receiving aquatic systems. Peatland

streams are naturally oligotrophic and phosphorus (P) enrichment, even by 10 $\mu g\ L^{-1}$, is a concern (Mainstone and Parr, 2002). Eutrophication has been highlighted as a key water quality problem in the UK and Ireland (EPA, 2004) with nutrient availability regarded as the most important factor regulating benthic algal production in oligotrophic rivers (Bowman et al., 2007). Catchment soil type and its ability to retain P have been identified as key determinants of P loading to receiving waters (Cummins and Farrell, 2003). Blanket peat soils have poor P adsorption capacity (Tamm et al., 1974) and P can easily be transferred to receiving water by the high rainfall and runoff these catchments are subjected to (Muller, 2000; Cummins and Farrell, 2003). Studies in Ireland, Finland, and the UK have shown that peatland forest clearfelling could deteriorate receiving water quality with increased nutrient and suspended sediment (SS) export (Cummins and Farrell, 2003; Nieminen, 2003; Rodgers et al., 2010).

Often, in Ireland and the UK, the earlier afforested blanket peat catchments were established without any riparian buffer areas, with trees planted to the stream edge (Broadmeadow and Nisbet, 2004; Ryder et al., 2011). Clearfelling of these catchments to the stream edge reduces canopy cover over the streams and increases sun light availability, resulting in increasing amounts of autochthonous energy production (Chizinski et al., 2010) and daily maximum temperatures (0.05–1.1°C) (Rodgers et al., 2008). These physicochemical status changes due to clearfelling activities can modify stream habitat and change biota assemblages. Protection of the aquatic systems draining these blanket peat forests is urgent as the Water Framework Directive (WFD) stipulates EU Member States must maintain ''high and good ecological status'' where it exists and to restore at least ''good status'' for all water bodies by 2015 (EU, 2000). However, low-alkalinity (<6.8 mg L^{-1} $CaCO^{3}$) catchments have been underrepresented in the derivation of ecological quality ratios (Kelly-Quinn et al., 2004; Kelly et al., 2008; O'Driscoll et al., 2012).

Studies of peatland forest clearfelling impacts on hydrology, soil erosion and nutrient release have been extensively reported, however, few studies focused on the impacts of peatland clearfelling on water ecological status. Macroinvertebrates and diatoms have been used successfully for assessment of ecological quality and aquatic ecosystems worldwide (Kelly et al., 1998; Leira and Sabater, 2005; Clarke and Hering, 2006; Chen et al., 2008). They have been considered as useful indicators for assessing potential impacts of forest clearfelling due to their well-described responses to environmental conditions (i.e., light, temperature, nutrients, sediment) that may change during forest operations (Naymik et al.,

2005; Reid et al., 2010). Forest harvesting has been reported to increase total macroinvertebrate abundance and biomass (Noel et al., 1986; Fuchs et al., 2003), and increase proportions of shredder,collector-filterer, collector-gatherer and collector-scraper functional feeding groups (Wallace, 1990; Kedzierski and Smock, 2001; Liljaniemi et al., 2002). Coincidentally, forest harvesting has also been reported to cause reductions in total macroinvertebrate abundance and biomass (Ormerod et al., 1993; Growns and Davis, 1994; Hynes, 1994). In the USA, Naymik et al., (2005) found that the changes in diatom assemblages were positively correlated with percentage of upstream area harvested, total nitrogen (N) and P concentrations. However, in addition to macroinvertebrate and diatom assemblages responding to forest harvesting, they can also be affected by naturally occurring physico-chemical riverine gradients (Richter et al., 1996; Potapova and Charles, 2002; Smucker and Vis, 2011) resulting in variable responses to forest harvesting induced changes. While diatom biomonitoring is considered a valuable tool’ in environmental management, concerns have been raised regarding its use in acid sensitive streams (O ‘Driscoll et al., 2012; Schneider et al., 2013). Timber production is set to increase across Europe, supported by governments for its potential to mitigate climate change through carbon sequestration and fossil fuel substitution (Laudon et al., 2011; Forest Service, 2008); there is therefore an urgent demand for more evidence to underpin evidence based policy developments regarding the future of afforested blanket peat.

This study aimed to improve our understanding of how forest clearfelling impacts on diatom and macroinvertebrate assemblages in acid sensitive peatland fed streams. Headwater streams provide the link with the catchment experiencing the land use change (Richardson and Danehy, 2007; Louhi et al., 2010) and the larger rivers which contain sensitive Annex II species such as salmonids and freshwater pearl mussels. Based on the findings of earlier studies of forest harvesting impacts on macroinvertebrates and diatom assemblages this study hypothesised that: (H_1) macroinvertebrate assemblages would differ significantly after forest clearfelling with associated reductions in biotic indices; (H_2) there would be corresponding significant differences observed for diatom assemblages following clearfelling with associated reductions in biotic indices, and that (H_3) the acid-tolerant feeding groups expected for low pH streams would prevail because of their ability to exploit both autochthonous and allochthonous food (Layer et al., 2013). How far these responses propagate to the larger catchment scale is unknown (Cummins and Farrell, 2003), however Rodgers et al., found that due to the dilution capacity of the main river, the P concentrations

in the main river were low after harvesting, indicating that catchment-based selection of the felling coupe size could limit the P concentrations in the main rivers after harvesting. Thus this study also hypothesised (H_4) that the clearfelling disturbance to the forested blanket peat catchment would not be detected in a catchment drainage system below the immediate area of disturbance. From an international perspective this study provides a unique opportunity to validate widely accepted biomonitoring tools in an area that has minor representation in the development of these tools. While broad surveys with a large number of sites are important for providing overall context, prioritising, and assigning status class, more intensive studies within individual catchments will elucidate some of these uncertainties.

2. Materials and methods

2.1 Site description

This study was based in the Nephin Beg Range, a Natura 2000 site in the north west of Ireland (Fig. 1), supporting Annex II species (European Communities, 2003) such as *Salmo salar* L., *Lutra lutra* L. and *Margaritifera margaritifera* L. Seven sites were selected on independent streams and were analogous in terms of slope, type of substrata, and land management use. Four of these streams received runoff from forestry which was felled during the study period (hereafter referred to as impact streams), and three drained afforested catchments which were not felled (hereafter referred to as control streams). Upstream catchment areas range from 10 ha to 32 ha and typically flow over bedrock, but in some sections have a peat floor (Table 1). *Pinus contorta* (lodgepole pine) and *Picea sitchensis* (sitka spruce) were the dominant tree types in the study catchments. In addition, six sites were selected (three) above and (three) below the confluence of the GSS study stream and the larger Glennamong river (GL) was selected to study how far the effects of forest clearfelling impacts propagate to the larger catchment scale (Fig. 1). A multiple before - after-control-impact (MBACI) experimental design was implemented (Bennett and Adams, 2004). One control stream (GCS) dried out for a short period during summer 2009 but not during summer 2010, thereby indicating inter-annual irregularity; and one impact stream (SSS) which was intended to be used as an impact site but during harvesting operations the main extraction mat was routed across catchment boundaries and so the 'disturbed' water drained into a neighbouring catchment and stream. As a result, the GCS and SSS were not used in the BACI statistical analysis. Forest clearfelling occurred in blocks/coupes of 10-20 ha and the study catchments were chosen to represent the felling of these coupe sizes. The stands were of similar age, planted in the late 1980s, and of similar density, with trees planted on ribbons at

1.5 m intervals, giving an approximate soil area of 3 m^2/ tree. The surface waters flowed along the furrows, were collected by collector drains, and then joined the streams. Ground mineral phosphate (GMP) was spot applied manually by hand at afforestation at a rate of approximately 13.2 kg P ha^{-1}. Bole-only harvesting was conducted using a Valmet 941 Harvester, and brash material was collected together to form the mats, thus protecting the soil surface, and reducing erosion.

2.2 Samples collected

Site locations were visited on eleven occasions over a period of 3 years, initially in March 2009. On each occasion, the study streams were sampled on the same day to minimise inter-site differences due to temporal dynamics. At each site, macroinvertebrates, were collected using a 1 min kick sample with a standard kick-net (230 mm x 225 mm frame, with 0.9 mm mesh) (Toner et al., 2005) and were immediately preserved in 70% Industrial Methylated Spirits and transported back to the laboratory for sorting and identification. Where possible, macroinvertebrates were identified to species level under a light microscope (x40 magnification) for most taxa, Ephemeroptera, Plecoptera and Trichoptera (EPT), with Coleoptera and Diptera identified mainly to genus. All authors of macroinvertebrate taxonomic names can be found in the relevant appendices.

Additionally, at each site periphyton was removed from five cobble surfaces (Kelly et al., 1998) with 100 ml of stream water and stone surface area was calculated (Dall, 1979). Laboratory analysis was carried out later the same day for periphyton chlorophyll a (Chl *a* g/m^2) and ash free dry mass (AFDM g/m^2). Periphyton samples for diatom analysis were cleaned (Hendley, 1974) and permanent slides were prepared using Naphrax (refractive index = 1.74) as a mountant. At least 300 valves were identified (Krammer and Lange-Bertalot, 1986, 1997, 2000, 2004) and counted per slide using a 1000x microscope equipped with a 100 phase contrast objective (CEN, 2004). Certain approaches were adopted for the identification of difficult species: *Eunotia exigua*, present in high numbers in girdle view, was difficult to distinguish from *Eunotia tenella* and *Eunotia meisteri* and so the three were combined and considered as *Eunotia exigua* complex. *Gomphonema parvulum* has been described with a number of varieties and attributed environmental preferences the latest of which is found in Jüttner et al. (2013), however populations in these samples had high morphological variability, and so have been termed *Gomphonema parvulum* complex to reduce the systematic error that dividing these samples can introduce (Kahlert et al., 2012). All authors' of diatom taxonomic names can be found in Table 2.

Water temperature and electrical conductivity were recorded in the field using a WTW Multiline P4 field meter. Water samples were taken at each sampling site and analysed on the same day for alkalinity, colour and SS using standard procedures (APHA, 1998). Nutrient analysis included soluble reactive P (SRP), ammonium- N (NH^{+}.N) and nitrate-N (NO^{+}_{3}-N) using a Konelab 20 Analyser (Konelab Ltd., Finland). An H-flume, a water level recorder and a data logger were installed at the GSS. Water levels in the H-flume were recorded every 5 min. Water level data was collected with an Orpheus mini water level recorder OTT-hydrometry UK at one gauging station downstream in the GL and readings were taken every 5 minutes. Automated ISCO water samples were taken hourly during flood events in the GSS and on selected days in the GL, for SRP, and SS measurements.

2.3 *Data analysis*

Diatom and macroinvertebrate species richness (S) and Shannon- Weiner index (H') (Shannon and Weaver, 1963) were calculated. Biotic indices, EPT, Q index (McGarrigle et al., 2002) and SI (Medin's acid index; Henrikson and Medin, 1986) for macroinvertebrates, and TDI (Trophic Diatom Index, Kelly et al., 1998), EQR (Kelly et al., 2008) and ACID, (Andren and Jarlman, 2008) for diatoms were calculated at each site before and after clearfelling. Relative abundances were calculated for diatom and macroinvertebrate assemblages and functional feeding groups for the latter following designations by LeRoy Poff et al. (2006). Mann Whitney U tests were used to test for differences in indices and functional feeding groups before and after clearfelling.

Relative abundances were calculated for the macroinvertebrate and diatom assemblages and only species present in more than 5% of samples were included. Data were ordinated using non-metric multi-dimensional scaling (NMDS) of a matrix of Bray-Curtis dissimilarity values derived from the relative abundance data (inthe PRIMER software package, Plymouth Marine Laboratories, UK). Ordination was considered successful when global stresswas less than 20% and results were visualised using non-metricmultidimensional scaling (n-MDS) (Clarke, 1993). A two way comparison (before and after clearfelling, at control and impact sites) were made with PERMANOVA software (Anderson, 2005) in order to determine whether clearfelling had an impact on the assemblages. P-values were calculated by permuting the observations 9999 times, so no assumptions of the distributional form of the data were required. The calculated statistic (pseudo-F) was calculated, like a traditional F-statistic, as the sum of the squared distances within groups (Anderson, 2001; McArdle and

Anderson, 2001). The similarity percentages (SIMPER) procedure was used to identify the major species contributing to the similarity measure obtained (Clarke and Warwick, 1994).

3. Results

3.1 Impact of clearfelling on the macroinvertebrate assemblages

Macroinvertebrate assemblages in the control streams were similar before and after felling with *Baetis rhodani* and *Leuctra hippopus* dominating (Table 2). In the impact streams however the assemblages changed from being dominated by *Nemoura cinerea*, simulid species, *Leuctra hippopus* and chironomid species pre-felling (Table 2) to assemblages completely dominated by chironomid species with the abundance of 98% post-felling (Table 2). PERMANOVA indicated that the interaction treatment x time was a significant sources of variation in the macroinvertebrate assemblages ($p < 0.05$) (Table 3). Post pairwise comparisons indicated that this was due to significant differences in assemblages before and after clearfelling in the impact streams ($p < 0.01$). The two-dimensional MDS plot of the macroinvertebrate assemblages shows distinct separation between groupings of the impact streams pre- and post-felling with the pre-felling samples generally clumped together with the control samples and away from the post-felling samples (Fig. 2a). SIMPER analysis indicated that the samples taken at the impact streams pre-felling comprised a diverse mix of *Nemoura cinerea*, *Leuctra hippopus*, simulid species, chironomid species, dicranota species and *Polycentropus kingi*. In contrast, post-felling, these sites were dominated by chironomids (83%) along with some *Nemoura cinerea* and *Polycentropus kingi*. No significant change was observed in the macroinvertebrate assemblages in the GL post-felling. Baetis rhodani, *Leuctra hippopus* and simulid species were the dominant macroinvertebrate species identified at the GLa and GLb pre- and post-felling.

3.2 Impact of felling on the diatom assemblages

The diatom assemblages in the control streams, were dominated by *Achnanthes oblongella*, *Gomphonema parvulum* complex, and *Eunotia exigua* complex pre- and post-felling (Table 2). In contrast to the macroinvertebrate assemblages, there were no significant differences between diatom assemblages in the impact streams pre- and post-felling. These streams were characterised by *Eunotia exigua* complex and *Pinnularia appendiculata* both pre and post-felling (Table 2). Although there were two distinct groupings of study streams samples from pre- and post-clearfelling in the two-dimensional diatom MDS (Fig. 2b), samples from the control streams pre- and post-harvesting were intermingled with those groups (white circles Fig. 2b). The results from the diatom post hoc pairwise comparisons

showed no significant differences between assemblages pre- and post-felling in either control or impact streams (Table 3). Similarly, no significant diatom assemblage change was observed in the GL post-felling. Diatom assemblages were dominated by *Achnanthes oblongella*, *Eunotia exigua* complex, *Gomphonema parvulum* complex and *Tabellaria flocculosa* at the GLa and GLb pre- and post-felling.

3.3 Impact of clearfelling on the biotic indices and functional feeding groups

The SI and ACID values were low for the impact streams ranging from 0–4 to 0–1 respectively before felling indicating an acid stressed system (Fig. 3a and b). The TDI values calculated for all impacts sites were 0 pre-felling indicating an oligotrophic system (Fig. 3b). Post-felling values of EPT, species richness and species diversity were significantly reduced in the impact streams ($p < 0.05$) (Fig. 3a). No significant differences were observed in the phytobenthic indices pre- and post-felling ($p > 0.05$) (Fig. 3b). Chl a and AFDM values fluctuated considerably in both the control and impact sites before and after clearfelling, however these values were not significantly different from before to after felling ($p > 0.05$). The Q index EQR highlighted a drop in the WFD ecological quality of the impact streams from Good to Moderate status after the clearfelling. The TDI–EQR calculated for the diatom assemblages indicated a consistently high status across impact sites irrespective of felling. Before clearfelling the impact sites were dominated by shredders and collector–filterers and after clearfelling this shifted to an almost complete dominance of collector– gatherers (Fig. 4).

3.4 Impact of clearfelling on environmental variables

Pre-harvesting nutrient values ranged from 2.58–12.2 $\mu g\ L^{-1}$ of TRP and 30.8–120 $\mu g\ L^{-1}$ NH4–N. The pre-harvesting pH values of the streams ranged from 4.0–6.7. Post-harvesting values of TRP, temperature and SS were significantly larger ($p < 0.05$) than preharvesting, indicating that harvesting exerted a significant response on the impact sites (Fig. 5). In the control streams, TRP, water temperature and SS were similar before and after harvesting ($p > 0.1$) (Fig. 4). Similarly, in the GLa and GLb, TRP, temperature and SS were similar before and after harvesting even when TRP concentrations rose from $12 \pm 3\ \mu g\ L^{-1}$ in the GSS pre-harvesting to a maximum of 101.7 $\mu g\ L^{-1}$ at the end of the harvesting period (Fig. 6). The increased P concentration at the GSS did not appear to be detectable in the GL (Fig. 7) which covers a catchment area of 1135 ha above its confluence with the GSS.

4. Discussion

Prior to clearfelling the streams examined were oligotrophic ($P < 10\ \mu g\ L^{-1}$) with regard to nutrient status. Both water chemistry and biotic assemblage results prior to clearfelling indicated the streams examined in this study were typical of acidic forest peat fed streams (Riipinen and Dobson, 2010; O'Driscoll et al., 2012). The issue of whether these streams are naturally acidic or suffering from exacerbated levels of acidity due to enhanced scavenging of acidic sulphur and nitrogen compounds from the coniferous forestry plantations (Allott et al., 1997; Cruikshanks et al., 2008; Kelly-Quinn et al., 1996, 2008) cannot be answered by this study as there is limited data available from before afforestation.

Removal of the shading effect of the trees as a result of clearfelling naturally implied a change in lighting conditions in the open stretches of the impact streams. The receiving stream water was found to be warmer post-felling. Forest clearfelling also caused significant increases in TRP and SS consistent with previous studies carried out in peatland forest catchments (Nieminen, 2003; Cummins and Farrell, 2003; Rodgers et al., 2011). Increases in TRP and SS in addition to the rise in temperature brought about by the clearfelling had a considerable impact on the macroinvertebrate assemblages in the streams thus supporting H_1. Alterations to sunlight availability and temperature are commonly associated with changes in aquatic invertebrate communities (Johnson and Jones, 2000; Chizinski et al., 2010). Stonefly families Leuctridae and Nemouridae were found abundant in the streams before felling and dramatically reduced after clearfelling as pollution load increased in the impacted streams. Assemblages were dominated by chironomids after felling, a pattern characteristic of severely disturbed freshwater habitats (Adamus and Brandt, 1990; Beyene et al., 2009; Smith et al., 2009; Ryder et al., 2011). Organic matter when present in SS is biologically active thus contributing to the oxygen consumption in streams during decomposition (Paavilainen and Paivanen, 1995; Greig et al., 2007; Rodgers et al., 2010). Organisms physiologically adapted to low oxygen tension exploit the excess nutrients available and thus dramatically increase in abundance (Miserendino et al., 2011). A corresponding reduction in EPT, macroinvertebrate diversity and species richness was also observed post-felling.

Even though forest harvesting increased the phosphorous concentrations and water temperature, no significant changes were observed in the diatom assemblages. Thus H_2 was rejected. Diatoms have previously been demonstrated to be sensitive indicators to many kinds of disturbances in streams as they respond quickly to changes in water quality (Smol and Stoermer, 2010). Phosphorus above a concentration of about $30\ \mu g\ L^{-1}$ can trigger eutrophication in freshwaters (Carpenter et al., 1998; Boesch et al., 2001). Many previous studies have shown marked changes of diatom assemblages post-harvesting (Naymik et al.,

2005; Yang et al., 2008; Wang et al., 2009). Lowe et al. (1986) observed higher abundances of 14 diatom species at harvested sites compared to the control sites 6 years after harvesting and attributed these differences to increased light availability. Phosphorus has been reported to be the most important variable in explaining diatom assemblages (Yang et al., 2008; Wang et al., 2009). Total phosphorus concentrations of 80–110 $\mu g\ L^{-1}$ could transform macrophyte dominated assemblages to alga-dominated assemblages in a lake (Yang et al., 2008). The expected shift from oligotrophic species to nutrient tolerant species did not happen in the impact streams. It is probable that because of the acidic nature of the catchment and the associated diatom assemblages the succession of more nutrient tolerant diatoms was prevented. O'Driscoll et al. (2012) investigated diatom assemblages in similar peatland catchments and found that alkalinity and conductivity were the main physicochemical drivers of the diatom assemblages and nutrient enrichment from forestry activities did not appear to have an influence on the diatom assemblages.

Recently, Schneider et al. (2013) highlighted uncertainties in diatom biomonitoring protocols in relation to interactions between pH and nutrients. They found that acid-tolerant taxa are generally associated with nutrient-poor conditions by the nature of habitat rather than true ecological niches. This has been reinforced by the findings in this study. The impact streams in this study are almost exclusively dominated by the *Eunotia exigua* complex both before and after felling. *Eunotia exigua* has been reported to dominate under high pollution stress (Passy, 2006). It has been suggested that pollution-tolerant diatoms may be tolerant to a broad range of chemical stressors (Guasch et al., 1998; Passy, 2006) which could explain the high numbers of *Eunotia exigua* prevailing under higher phosphate concentrations. These findings highlight the caution that should be placed on biomonitoring field surveys when inferring chemical preferences by associations between taxa and water chemistry. Additionally, the Q-index developed for water ecological status in inland waters identified a reduction in status class as defined by the WFD in the impact streams following clearfelling. The TDI-EQR classified these streams as high status and did not recognise an impact due forest clearfelling. However, this result should be treated with a certain amount of caution as the TDI-EQR was not calibrated for low alkalinity sites (i.e. <6.8 mg L^{-1} $CaCO_3$) (O'Driscoll et al., 2012).

Elevated levels of sunlight and increased stream water temperature have been reported to increase algal production (Lam, 1981; Holopainen and Huttunen, 1998). Periphyton biomass however, did not significantly increase in the impact streams after clearfelling. It is possible that high suspended sediment outwash from the clearfelling

disturbance, diminished the light penetration in the stream water, thus reducing algal production (Holopainen and Huttunen, 1998). However, the irruption of chironomids after felling indicated that it was more likely that the felling in this case increased grazing thus lowering biomass (Nislow and Lowe, 2006; Kobayashi et al., 2010). Functional feeding groups shifted from shredders and collector filterers to a complete dominance of collector–gatherers, therefore, H_3 was rejected. The shift was driven by chironomids, shown to be omnivorous and opportunistic in their food preferences (Marziali et al., 2009; Smith et al., 2009).

This study highlighted that in these cases of felling relatively small coupes of trees the response observed at the study site was not propagated to the larger catchment scale, thus H_4 was accepted. Care should be taken in interpreting this on a larger scale or in the case of harvesting of multiple small coups, in space or time but it does give some guidance on minimising downstream impacts in acid sensitive areas. Further work is required focusing on a range of catchment sizes and dilution factors of receiving rivers. The management strategy does not reduce the total P load leaving the harvested catchment (Rodgers et al., 2010) which is particularly important for catchments draining into sensitive lakes which act as a natural sink for any increased run off of nutrients and sediment (Drinan et al., 2013). Furthermore, this study provides valuable evidence on the role of headwater streams in detecting forest harvesting impacts, which were not detected in the main river channel, but have been shown to induce significant changes in lake water chemistry (Drinan et al., 2013). This highlights the need for the further development of mitigation methods such as the novel grass seeding method (O'Driscoll et al., 2011) which target minimising the nutrient source before it enters the receiving waters thus protecting these unique and sensitive ecosystems.

5. Conclusion

The findings of this study suggest that periphytic diatoms in acid sensitive blanket peat fed catchments are insensitive to forest clearfelling impacts. This confirms that bio-assessment protocols should focus on multiple groups in assessing environmental pressures because of interactions between groups (Heino, 2010; Kelly, 2011), but also because of possible limitations of individual groups (Schneider et al., 2013). The lack of impact on diatom assemblages is likely due to the natural acidic nature of these streams and diatom assemblages that are already acid stressed. The findings of this study suggest that the anticipated response of forest harvesting on the biota is dependent on the underlying hydro-morphology of the streams rather than being consistent with a trophic or an acidic response (Drinan et al., 2013). Harvesting a smaller coupe size was an efficient management practice

in protecting the larger salmonid river against the additional nutrient and suspended sediment input post-harvesting although the cumulative impact of harvesting multiple coupes in the one catchment compared to a single larger one-off harvest was not evaluated in this study. There is a need for the further development of mitigation methods which reduce the nutrient export from these catchments thus protecting the sensitive receiving lakes from nutrient loading.

Acknowledgements

This study was funded by the SANIFAC and For Site projects which are funded by the Department of Agriculture, Fisheries and the Marine under the STIMULUS Programme 2007–2013. The authors gratefully acknowledge the assistance of the EPA, the MI and Coillte.

Appendix A. Supplementary material

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.foreco.2013.09.018.

References

Adamus, P.R., Brandt, K., 1990. Impacts on Quality of Inland Wetlands of the United States: A Survey of Indicators, Techniques, and Applications of Community Level Biomonitoring Data, EPA/600/3-90/073. USEPA Environmental Research Lab, Corvallis, Oregon, pp. 406.

Allott, N., Brennan, M., Cooke, D., Reynolds, J., Simon, N., 1997. AQUAFOR Report 4. Stream Chemistry, Hydrology and Biota Galway-Mayo region. A Study of the Effects of Stream Hydrology and Water Chemistry in Forested Catchments on Fish and Macroinvertebrates, COFORD, Dublin, Ireland, pp. 1–158.

Anderson, M.J., 2001. A new method for non-parametric multivariate analysis of variance. Austral Ecol. 26, 32–46.

Anderson, M.J., 2005. PERMANOVA: A FORTRAN Computer Program For Permutational Multivariate Analysis of Variance. Department of Statistics, University of Auckland, New Zealand.

Andren, C., Jarlman, A., 2008. Benthic diatoms as indicators of acidity in streams. Fundam. Appl. Limnol. 173, 237–253.

APHA, 1998. Standard Methods for the Examination of Water and Wastewater, 20th ed. American Public Health Association, Washington, DC.

Bennett, L.T., Adams, M.A., 2004. Assessment of ecological effects due to forest

harvesting: approaches and statistical issues. J. Appl. Ecol. 41, 585–598.

Beyene, A., Addis, T., Kifle, D., Legesse, W., Kloos, H., Triest, L., 2009. Comparative study of diatoms and macroinvertebrates as indicators of severe water pollution: case study of the Kebena and Akaki rivers in Addis Ababa, Ethiopia. Ecol. Ind. 9, 381–392.

Boesch, D.F., Brinsfield, R.B., Magnien, R.E., 2001. Chesapeake Bay eutrophication: scientific understanding, ecosystem restoration, and challenges for agriculture. J. Environ. Qual. 30, 303–320.

Bowman, M.F., Chambers, P.A., Schindler, D.W., 2007. Constraints on benthic algal response to nutrient addition in oligotrophic mountain rivers. River Res. Appl. 23 (8), 858–876.

Broadmeadow, S., Nisbet, T.R., 2004. The effects of riparian management on the freshwater environment: a literature review of best management practices. Hydrol. Earth Syst. Sci. 8, 286–305.

Carpenter, S.R., Caraco, N.F., Correll, D.L., Howarth, R.W., Sharpley, A.N., Smith, V.H., 1998. Non-point pollution of surface waters with phosphorus and nitrogen. Ecol. Appl. 8, 559–568.

CEN, 2004. Water Quality – Guidance Standard for The Identification, Enumeration and Interpretation of Benthic Diatom Samples From Running Waters. EN 14407: 2004. Comite European de Normalisation, Geneva.

Chen, G., Dalton, C., Leira, M., Taylor, D., 2008. Diatom based total phosphorus (TP) and pH transfer functions for the Irish Ecoregion. J. Paleolimnol. 40, 143–163.

Chizinski, C.J., Vondracek, B., Blinn, C.R., Newman, R.M., Atuke, D.M., Fredricks, K., Hemstad, N.A., Merten, E., Schlesser, N., 2010. The influence of partial timber harvesting in riparian buffers on macroinvertebrate and fish communities in small streams in Mennesota, USA. For. Ecol. Manage. 259, 1946–1958.

Clarke, K.R., Warwick, R.M., 1994. Change in Marine Communities. Plymouth Marine Laboratory, pp. 144.

Clarke, R.T., Hering, D., 2006. Errors and uncertainty in bioassessment methods – major results and conclusions from the STAR project and their application using STARBUGS. Hydrobiologia 566, 433–439.

Clarke, K.R., 1993. Non-parametric multivariate analyses of changes in community structure. Aust. J. Ecol. 18, 117–143.

Cruikshanks, R., Lauridsen, R., Harrison, A., Hartl, M.G.J., Kelly-Quinn, M., O'Halloran, J., Giller, P.S., 2008. Putting the sodium dominance index to the test as a measure of acid sensitivity across diverse geological conditions and with reference to the influence of plantation forests. Water Air Soil Pollut. 190, 221– 229.

Cummins, T., Farrell, E.P., 2003. Biogeochemical impacts of clearfelling and reforestation on blanket peatland streams I. Phosphorus. Forest Ecol. Manage. 180 (1–3), 545–555.

Dall, P.C., 1979. A sampling technique for littoral stone dwelling organisms. Oikos 33, 106–112.

Drinan, T.J., Graham, C.T., O'Halloran, J., Harrison, S.S.C., 2013. The impact of catchment conifer plantation forestry on the hydrochemistry of peatland lakes. Sci. Total Environ. 443, 603–620.

EPA, 2004, 2004. Eutrophication of Inland and Estuarine Waters. In The Report: Ireland's Environment. Environmental Protection Agency, Dublin, Ireland.

European Union, 2000. Directive 2000/60/EC of the European parliament and of the council of 23 October 2000 establishing a framework for community action in the field of water policy. Official J. Euro. Commun. L327, 1–73.

European Communities, 2003. Council directive 92/43/EEC of 21May 1992 on the conservation of natural habitats and of wild fauna and flora. As amended by the Accession Act of Austria, Finland and Sweden (1995) and the Accession Act of the Czech Republic, the Republic of Estonia, the Republic of Cyprus, the Republic of Latvia, the Republic of Lithuania, the Republic of Hungary, the Republic of Malta, the Republic of Poland, the Republic of Slovenia and the Slovak Republic. Official Journal of the European Union L 236 33 23.9.2003. Brussels. European Commission 1992/95/2003. 2003b.

Forest Service, 2008. Indicative Forestry Statement, The Right Trees in The Right Places. Department of Agriculture, Fisheries and Food. Report.

Fuchs, S.A., Hinch, S.G., Mellina, E., 2003. Effects of streamside logging on stream macroinvertebrate communities and habitat in the sub-boreal forests of British Columbia Canada. Can. J. For. Res. 33 (8), 1408–1415.

Greig, S.M., Sear, D.A., Carling, P.A., 2007. A review of factors influencing the availability of dissolved oxygen to incubating salmonid embryos. Hydrol. Process. 21, 323–334.

Growns, I.O., Davis, J.A., 1994. Effects of forestry activities (clearfelling) on stream macroinvertebrate fauna in South-Western Australia. Aust. J. Mar. Freshw. Res. 45, 963–975.

Guasch, H., Ivorra, N., Lehmann, V., Paulsson, M., Real, M., Sabater, S., 1998. Community composition and sensitivity of periphyton to atrazine in flowing waters: the role of environmental factors. J. Appl. Phycol. 10, 203–213.

Heino, J., 2010. Are indicator groups and cross-taxon congruence useful for predicting biodiversity in aquatic ecosystems? Ecol. Ind. 10 (2), 112–117.

Hendley, N.I., 1974. The permanganate method for cleaning freshly gathered diatoms. Microscopy 32, 423–426.

Henrikson, L., Medin, M., 1986. Biologisk Bedomning av Forsurningspaverkan Palelangens Illfloden Och Grundomraden. Report to Alvsborgs County Administrative Board, Aquaekologerna (in Swedish). Holopainen, A.-L., Huttunen, P., 1998. Impact of forestry practices on ecology of algal communities in small brooks in the Nurmes experimental forest area, Finland. Boreal Environ. Res. 3, 63–73.

Hynes, H.B.N., 1994. Historical perspective and future direction of biological monitoring of aquatic systems. In: Loeb, S.L., Spacie, A. (Eds.), Biological monitoring of aquatic systems. Lewis Publishers, Boca Raton, pp. 11–22.

Johnson, S.L., Jones, J.A., 2000. Stream temperature responses to forest harvest and debris flows in western Cascades, Oregon. Can. J. Fish. Aquat. Sci. 57, 30–39.

Juttner, I., Ector, L., Reichardt, E., Van de Vijver, B., Jarlman, A., Krokowski, J., Cox, E.J., 2013. Gomphonema varioreduncum sp. nov., a new species from northern and western Europe and a re-examination of Gomphonema exilissimum. Diatom Res. 28 (3), 303–316.

Kahlert, M., Kelly, M., Albert, R.-L., Almeida, S.F.P., Bešta, T., Blanco, S., Coste, M., Denys, L., Ector, L., Frankova, M., Hlubikova, D., Ivanov, P., Kennedy, B., Marvan, P., Mertens, A., Miettinen, J., Picinska-Fałtynowicz, J., Rosebery, J., Tornes, E., Vilbaste, S., Vogel, A., 2012. Identification versus counting protocols as sources of ncertainty in diatom-based ecological status assessments. Hydrobiologia 695, 109–124.

Kedzierski, W.M., Smock, L.A., 2001. Effects of logging on macroinvertebrate production in a sand-bottomed, low gradient stream. Freshw. Biol. 46, 821–833.

Kelly, M.G., Cazaubon, A., Coring, E., Dell'Uomo, A., Ector, L., Goldsmith, B., Guasch, H., Hurlimann, J., Jarlman, A., Kawecka, B., Kwandrans, J., Laugaste, R., Lindstrom, E.-A., Leitao, M., Marvan, P., Padisak, J., Pipp, E., Prygiel, J., Rott, E.,

Sabater, S., van Dam, H., Vizinet, J., 1998. Recommendations for the routine sampling of diatoms for water quality assessments in Europe. J. Appl. Phycol. 10, 215–224.

Kelly, M., Juggins, S., Guthrie, R., Pritchard, S., Jamieson, J., Rippey, B., Hirst, H., Yallop, M., 2008. Assessment of ecological status in U.K. rivers using diatoms. Freshw. Biol. 53, 403–422.

Kelly, M., 2011. The Emperor's new clothes? a comment on Besse-Lototskaya et al. (2011). Ecol. Ind. 11, 1492–1494.

Kelly-Quinn, M., Tierney, D., Coyle, C., Bracken, J.J., 1996. Factors affecting the susceptibility of Irish soft-water streams to forest-mediated acidification. Fish Manage Ecol. 3, 287–301.

Kelly-Quinn, M., Baars, J.R., Bradley, C., Dodkins, I., Harrington, T.J., Ni Chathain, B., O'Connor, M., Rippey, B., Trigg, D., 2004. Characterisation of Reference Conditions and Testing of Typology of Rivers (RIVTYPE). Draft report to the EPA.

Kelly-Quinn, M., Cruikshanks, R., Johnson, J., Matson, R., Baars, J-R., Bruen, M., 2008. Forestry and surface water acidification. Ireland: Report to the Western River Basin District Working Group, pp. 81. <http://www.wfdireland.ie/docs/ 22_ForestAndWater/>

Kobayashi, S., Gomi, T., Sidle, R.C., Takemon, Y., 2010. Disturbances structuring macroinvertebrate communities in steep headwater streams: relative importance of forest clear cutting and debris flow occurrence. Can. J. Fish. Aquat. Sci. 67 (2), 427–444.

Krammer, K., Lange-Bertalot, H., 1986. Die Susswasserflora von Mitteleuropa 2: Bacillariophyceae. Teil 1: Naviculaceae. Gustav Fisher Verlag, Stuttgart, pp. 876.

Krammer, K., Lange-Bertalot, H., 1997. Die Susswasserflora von Mitteleuropa, II: 2. Bacillariophyceae. Teil 2: Bacillariaceae, Epithemiaceae, Surirellaceae. 2te Auflage. Gustav Fisher Verlag, Stuttgart, pp. 594.

Krammer, K., Lange-Bertalot, H., 2000. Die Susswasserflora von Mitteleuropa 2: Bacillariophyceae. Teil 3: Centrales, Fragilariaceae, Eunotiaceae. 2te Auflage. Gustav Fischer Verlag, Stuttgart.

Krammer, K., Lange-Bertalot, H., 2004. Die Susswasserflora von Mitteleuropa 2: Bacillariophyceae. Teil 4: Achnanthaceae. Kritische Erganzungen zu Achnanthes s.l., Navicula s. str., Gomphonema. Spektrum Akademischer Verlag/Gustav Fisher, Heidelberg, pp. 468.

Lam, W.Y.C., 1981. Ecological studies of phytoplankton in the Waikato River and its catchment. New Zealand. J. Mar. Freshwater Res. 15, 95–103.

Laudon, H., Sponseller, R.A., Lucas, R.W., Futter, M.N., Egnell, G., Bishop, K., Agren, A., Ring, E., Hogberg, P., 2011. Consequences of more intensive forestry for the sustainable management of forest soils and waters. Forests 2, 243–260.

Layer, K., Hildrew, A.G., Woodward, G., 2013. Grazing and detritivory in 20 stream food webs across a broad pH gradient. Oecologia 171, 459–471.

Leira, M., Sabater, S., 2005. Diatom assemblages distribution in Catalan rivers, NE Spain, in relation to chemical and physiographical factors. Water Res. 39 (1), 73–82.

LeRoy Poff, N., Olden, J.D., Vieira, N.K.M., Finn, D.S., Simmons, M.P., Kondratieff, .C., 2006. Functional trait niches of North American lotic insects: traits-based ecological applications in light of phylogenetic relationships. J. North Am. Benthol. Soc. 25 (4), 730–755.

Liljaniemi, P., Vouri, K., Ilyashuk, B., Luotonen, H., 2002. Habitat characteristics and macroinvertebrate assemblages in boreal forest streams: relations to catchment silviculture activities. Hydrobiologia 474, 239–251.

Louhi, P., Maki-Petays, A., Erkinaro, J., Paasivaara, A., Muotka, T., 2010. For. Ecol. Manage. 260, 1315–1323.

Lowe, R.L., Golladay, S.W., Webster, J.R., 1986. Periphyton response to nutrient manipulation in streams draining clearcut and forested watersheds. J. North Am. Benthol. Soc. 5, 221–229.

Mainstone, C.P., Parr, W., 2002. Phosphorus in rivers – ecology and management. Sci. Total Environ. 282, 25–47.

Marziali, L., Giordano Armanini, D., Cazzola, M., Erba, S., Toppi, E., Buffagni, A., Rossaro, B., 2009. Responses of chironomid larvae (Insecta, Diptera) to ecological quality in Mediterranean river mesohabitats (South Italy). River Res. Appl. 26, 1036–1051.

McArdle, B.H., Anderson, M.J., 2001. Fitting multivariate models to community data: a comment on distance-based redundancy analysis. Ecology 82, 290–297.

McGarrigle, M.L., Bowman, J.J., Clabby, K.J., Lucey, J., Cunningham, P., MacCarthaigh, M., Keegan, M., Cantrell, B., Lehane, M., Clenaghan, C., Toner, P.F., 2002. Water Quality in Ireland. EPA, Wexford.

Miserendino, M.L., Casaux, R., Archangelsky, M., Di Prinzio, C.Y., Brand, C., Kutschker, A.M., 2011. Assessing land – use effects on water quality, in – stream habitat, riparian ecosystems and biodiversity in Patagonian northwest streams. Sci. Total Environ. 409, 612–624.

Muller, M., 2000. Hydrogeographical studies in the Burrishoole catchment, Newport, Co. Mayo, Ireland: affects of afforestation on the run-off regime of small mountain spate river catchments. Verhandlung Int. Vereinigung Limnol. 27, 1146–1148.

Naymik, J., Pan, Y., Ford, J., 2005. Diatom assemblages as indicators of timber harvest effects in coastal Oregon streams. J. North Am. Benthol. Soc. 24 (3), 569–584.

Nieminen, M., 2003. Effects of clearcutting and site preparation on water quality from a drained Scots pine mire in southern Finland. Boreal Environ. Res. 8, 53– 59.

Nislow, K.H., Lowe, W.H., 2006. Influences of logging history and riparian forest characteristics on macroinvertebrates and brook trout (Salvelinus fontinalis) in headwater streams (New Hampshire, USA). Freshw. Biol. 51 (2), 388–397.

Noel, D.S., Martin, C.W., Federer, C.A., 1986. Effects of forest clearcutting in New England on stream macroinvertebrates and periphyton. Environ. Manage. 10 (5), 661–670.

O'Driscoll, C., Rodgers, M., O'Connor, M., Asam, Z.-Z., de Eyto, E., Poole, R., Xiao, L., 2011. A potential solution to mitigate phosphorus release following clearfelling in peatland catchments. Water Air Soil Pollut. 221, 1–11.

O'Driscoll, C., de Eyto, E., Rodgers, M., O'Connor, M., Asam, Z-Z., Xiao, L., 2012. Diatom assemblages and their associated environmental factors in upland peat forest rivers. Ecol. Ind. 18, 443–451.

Ormerod, S.J., Rundle, S.D., Lloyd, E.C., Douglas, A.A., 1993. The influence of riparian management on the habitat structure and macroinvertebrate communities of upland streams draining plantation forests. J. Appl. Ecol. 30, 13–24.

Paavilainen, E., Paivanen, J., 1995. Peatland Forestry: Ecology and Principles. Springer, Berlin.

Passy, S.I., 2006. Diatom community dynamics in streams of chronic and episodic acidification: the roles of the environment and time. J. Phycol. 42, 312–323.

Potapova, M.G., Charles, D.F., 2002. Benthic diatoms in USA rivers: distributions along spatial an environmental gradients. J. Biogeogr. 29, 167–187.

Reid, D.J., Quinn, J.M., Wright-Stow, A.E., 2010. Responses of stream macroinvertebrate communities to progressive forest harvesting: influences of harvest intensity, stream size and riparian buffers. For. Ecol. Manage. 260 (10), 1804–1815.

Richardson, J.S., Danehy, R.J., 2007. A synthesis of the ecology of headwater streams and their riparian zones in temperate forests. Forest Sci. 53, 131–147.

Richter, B.D., Baumgartner, J.V., Powell, J., Braun, D.P., 1996. A method for assessing hydrological alteration within ecosystems. Conserv. Biol. 10 (4), 1163–1174.

Riipinen, M.P., Dobson, M., 2010. Benthic organic matter biomass and invertebrate community structure in five conifer plantation streams in the peak district (Derbyshire, England). Freshwater Forum 28, 61–75.

Rodgers, M., Xiao, L., Muller, M., O'Connor, M., de Eyto, E., Poole, R., Robinson, M., Healy, M., 2008. Quantification of Erosion and Phosphorus Release from a Peat Soil Forest Catchment. (EPA STRIVE Report). Published by Environmental Protection Agency, Ireland.

Rodgers, M., O'Connor, M., Healy, M.G., O'Driscoll, C., Asam, Z., Nieminen, M., Poole, R., Muller, M., Xiao, L., 2010. Phosphorus release from forest harvesting on an upland blanket peat catchment. For. Ecol. Manage. 260 (12), 2241–2248.

Rodgers, M., O'Connor, M., Robinson, M., Muller, M., Poole, R., Xiao, L., 2011. Suspended solid yield from forest harvesting on upland blanket peat. Hydrol. Process. 25, 207–216.

Ryder, L., de Eyto, E., Gormally, M., Sheehy-Skeffington, M., Dillane, M., Poole, R., 2011. Riparian zone creation in established coniferous forests in Irish upland peat catchments: physical, chemical and biological implications. Biol. Environ.: Proc. Roy. Irish Acad. 111B, 1–20.

Schneider, S.C., Kahlert, M., Kelly, M.G., 2013. Interactions between pH and nutrients on benthic algae in streams and consequences for ecological status assessment and species richness patterns. Sci. Total Environ. 444, 73–84.

Shannon, C.E., Weaver, W., 1963. The Mathematical Theory of Communication, Urbana, pp. 125.

Smucker, N.J., Vis, M.L., 2011. Diatom biomonitoring of streams: reliability of reference sites and the response of metrics to environmental variations across temporal scales. Ecol. Ind. 11, 1647–1657.

Smith, B.J., Davies, P.E., Munks, S.A., 2009. Changes in benthic macroinvertebrate communities in upper catchment streams across a gradient of catchment forest operation in history. For. Ecol. Manage. 257 (10), 2166–2174.

Smol, J. P., Stoermer, E. F. (Eds.). 2010. The diatoms: applications for the environmental and earth sciences. Cambridge University Press.

Tamm, C.O., Holman, H., Popovic, B., Wiklander, G., 1974. Leaching of plant nutrients from soils as a consequence of forest operations. Ambio 3, 221–222.

Toner, P., Bowman, K., Clabby, K., Lucey, J., McGarrigle, M., Concannon, C., Clenaghan, C., Cunningham, P., Delaney, J., O'Boyle, S., MaCarthaigh, M., Craig, M., Quinn, R., 2005. Water Quality in Ireland 2001–2003. Environmental Protection Agency, Wexford.

Wallace, J.B., 1990. Recovery of lotic macroinvertebrate communities from disturbance. Environ. Manage. 14 (5), 605–620.

Wang, Q., Zhi, C., Hamilton, P., Kang F., 2009. Diatom distributions and species optima for phosphorus and current velocity in rivers from ZhuJiang Watershed within a Karst region of South-Central China. Fundam. Appl. Limnol. Archiv fur Hydrobiol. 175 (2), pp. 125–141(17).

Yang, X., Anderson, J., Dong, X., Shen, J., 2008. Surface sediment diatom assemblages and epilimnetic total phosphorus in large, shallow lakes of the Yangtze floodplain: their relationships and implications for assessing long-term eutrophication. Freshwater Biol. 53 (7), 1273–1290.

Table 1 Details on study site catchment locations, harvesting and physical and chemical characteristics pre-felling

Stream Management details	Latitude (N)	Longitude (W)	Altitude (m)	Upstream Catchment Area (ha)	% of Catchment felled	Harvest date	Temperature (°C)	Conductivity (µS cm^{-1})	pH	Colour (PtCo)	Alkalinity (mg L_{-1} $CaCO^3$)	PO_4-P (µg L^{-1})	NH4-N (µg L^{-1})
Control													
GH	53°56'49.8"	9°39'31.3"	101	10	0	-	9.9 *(2.6)*	68 *(10)*	5.2 *(0.7)*	69 *(5.7)*	0.8 *(0.3)*	5.81 *(1.8)*	31.1 *(28.2)*
SCS	53°59'19.6"	9°31'21.8"	240	23	0	-	9.92 *(2.7)*	84 *(13)*	6.3 *(0.3)*	143 *(26.8)*	9.35 *(4.5)*	7.51 *(1.5)*	42.36 *(39.8)*
GCS	53°58'11.3"	9°36'55.5"	56	10	0	-	10.4 *(2.5)*	128 *(21)*	3.9 *(0.5)*	209 *(4.9)*	-4.21 *(2.6)*	12.2 *(7.5)*	44.36 *(32.4)*
Treatment													
TEEV1	53°57'41.7"	9°31'15.0"	280	20	75	July-August 2009	10.1 *(3.8)*	123.5 *(2)*	4.1 *(0.7)*	85 *(7.7)*	1.37 *(0.4)*	3.20 *(0.2)*	120 *(41.3)*
TEEV2	53°57'39.4"	9°31'08.8"	268	32	30	July-August 2009	10.15 *(3.8)*	108 *(19)*	5.1 *(1.0)*	61 *(2.8)*	1.76 *(1.1)*	2.58 *(0.9)*	53.6 *(36.1)*
GSS	53°58'02.5"	9°36'42.8"	87	10	100	January-February 2011	9.73 *(2.6)*	145 *(44)*	4.0 *(1.0)*	242 *(23.5)*	-2.27 *(0.6)*	7.10 *(2.6)*	37.4 *(16.4)*
SSS	53°59'32.0"	9°31'50.1"	220	10	50	January-February 2011	9.47 *(3.4)*	125 *(34)*	4.0 *(0.6)*	208 *(71)*	-2.33 *(1.5)*	9.80 *(2.4)*	58.55 *(35.4)*
Main river													
GLa	53°58'18.0"	9°37'02.4"	107	1135	0	-	10.65 *(3.9)*	63 *(6)*	5.9 *(0.9)*	86 *(25)*	6.7 *(0.3)*	7.42 *(1.5)*	30.8 *(8.8)*
GLb	53°58'01.8"	9°36'35.2"	50	1145	1	-	11.54 *(4.5)*	72 *(7)*	6.7 *(1.0)*	78 *(28)*	10.7 *(0.1)*	6.5 *(1.1)*	33.0 *(10.1)*

The standard deviation is indicated by italics and ( ).

Table 2 a and b The dominant macroinvertebrate (a) and diatom (b) taxa in the control and study sites before and after clearfelling

	Dominant taxa in the control streams	% of the total abundance	Dominant taxa in the study streams	% of the total abundance
Pre-harvesting macroinvertebrates				
	Baetis rhodani (Pictet)	43.2	Nemoura cinerea (Retszius)	47.7
	Leuctra hippopus (Kempny)	27.5	Simulidae sp	19.5
	Amphinemura sulcicollis (Stephens)	5.3	*Leuctra hippopus* (Kempny)	12.7
	Simulidae sp	3.6	Chironomid sp	5.8
	Brachypteri risi (Morton)	3.5	*Polycentropus kingi* (McLachlan)	3.9
Post- harvesting macroinvertebrates				
	Baetis rhodani (Pictet)	41.5	Chironomid sp	98
	Simulidae sp	26.4	Nemoura cinerea (Retszius)	1.4
	Brachypteri risi (Morton)	11.6	*Polycentropus kingi* (McLachlan)	0.2
	Leuctra hippopus (Kempny)	9.3	*Chloroperla torrentium* (Pictet)	0.2
	Chloroperla torrentium (Pictet)	2.5	Tipulidae sp	0.2
Pre-harvesting diatoms				
	Achnanthes oblongella Østrup	59.7	*Eunotia exigua* complex	70.2
	Gomphonema parvulum complex	11.7	*Pinnularia appendiculata* (Agardh) Cleve	13.9
	Eunotia rhomboidea Hustedt	8.6	*Eunotia subarcuatoides* Alles, Norpel, Lange-Bertalot	2.3
	Eunotia exigua complex	6.0	*Eunotia bilunaris var. mucophila* Lange-Bertalot and Norpel	5.1
	*Achnanthidium minutissimum type 1 * sensu* Potapova et Hamilton	3.3	*Eunotia paludosa* Grunow	2.9
Post- harvesting diatoms				
	Achnanthes oblongella Østrup	58.5	*Eunotia exigua* complex	63.6
	Gomphonema parvulum complex	17.2	*Pinnularia appendiculata* (Agardh) Cleve	13.3
	Eunotia exigua complex	6.1	*Eunotia subarcuatoides* Alles, Norpel, Lange-Bertalot	7.9
	Pinnularia appendiculata (Agardh) Cleve	3.0	*Eunotia paludosa* Grunow	2.7
	*Achnanthidium minutissimum type 1 * sensu* Potapova et Hamilton	3.0	*Pinnularia subcapitata* Gregory	2.7

Table 3 a and b Permutation multivariate analysis of variance on macroinvertebrate and diatom assemblages among levels of the factors treatment (control and study) and time (before and after clearfelling)

	df	SS	MS	F	P(perms)
Macroinvertebrates					
Time	1	5978.20 (9.32)	5978.20	2.5717	0.025
Treatment	1	12219.00 (19.06)	12219.00	5.2566	0.001
Time x Treatment	1	6278.50 (9.78)	6278.50	2.7009	0.024
Residual	16	37193.00 (58.01)	2324.60		
Total	19	64119.00			
Diatoms					
Time	1	2656.90 (5.72)	2656.90	1.2323	0.261
Treatment	1	6083.20 (13.10)	6083.20	2.8216	0.087
Time x Treatment	1	1948.90 (4.20)	1948.90	0.9040	0.412
Residual	16	34495.00 (74.27)	2156.00		
Total	19	46445.00			

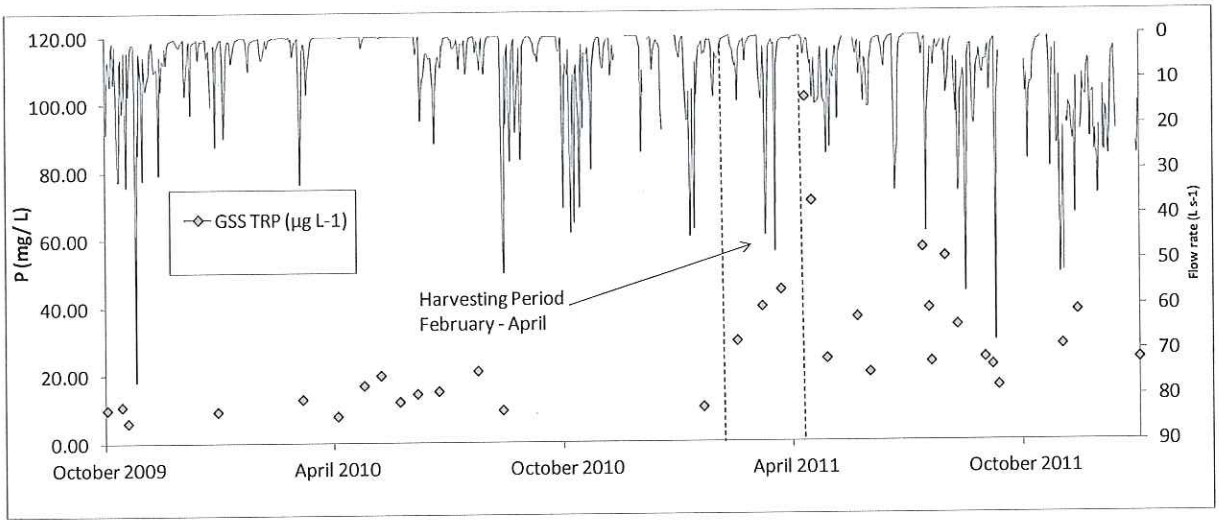


Figure 6

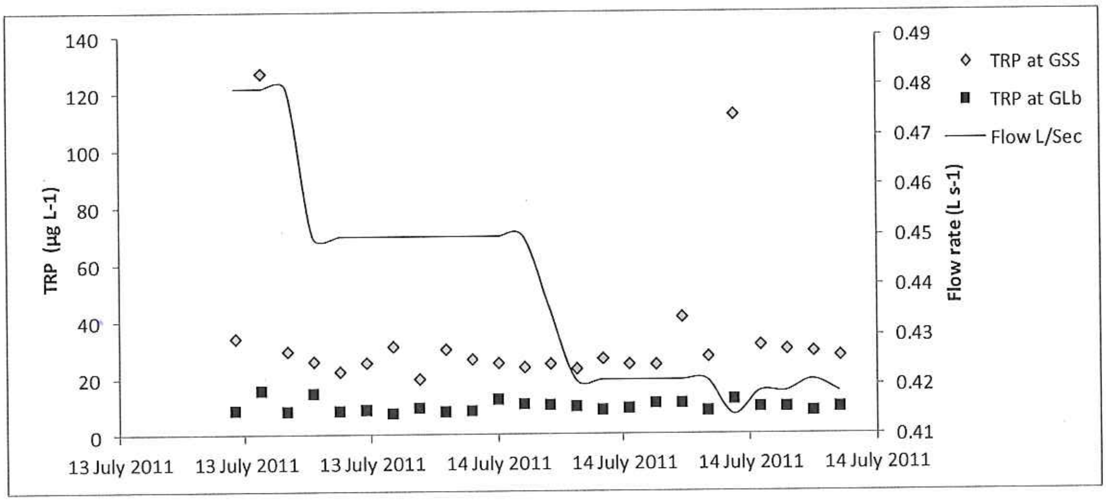


Figure 7

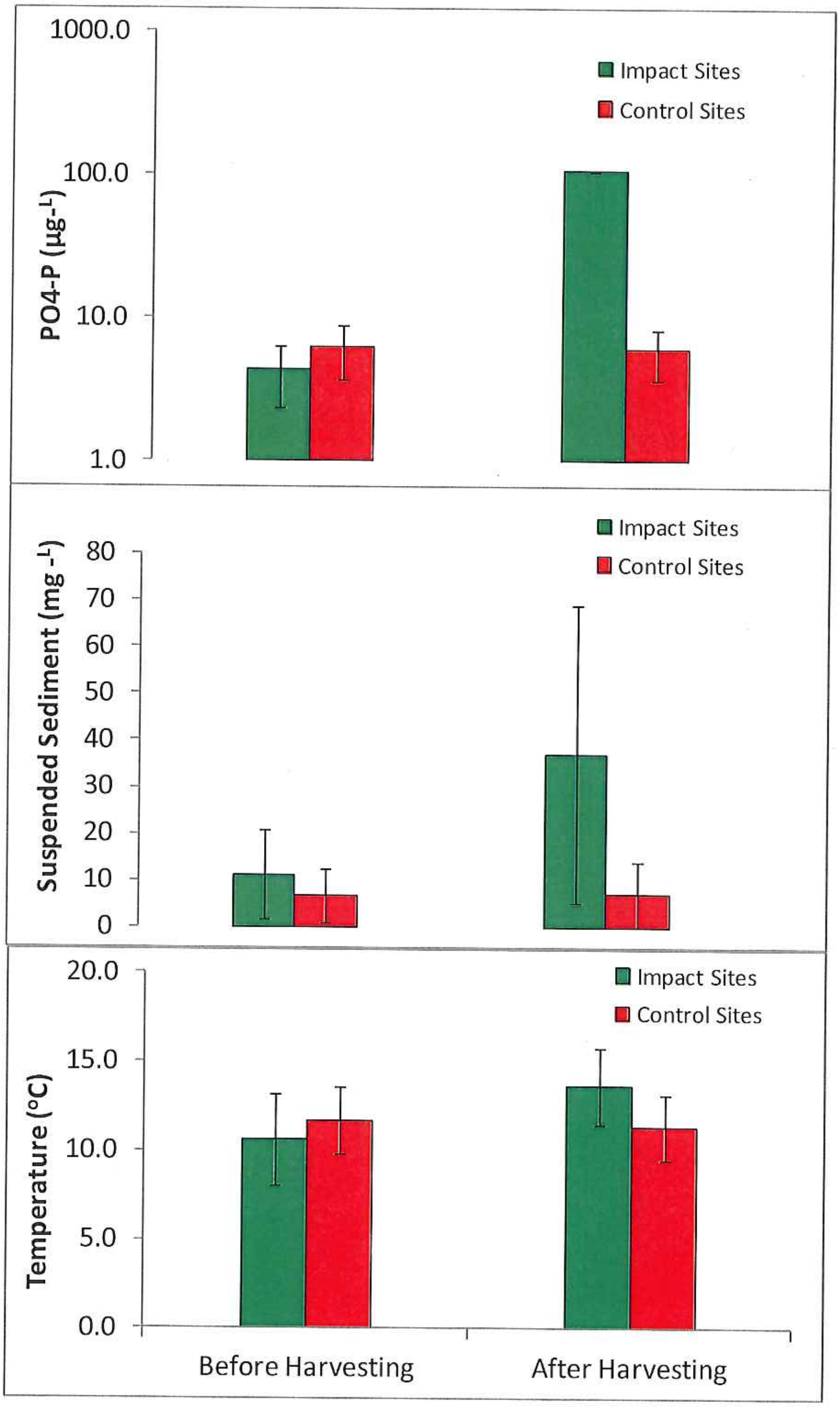
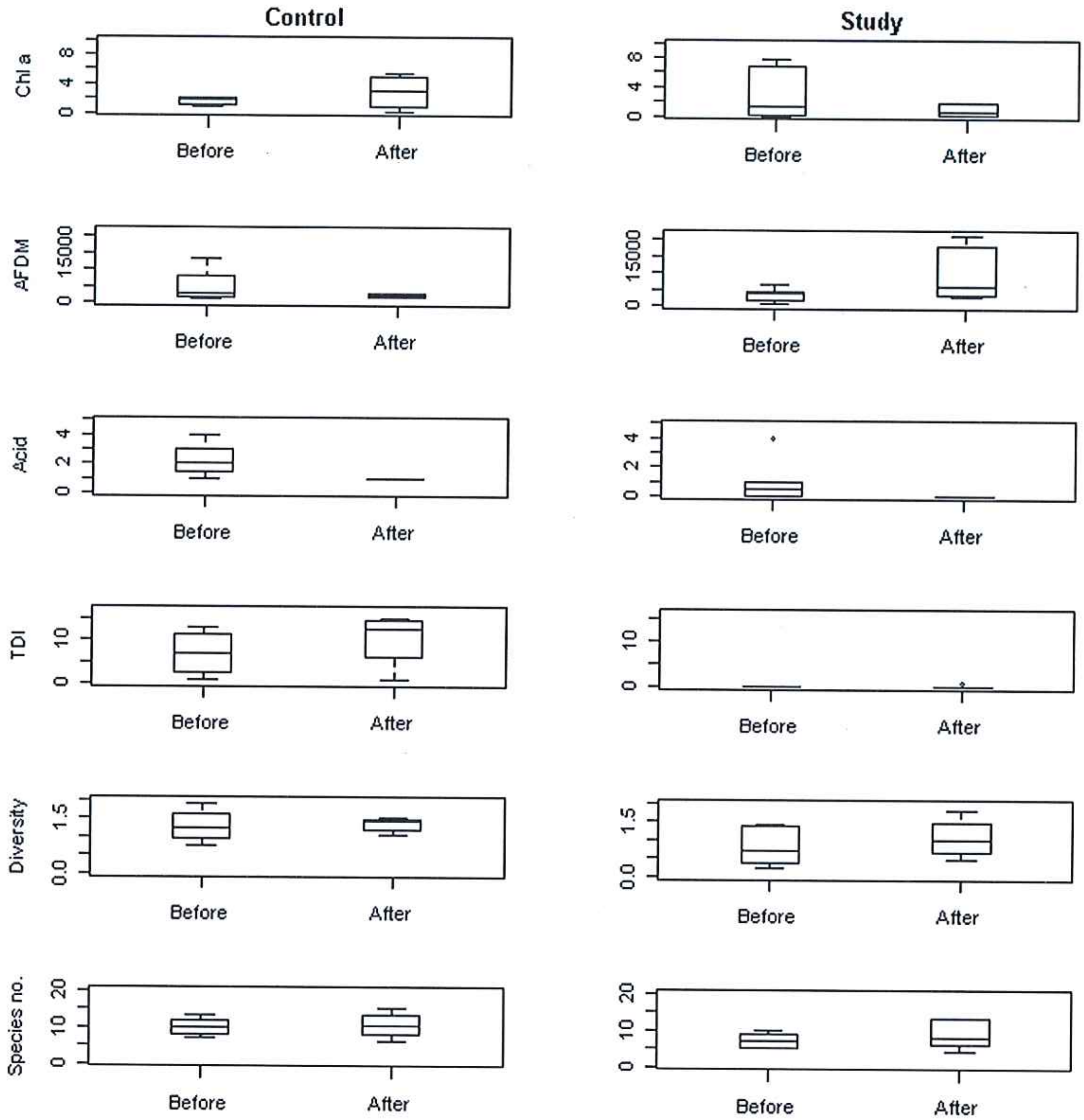

1000.0
100.0
10.0
1.0
PO4-P (µg-1)
Impact Sites
Control Sites
80
70
60
50
40
30
20
10
0
Suspended Sediment (mg -1)
Impact Sites
Control Sites
20.0
15.0
10.0
5.0
0.0
Temperature (°C)
Impact Sites
Control Sites
Before Harvesting
After Harvesting

Figure 5

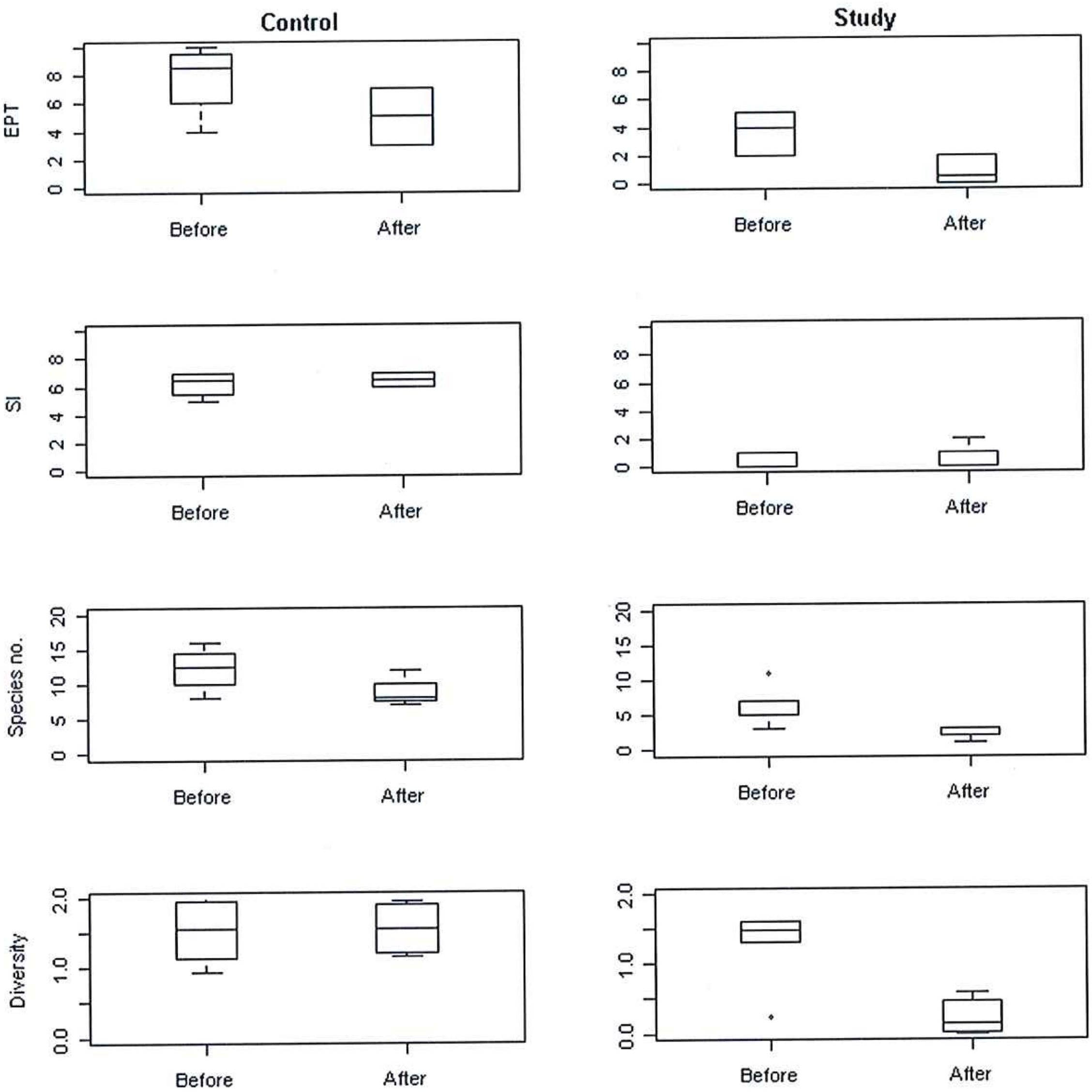
1
0.9
0.8
0.7
0.6
0.5
0.4
0.3
0.2
0.1
0
Relative Abundance
Before
After
Shredder
Predator
Herbivore
Collector-filterer
Collector-gatherer

Figure 4



b)

Figure 3 a and b



a)

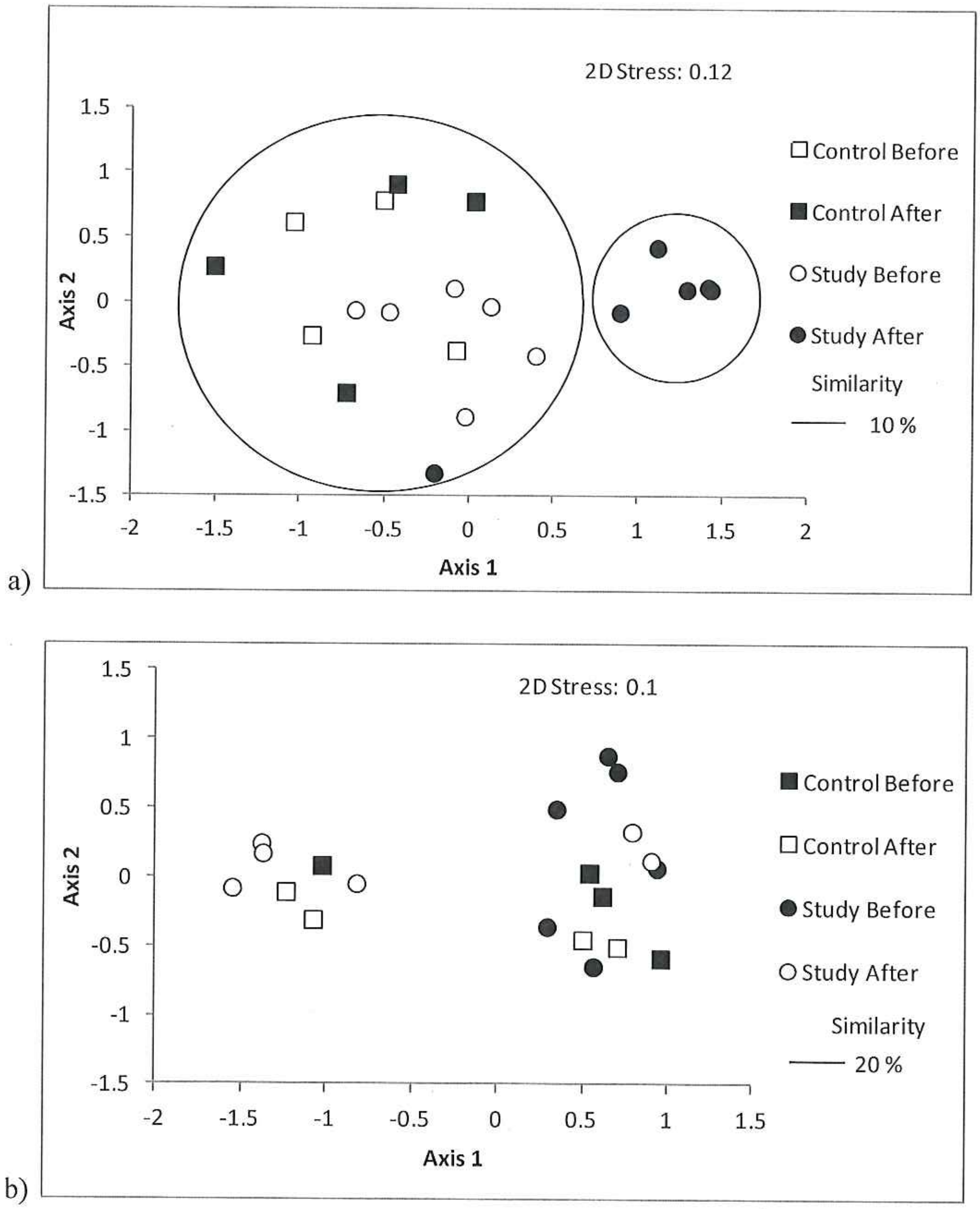
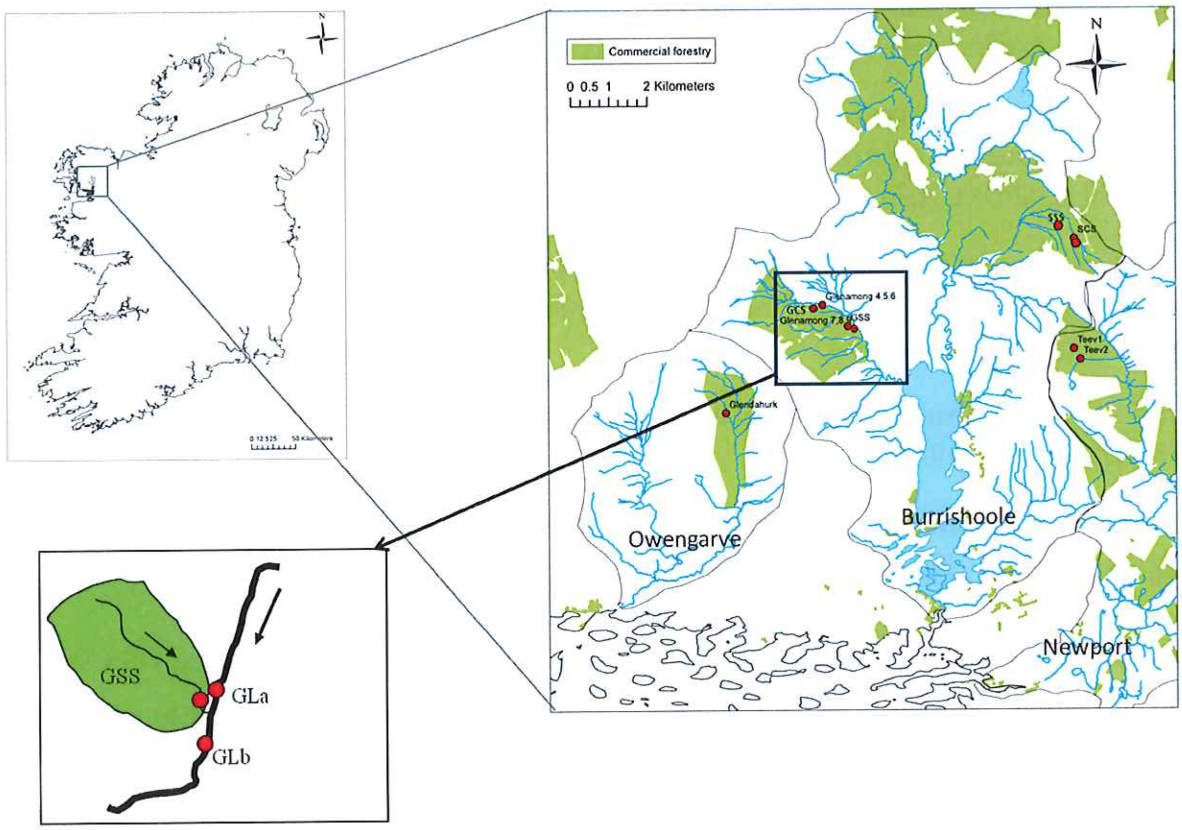


Figure 3a and b



N
Commercial forestry
0 0.5 1 2 Kilometers
SSS
SCS
Glenamong 4,5,6
GCS
Glenamong 7,8
GSS
Teev1
Teev2
Glendahurk
Owengarve
Burrishoole
Newport
GSS
GLa
GLb

Figure 1

Figure 1 Geographical location of the study streams and a schematic of the sampling sites above and below the confluence of the small first order stream and larger river.

Figure 2 a and b MDS ordination of Bray – Curtis similarities from (a) macroinvertebrates species and (b) diatom abundances data for the five study sites at two sampling times before and two after felling; with superimposed cluster analysis at indicated similarity levels.

Figure 3 a and b Comparison between treatments control and study, before and after harvesting of (a) macroinvertebrate indices (EPT, SI, Species no. and diversity) and of (b) phytobenthic indices (Chl a, AFDM, Acid, TDI, diversity and species no.)

Figure 4 Changes in the relative abundance of functional feeding groups in streams draining impact sites before and after clearfelling. Data for before and after are averages of the three impact streams in two seasons.

Figure 5 Significant changes in physical and chemical variables at the control and study sites before and after clearfelling

Figure 6 The daily flow and daily discharge-weighted mean total reactive phosphorus (TRP) concentrations at the GSS study stream during the study period.

Figure 7. The instantaneous total reactive phosphorus (TRP) concentrations at the GSS study stream and at the downstream of the confluence in the main Glennamong river (GLb) with GSS flow rate (Q) in a storm event.