An assessment of recovery targets, endpoints and timelines associated with the proposed restoration of Loch Fitty (Fife)

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SUMMARY

It is proposed that Loch Fitty will be drained and temporarily removed for a period of approximately 7.5 years, while the underlying coal is removed. Comprehensive planning and development documents have been prepared by the Scottish Coal Company Limited (SCCL) and its development team in relation to this proposal. A clear statement of intent in relation to recovery periods and end point water quality targets is now required. This report summarises the Water Framework Directive (WFD) targets to be met for the re-instated Loch Fitty to attain Good Ecological Status and makes recommendations on a recolonisation strategy designed to achieve this. Additional mitigation/monitoring measures that may be required to attain and sustain good status, and a likely timescale for recovery, are also discussed.

WFD ecological water quality targets for the reinstated Loch Fitty

During the reconstruction phase, Loch Fitty will be deepened in comparison to the existing loch. The new loch will have a mean depth of 3.6, whereas the existing loch has a mean depth of 2.3 m. This deepening changes the WFD lake type from high alkalinity, very shallow (HAVS) to high alkalinity shallow (HAS) and this is associated with more challenging WFD water quality targets. However, the benefits of deepening the loch, as explained in the planning and development documents, are recognised.

Total phosphorus: The procedures for setting site specific total phosphorus (TP) targets for lakes are well defined. For the re-instated Loch Fitty, these give annual average TP concentration targets of 22 $\mu g\ L^{-1}$ for the Good/Moderate boundary, 45 $\mu g\ L^{-1}$ for the Moderate/Poor boundary and 89 $\mu g\ L^{-1}$ for the Poor/Bad boundary.

Nitrogen: There are no agreed nitrogen (N) targets for lakes at the moment. However, a recent review by Maberly & Carvalho (2010) suggested a method for setting targets based on total nitrogen (TN) concentration. If this method is adopted, it would put the mean annual TN concentration of the Good/Moderate boundary for the reinstated Loch Fitty at 0.6 N mg L^{-1}.

Chlorophyll *a* concentration: Methods for assessing the status of phytoplankton in lakes under the WFD have been formally adopted for phytoplankton chlorophyll *a*. The growing season mean chlorophyll *a* standard for the re-instated Loch Fitty is 13.3 $\mu g\ L^{-1}$ for the Good/Moderate boundary. The annual mean chlorophyll *a* value for this boundary is 7.5 $\mu g\ L^{-1}$.

Phytoplankton: The phytoplankton trophic index (PTI) is still being developed and there are no class boundary values available at present. However, in terms of species composition, typical High/Good status phytoplankton taxa for high alkalinity lakes such as Loch Fitty would include chrysophytes such as *Bitrichia*, *Uroglena* and *Dinobryon*, and the diatoms *Tabellaria* and *Urosolenia*. In contrast, poor quality taxa, reflecting an “impacted” state, would include the green algae *Actinastrum* and *Scenedesmus*, and the cyanonbacteria *Microcystis*, *Planktothrix* and *Aphanizomenon*.

Cyanobacterial abundance: This metric is calculated from the total biovolume of cyanobacteria in a sample of lake water, and comprises the average value from samples collected over the summer period (i.e. July to September). For high alkalinity shallow lakes, such as the restored Loch Fitty, it is recommended that values of this metric should be $< 0.71\ mm^3\ L^{-1}$.

Phytobenthos: There is no metric available to assess the status of phytobenthos in lakes at present. However, a reference condition can be calculated from its alkalinity type. For the re-instated Loch Fitty, the reference (High status) lake diatom trophic index value would be 25.

Macrophytes: Loch Fitty is currently classified as having Good Status under the WFD for its macrophyte community according to the results of a survey undertaken in 2008. So, it has been assumed that the aquatic plant community in the restored Loch Fitty needs to be as good as, or better than, it was in 2008. So, preliminary targets for the restored Loch Fitty are:

Lake macrophyte nutrient index (LMNI)	≤ 6.71
Total species	≥ 23
Number of functional groups	≥ 13
Plant cover per species	≥ 1.82
Relative cover of filamentous algae	≤ 0.03%

A Microsoft Excel© based tool has been developed for assessing the ecological status of a loch based on its macrophytes by Nigel Willby, Stirling University, and this was used in the assessment by SEPA in 2009 (Ross Doughty, *pers. comm.*). An updated version of this tool may be available from either Nigel Willby or SEPA to calculate more accurate quality targets for future macrophyte communities in the restored Loch Fitty.

Macroinvertebrates: European targets for macroinvertebrates are under development, but not available yet. So, there are currently two WFD tools that can be used for assessing the ecological status of macroinvertebrates in lakes. These are the Lake Acidification Macroinvertebrate Metric (LAMM) and Chironomid Pupal Exuvial Technique (CPET). LAMM is for assessing the impacts of acidification, which is not a recognised pressure at Loch Fitty. So, for the purposes of this review, it is proposed to set targets for the new Loch Fitty using the CPET approach. This approach gives a reference value of -0.158, but there are no values available at present for the Good/Moderate, Moderate/Poor or Poor/Bad status class boundaries. So, all that can be said right now is that the CPET target for the new Loch Fitty needs to be close to -0.158.

Fish: There is no WFD compliant fish classification tool for standing waters at present, or formal sets of relevant targets. It has been suggested that only brown trout, eel, stone loach & 3-spined stickleback should be in the re-instated loch (meaning that minnow, perch, pike and roach need to be removed). Due to their negative impacts on water quality through zooplankton predation, the case for removing roach is overwhelming. However, the case for removing minnow, perch and pike is less clear, especially since the latter have been there for more than 200 years. The "brown trout, eel, stone loach & 3-spined stickleback" target seems to err on the side of caution in the context of WFD requirements.

Restoration/recolonisation strategy

Phosphorus: Meeting P concentration targets can be helped by encouraging colonisation, through replanting, of submerged and floating macrophytes as they reduce nutrient release from sediments and help establish a clear water state. Although non-invasive, native species could be added – such as white and yellow water lilies – it should be noted that accidental introductions of invasive, non-native species could degrade the WFD status of the loch in terms of its macrophyte community.

Phytoplankton/phytobenthos: Phytoplankton and phytobenthos species will recolonise the re-instated loch naturally from the inflows, *via* bird transfers and as inoculums associated with macrophytes replanting. This process requires no specific restoration/recolonisation strategy to be put in place.

Macrophytes: Macrophytes from the existing Loch Fitty will be transferred to special holding ponds and replanted into the re-instated Loch Fitty once it has been refilled. The use of this locally sourced material to replant the loch will maintain the integrity of the Loch Fitty aquatic plant populations. However, conditions in the holding ponds need to be specified more clearly, especially in relation to the availability of suitable depths (0.3-3m) for a range of macrophytes to ensure success. Sourcing low nutrient water to refill the ponds with may be a problem. Rainwater alone would not be suitable as it would have too low a pH value to suit plants from a high alkalinity lake, although this could be adjusted. It is essential that the health of the holding ponds is closely monitored, especially over the growing season, to avoid loss of important species or contamination by invasive species.

Macroinvertebrates: The restoration/recolonisation of macroinvertebrate species should focus on creating the right conditions (i.e. low nutrients, mixed substrates, native flora) within the loch. Then, the species required will recolonise naturally. For this reason, no formal recolonisation plans are needed. An important part of creating the right conditions is the improvement of the shoreline to provide more diverse habitats. If the right conditions are created within the loch, the feeder streams will provide a source of the required macoinvertebrate species if the quality of these streams has been improved before the loch is re-instated. In addition, many benthic invertebrate species are the juvenile stages of flying adults that can readily recolonise the loch by egg deposition if presented with the right conditions. Another vector of macroinvertebrates, especially crustacaea, is aquatic birds. Experience based on the development of reservoirs suggest that recolonisation by macroinvertebrates occurs naturally within a few years.

Fish: Restoration/recolonisation by fish in the new Loch Fitty is based on the deliberate introduction of desirable fish species and numbers by stocking. However, potential uncontrolled recolonisation by undesirable fish species, especially roach, from connected watercourses needs to be addressed because the eradication of unwelcome species is likely to be extremely difficult if they become established in the loch, unless drastic measures are taken. Manual capture by netting or electric fishing is highly unlikely to remove the entire populations and any remaining individuals are likely to re-establish abundant populations within a few years.

Monitoring and assessment

A range of monitoring and assessment procedures will need to be carried out over the recovery period to ensure that any developing problems are recognised early enough for remedial action to be taken. As a minimum, the following should be monitored:

Catchment nutrient input: Nutrient inputs (N, P, Si) from the feeder streams should be monitored at fortnightly intervals during the recovery period. In addition, targeted high flow event sampling should be undertaken to assess event driven inputs, such as sediment transport. Quarterly streamwalks along the inflows, at high and low flows, should be undertaken to identify any new nutrient source 'hotspots' within the catchment so that they can be addressed before they can affect the water quality of the loch.

P release from sediments: The amount of release sensitive sediment P and P released to the water column should be assessed over the summer period to underpin any follow up management action that is needed to control sediment P release during the recovery process.

Chlorophyll *a*, N, P and Si concentrations, phytoplankton and zooplankton abundances: Samples for the determination of in-lake chlorophyll *a*, N, P and Si concentrations, and phytoplankton and zooplankton species composition and abundance should be collected at monthly intervals over the recovery period. This will allow monitoring of progress towards WFD targets and provide information to underpin decisions on remedial action if problems develop.

Phytobenthos: Samples of benthic diatoms should be collected annually by brushing or scraping the upper surface of cobbles or small boulders obtained from the littoral zone of the loch, or from submerged stems of emergent macrophytes in areas dominated by fine sediments.

Macrophytes: Annual surveys of macrophytes species composition, percentage cover and growing depth should be undertaken over the recovery period. This will provide information on progress towards the WFD targets and early warning of problems that may require management intervention.

Macroinvertebrates: Surveys of the macroinvertebrate communities should be undertaken four times a year, between April and October. Sampling should include skimming the surface of the lake with a hand net.

Fish: Annual fish surveys should be undertaken over the recovery period.

Recovery times

In terms of sediment P concentrations, the old Loch Fitty would be expected to recover chemically from nutrient pollution, alone, in about 5 years if external inputs were drastically reduced. However, this is not the only problem that needs to be addressed at this site to achieve WFD Good Ecological Status. There are many hydromorphological pressures caused by the creation of artificial shoreline structures and the dumping/leaching of waste material into the loch. In particular, the latter has resulted in significant heavy metal pollution of the sediments. There is also a strong presence of invasive non-native fish species that also contribute to the downgrading of the ecological status of the waterbody. Draining the loch, removing the polluted sediments and non-native fish species, naturalising the shoreline and re-instating the loch is probably the only practical way for the loch to be restored to Good Ecological Status. The proposal by SCCL to incorporate this into their plans to extend surface mining within the catchment, if properly managed, provides a good opportunity for this to be achieved. Once re-instated, it is expected that ecological recovery to Good Ecological Status will take 5-10 years.

Management of the recovery process

There have been many attempts to restore the ecosystem function of lakes following degradation by some type of external pressure. However, there appear to be no examples of lakes being restored to particular water quality targets, as has been proposed for Loch Fitty. So, this is a challenge in terms of our knowledge and understanding of lake recovery processes. To reduce the risk of the desired endpoint not being reached, it is important that the recovery process is actively managed.

The recovery of the re-instated Loch Fitty may be adversely affected by invasive non-native species of fish, macroinvertebrates and plants. The risk of this happening needs to be kept to a minimum, because eradication is difficult or impossible once infestation has occurred. Measures to reduce the risk may include raising public awareness, controlling public access and ensuring holding ponds do not become infested, and it is possible that an invasive species action plan may need to be drawn up. Care should be taken not to introduce a seed bank of unwelcome species as part of the macrophyte rooting medium added to the bed of the new loch. In terms of establishing the required macrophyte community, it has been shown that pilot investigations into the recruitment/growth rates of the native species that are being re-introduced can be beneficial in terms of creating an evidence base upon which any necessary management plans or mitigation actions can be based.

As Loch Fitty is a calcareous, hardwater site, it is particularly vulnerable to invasion by species such as zebra mussels, signal crayfish and alien gammarids. Invasion by any of these species would reduce the WFD ecological status class of the reinstated loch. To avoid this problem, source materials used for rebuilding the loch need to be carefully controlled and any work/leisure craft used on the loch need to be decontaminated to prevent the spread of these animals. It is recommended that guidance documents recently published by the GB non-native species secretariat be consulted.

Invasive fish species are extremely difficult to eradicate and it is best to avoid them entering the re-instated loch in the first place. One of the key requirements of the restoration process is to remove roach from the loch, because these fish feed heavily on zooplankton and, in doing so, reduce the capacity of these crustacea to keep algal biomass under control. As a result, the water becomes increasingly turbid and high phytoplankton densities outcompete submerged macrophytes for available light. This can have a serious impact on the macrophyte community, preventing it meeting WFD targets.

Finally, it is evident from the in-lake data that are available for the existing Loch Fitty, that algal growth and biomass accumulation in the lake is limited by both N and P availability over the summer months. So, managing N and P sources within the catchment to reduce their inputs to the loch would benefit the restoration measures.

Case studies

A review of a number of case studies revealed the following general information in relation to the proposed restoration work:

- sediment removal has been widely used in restoration projects

- a response in P, chlorophyll *a* and, to a lesser extent, N concentrations were the most commonly reported in-lake responses

- no studies have been reported that have directly addressed WFD-compliance, or any other ecological recovery targets for the restored lakes

- recolonisation by macrophytes, phytoplankton and zooplankton has been reported for a small number of lakes after sediment removal

- ecological recovery is currently being studied in detail in a range of created lakes and practical management lessons may be learned from these

- a small number of lakes within the Czech Republic have been created on spent brown coal mining sites; the recolonisation of the fish community has been managed and monitored in these lakes

- there are number of sites to which visits could be made that would provide useful information on recovery processes.

In terms of practical experience, a number of specific recommendations have been made concerning further contacts and site visits.

Based on the results outlined above, it would appear to be sensible to wait until the macrophyte community has established and reached a steady state before introducing the fish. This would reduce the likelihood of negative fish impacts in the important early years of macrophyte colonisation and allow targeted contingency management to be put in place where necessary. Additionally, planting of desirable species will increase the resilience of the system to undesirable monocultures, although the end point community structure may not be solely dependent on the stocked plant community structure. Finally, as outlined already in this document, it is essential that non-native invasive species (especially macrophytes and fish), and perhaps also non-native waterfowl species, be excluded from the site during the early macrophyte colonisation phase where possible.

Conclusions

The proposed surface mining project provides a unique opportunity to restore Loch Fitty. It will address some key improvements that are essential for helping the loch meet WFD targets, i.e. good water quality and good hydromorphological conditions.

Good water quality can only be attained by addressing sources of nutrient pollution within the loch and its catchment, and by removing heavy metal laden loch bed sediments. This will ensure that previous water quality problems at the loch do not re-occur and that the improvement in status of the waterbody is sustainable over time.

Ensuring that invasive non-native species cannot return to the loch requires the ecological quality of the inflows and outflow to be improved. This is an additional benefit of the proposed loch restoration project.

Overall, the proposal of SCCL for restoring Loch Fitty has a better chance of delivering the required end result of Good Ecological Status than the River Basin Management Plan proposal, because failure to meet WFD ecological quality targets at this site cannot be resolved by catchment management, alone. Pressures due to hydromorphological modification, non-native species and the accumulation of heavy metals in the sediments also need to be addressed.

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1. Introduction

1.1 Background

In March 2011, the Scottish Coal Company Limited (SCCL; contact - Theo Philip) awarded the NERC Centre for Ecology and Hydrology (CEH) a contract to report on water quality and ecological targets for the restoration of Loch Fitty that were in line with the requirements of the European Water Framework Directive (WFD; European Parliament, 2000). CEH were also asked to discuss the likelihood of, and timescales for, the recovery, following the completion of a range of proposed restoration works. It has been proposed that these works will be undertaken both during (catchment management) and on completion (loch reinstatement) of surface mining works in the area. The aims and objectives of the project, outlined below, were compiled after an initial assessment of the available data and information on the loch, and following discussion with the Scottish Environment Protection Agency (SEPA), SCCL and Dundee University on 3rd February 2011.

Currently, Loch Fitty fails to meet the "lake-type specific" water quality standards that are specified for the site under the WFD, and it is classified as being of poor status in the Scotland River Basin Management Plan for the area (RBMP Water body information sheet for water body 100278 in Forth; 15/3/2010). It is widely believed that the recent 'poor' water quality conditions have resulted from elevated nutrient inputs to the loch from sources within the catchment in combination with the frequent re-suspension of the loch bed sediments (Jarvis & Quinn, 2010). Over the years, Loch Fitty has also undergone some significant hydromorphological changes, including hard engineering of the shoreline and re-alignment of feeder streams. These have resulted in historical changes to its shape, shoreline and hydrology (SCCL, 2010h) that are likely to have contributed to the current water quality problems (Rowan, 2010; Moss, 2008).

SCCL have published an Environmental Statement detailing their proposal to undertake development and restoration work at the existing St. Ninians Mine in Fife (SCCL, 2010a). These works include the excavation and surface mining of coal reserves that are situated beneath Loch Fitty and the subsequent restoration of the loch. In outline, it is proposed to combine the further development of the mine with the opportunity to improve the water quality and ecological status of Loch Fitty by:

(i) removing all nutrient and heavy metal laden sediments from the loch,

(ii) developing and implementing a sediment and nutrient management plan within the upstream catchment,

(iii) removing hard engineered shorelines and false partitions,

(iv) re-instating the loch with a greater variety of depth zones, and

(v) creating shorelines with undulations and long shallow zones, together with a number of islands.

The main steps involved in the restoration work are as follows:

(i) de-water Loch Fitty and provide temporary diversions of surface watercourses around the site (months 1-6),

(ii) create an ecological mitigation corridor to the south-east of the site to compensate for temporary loss of wetland habitat (months 1-6),

(iii) undertake coaling operations (months 1-72) to include:

a. temporary storage of overburden together with smaller linear storage areas for screening purposes,
b. temporary storage of topsoil and subsoil for re-use in restoration works,
c. provision of temporary flood storage and water treatment facilities,
d. restore Loch Fitty following completion of mining works,
e. restore Loch Fitty to WFD "good ecological status",
f. provide new habitats to help meet the targets of local and national Biodiversity Action Plan (LBAP & UKBAP),
g. reinstate existing land-use and landscapes around the loch, and
h. re-establish severed links and improve public access and recreational opportunities.

1.2 Main objectives of the CEH study

It is proposed that Loch Fitty will be drained and temporarily removed for a period of approximately 7.5 years, while the underlying coal is removed (SCCL, 2010h). Comprehensive planning and development documents have been prepared by SCCL (SCCL, 2010 a-h) and its development team in relation to this proposal, and a clear statement of intent in relation to recovery periods and end point water quality targets is now required. The main aim of the CEH study is to provide this information in the form of a report, written in association with SCCL and SEPA.

In relation to the above, the key objectives of this project are to:

(i) review all available data and information on Loch Fitty and its catchment that are relevant to the achievement of Good Ecological Status for Loch Fitty following the proposed development and restoration works,

(ii) use existing data and information to further characterise the current chemical and ecological status of Loch Fitty and clarify existing nutrient sources following the outcome of ongoing sediment analysis work (i.e. catchment *cf.* in-lake sediments), and, where possible, comment on nutrient transformation processes that should be taken into account during restoration works,

(iii) conduct a survey of the literature and of relevant experts across Europe to identify similar lake restoration/creation projects; highlight opportunities for site visits to gain firsthand experience of related projects,

(iv) develop type specific, and site specific where applicable, water quality targets for the re-instated Loch Fitty, where possible, that will ensure compliance with WFD requirements for Good Ecological Status,

(v) summarise the relevant literature and consult scientific experts to produce a recolonisation strategy and estimated timeline for the ecological recovery of the target organisms (taking cognisance of the work already undertaken in this regard, especially in relation to macrophytes and fish),

(vi) use conceptual models to create a list of recovery scenarios and an associated "tool-box" of management options that can be used to manage the recovery process, if necessary, to mitigate against a transition towards undesirable recovery endpoints, and

(vii) highlight any additional issues that may need to be addressed before, during and after re-instatement, to ensure that previous water quality problems at the loch do not re-occur and that the target improvement in status of the waterbody is sustainable over time.

This report summarises the WFD targets to be met by the re-instated Loch Fitty for it to attain WFD good ecological status and makes recommendations on a recolonisation strategy designed to achieve this. Additional mitigation/monitoring measures that may be required to attain and sustain good status, and a likely timescale for recovery, are also discussed.

2. The proposed Loch Fitty restoration project

2.1 Background

Loch Fitty is a shallow lowland loch in the River Leven catchment, near Dunfermline (**Figure 2.1**). The loch was surveyed by Farrer (1999). In this report, the loch is described as a large, mesotrophic loch used for recreational fishing that is subject to periodic algal blooms. The most recent such bloom was recorded here in 2010 (Ross Doughty, *pers. comm*.). Historical maps suggest that the shape and hydrology of the loch have been modified many times over the last few hundred years (SCCL, 2010h). Twenty five percent of the current shoreline of the loch is now hard engineered.

Figure 2.1 Site location (after SCCL, 2010a)

The loch, which is classified as a shallow, lowland, large, high alkalinity, lake under the WFD, has been surveyed by SEPA as part of the River Basin Management Planning (RBMP) process. The most recent RBMP classification of the site found the overall status of Loch Fitty to be ‘Poor’, with high confidence (RBMP Water body information sheet for water body 100278 in Forth; 15/3/2010). This is because, although the loch passed on its overall chemical water quality, its overall ecological status was found to be ‘Poor’ (with high confidence) due to high phosphorus and chlorophyll *a* concentrations in the water. Historical data supplied by the SEPA suggest that ecological water quality at the site has been steadily deteriorating over the last 10 years (Figures 2.2 & 2.3). This, combined with a parasitic infection within the fish community, led to the closure of the fishery in 2007.

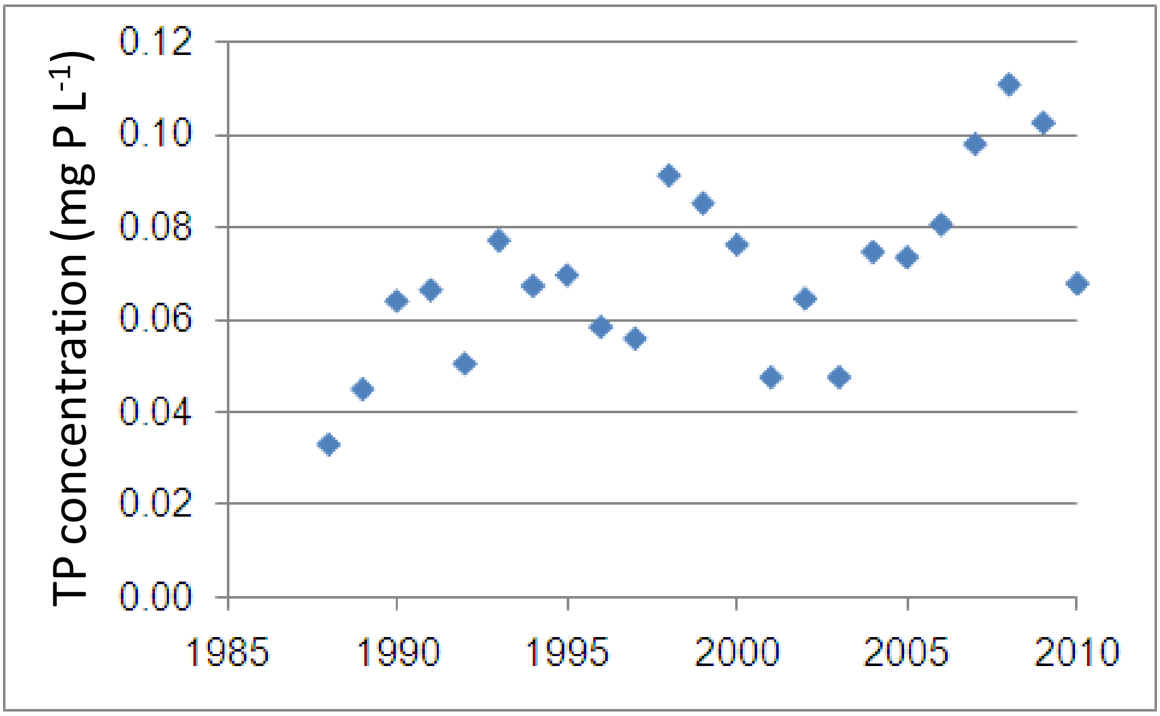

Figure 2.2 Open water total phosphorus (TP) concentrations in Loch Fitty, 1988-2010

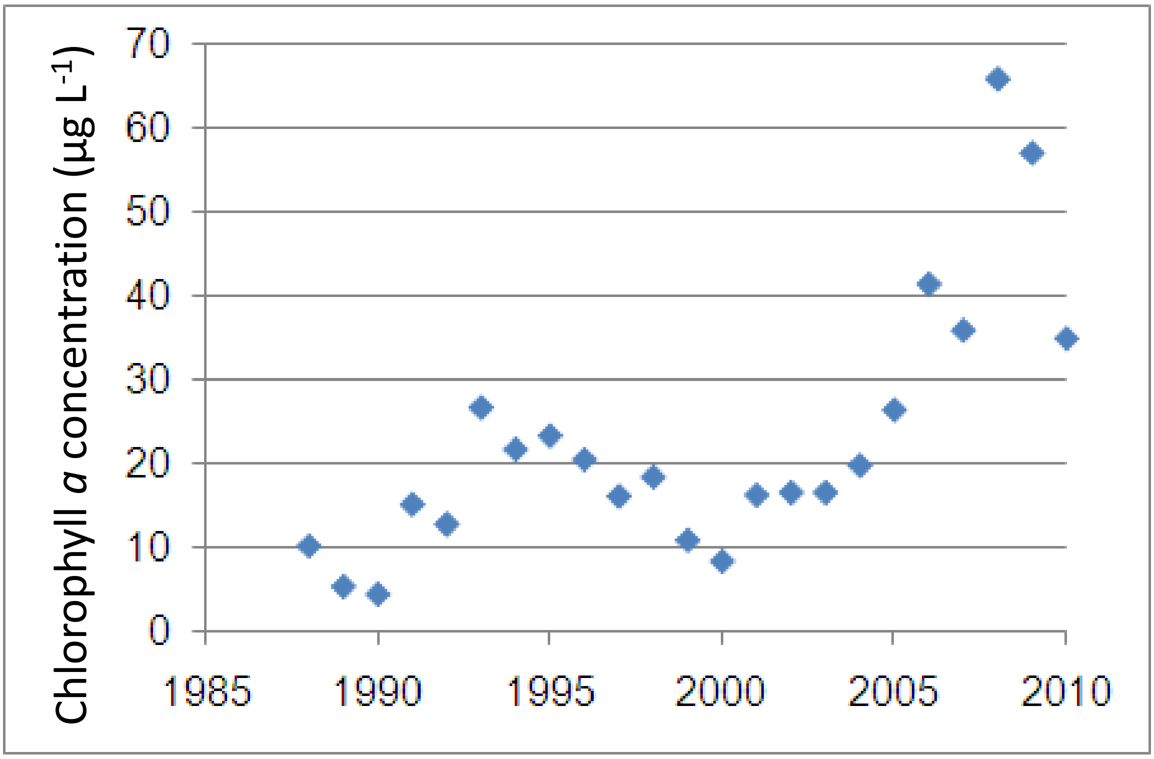

Figure 2.3 Open water chlorophyll*a* concentrations in Loch Fitty, 1988-2010.

To access 3.4 million tonnes of low sulphur, high quality coal reserves underneath the loch, SCCL are proposing to extend the current area of surface mining (Figure 2.4) by temporarily draining Loch Fitty and re-instating it when mining is complete. This would involve creating a water diversion channel by directly linking the current inflows to the current outflow and dewatering the loch. An ecological mitigation corridor would be created to minimise the impacts of these activities, and of the proposed surface mining work, on the local ecology. This will include translocation habitat for aquatic macrophytes and temporary fish ponds.

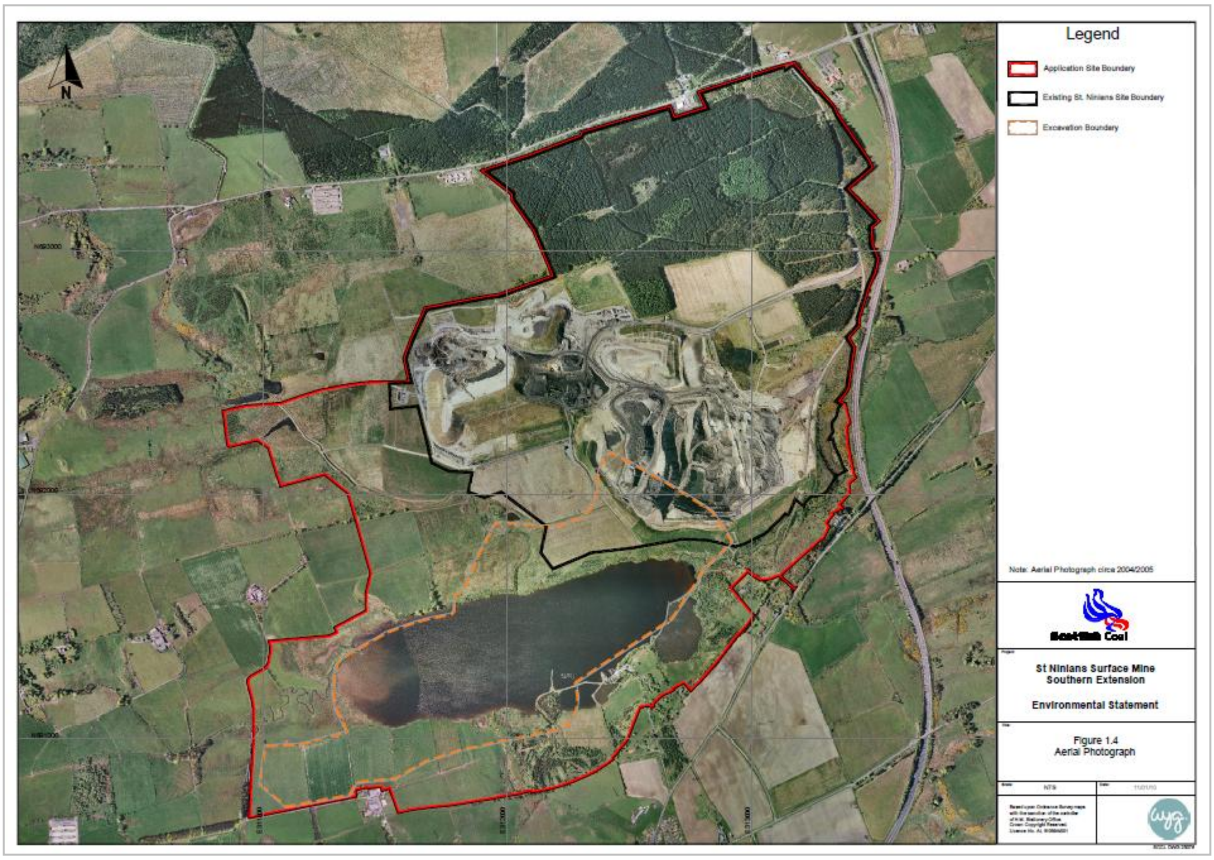

Figure 2.4 Current and proposed surface mining at Loch Fitty (after SCCL, 2010a).

The re-instated Loch Fitty will be a single standing waterbody of similar area to the current loch and with all of the main inflows and outflow re-instated to their original position, with the exception of the Lassodie Burn, which will be reinstated along its original course, away from Loch Fitty and a minor, unnamed watercourse which will be used to supply the ecological corridor. It is argued that the proposed project provides a unique opportunity to enhance/improve/restore Loch Fitty to meet WFD ecological quality targets.

2.2 The loch

When mining is complete, i.e. after about 6 years, a new Loch Fitty will be created over an 18 month period by backfilling the surface mining area and lining the new loch basin with a 2-4m thick layer of impermeable clay. The re-instated loch will then be refilled by pumping water into it from existing boreholes around the loch. The likely nutrient content of the water from these boreholes is unclear from the available documentation. Once full, the natural inlets and outlet will be reconnected to the loch and the Lassodie Burn, which currently flows into the loch (Figure 2.5), will bypass it as explained above.

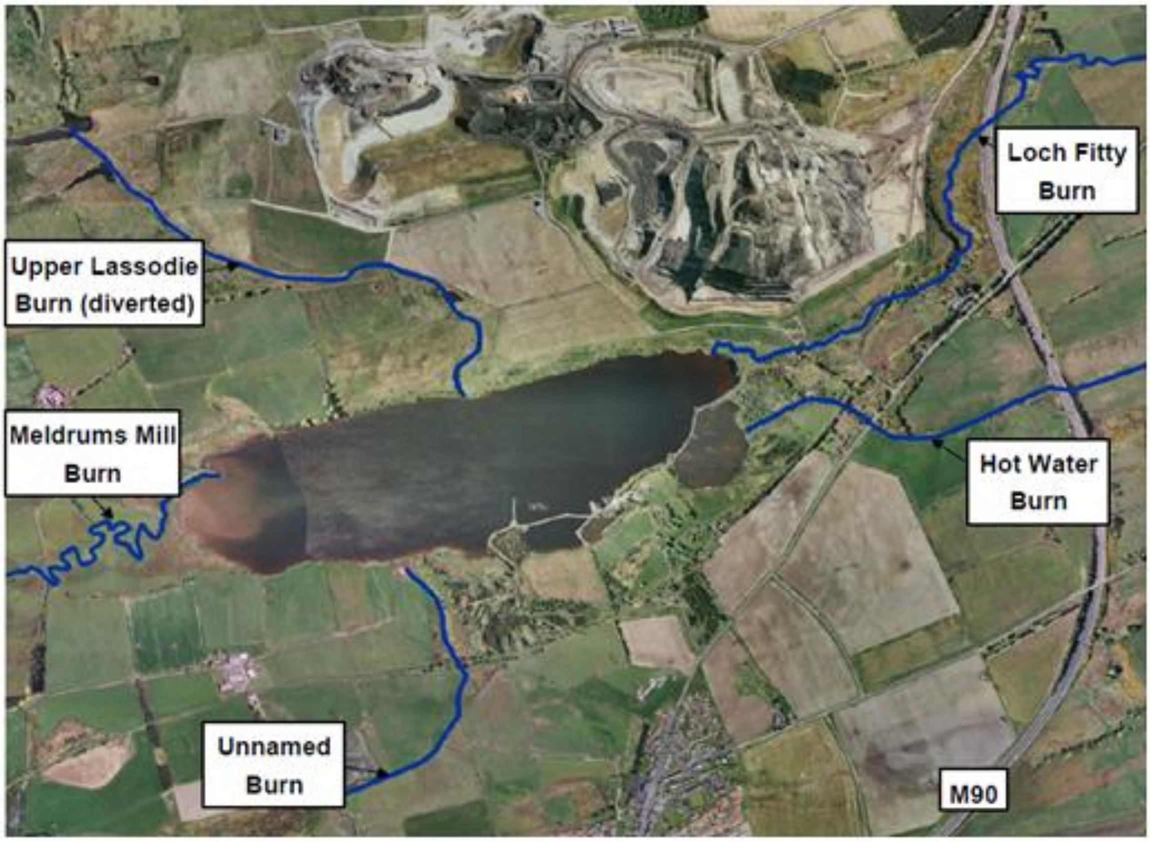

Figure 2.5 Inflows to, and outflow from, Loch Fitty before restoration (after SCCL, 2010h)

The bathymetry of the existing loch is shown in Figure 2.6. It is estimated that the volume of the reinstated loch will be about 1.69 x 10^6 m^3 when the water level is at 126.3 m.a.o.d. This is slightly greater than the corresponding value for the current loch, which is 1.54 x 10^6 m^3. The estimated water retention time of the new loch will be about 60 days (Theo Phillip, *pers. comm.*).

As it stands, the water level of the current loch is very responsive to changes in the rate of flow of its inflows, with the highest loch levels being recorded approximately 6.5h after peak flows are recorded in the main inflow stream, Meldrums Mill Burn. This results in the level of the existing loch varying by about 1m over the course of a year (Figure 2.7).

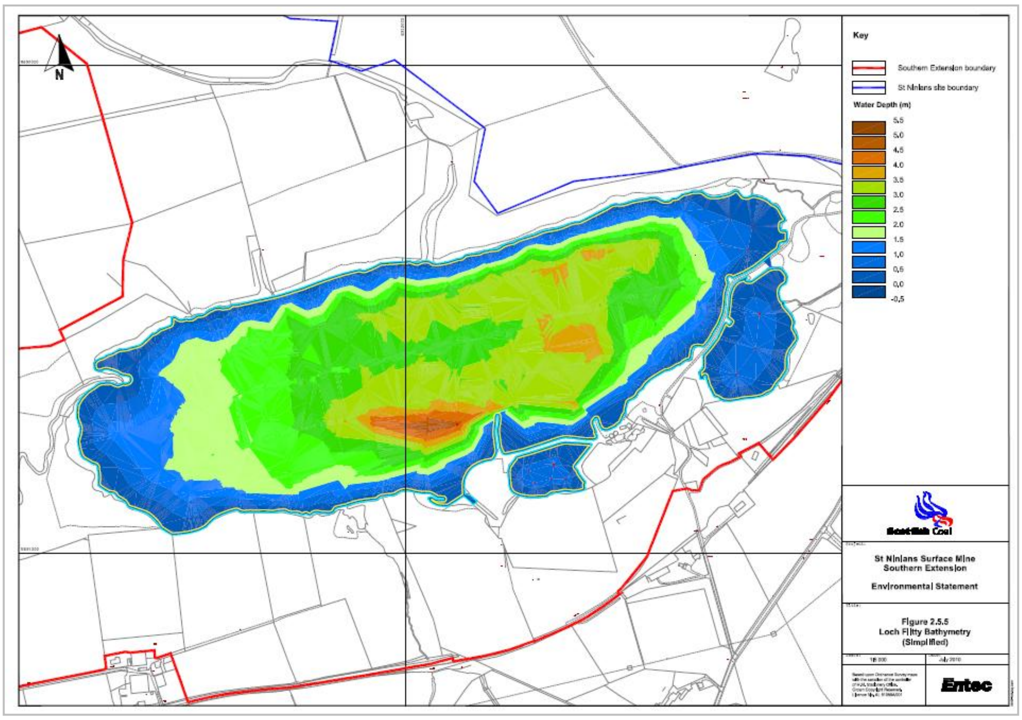

Figure 2.6 Simplified bathymetric map of Loch Fitty (after SCCL, 2010c).

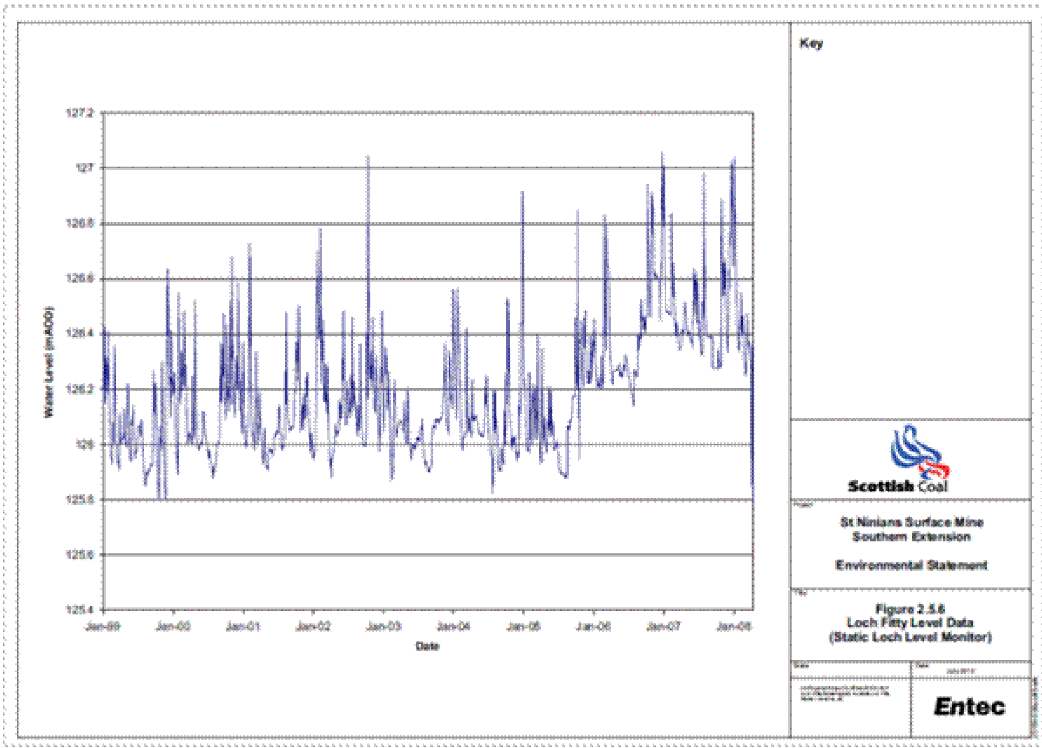

Figure 2.7 Loch Fitty water level, 1989 – 2008 (after SCCL, 2010c). *(NB values before Jan 2006 are considered to be unreliable.)*

2.3 The catchment

The surface water catchment of the new Loch Fitty will be similar to that of the old loch, prior to the realignment of the upper Lassodie Burn in 1998. The catchment will drain a total land area of 18.43 km^2 (Table 2.1), comprising sub-catchments drained by the individual feeder streams, as shown in Figure 2.8, apart from that of the Lassodie Burn. The loch, itself, discharges to the Lochfitty Burn.

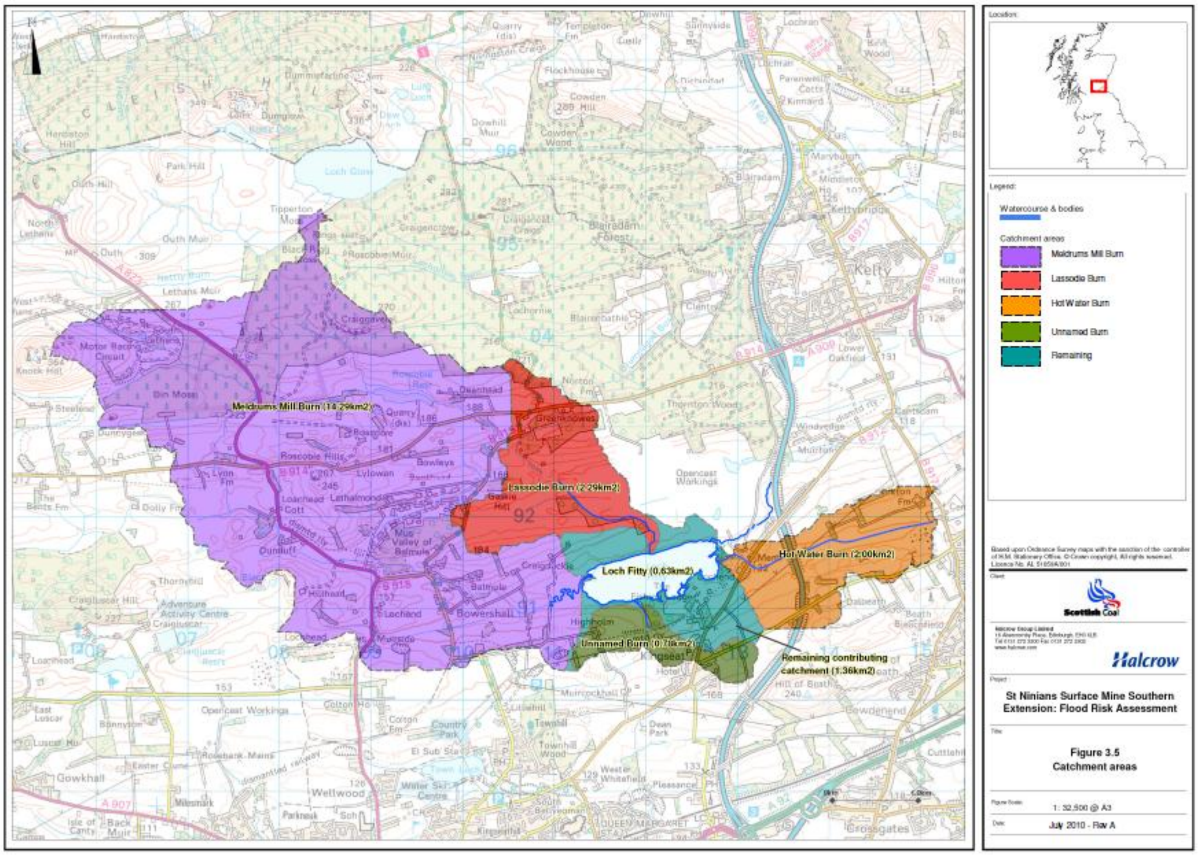

Figure 2.8 Catchment and subcatchments of Loch Fitty before restoration (after SCCL, 2010h).

The catchment is mainly agricultural in nature (arable and livestock grazing), and field boundaries and land use in this area seem to have changed little since the mid 1800s, apart from the development of some mining activity and the construction of the nearby motorway (M90). However, it should be noted that levels of fertiliser usage on farms within the catchment and, consequently, levels of nutrient laden runoff from these farms, is likely to have increased in recent years in line with national trends.

Many of the channels that drain into the old Loch Fitty were artificially straightened in the early 1800s (SCCL, 2010h). As a result, their banks have become unstable and large quantities of fine sediment have been delivered to the old loch from this source. There may also be other sources of nutrients within the catchment, such as septic tanks and slurry pits. The problem of elevated nutrient inputs to the loch from these sources need to be solved before the new loch is re-instated.

2.4 The ecology and water quality

The overall objective for the restoration of Loch Fitty is to create a loch of good ecological status that is in accordance with the aims of the WFD and the Scotland River Basin Management Plan. In relation to this, the two most important elements in establishing a high quality loch habitat and good ecological conditions for the longer term are:

- good water quality
- good hydromorphological conditions.

A key aim for the restored loch is for it to mimic the 'natural' physical conditions of the old loch. This includes the following improvements to its current state:

- more natural shoreline,
- varied bathymetry,
- islands to encourage otters,
- beach areas for wading birds, and
- suitable underwater habitat for aquatic plants.

The extent and location of the features listed above are summarised in Figure 2.9.

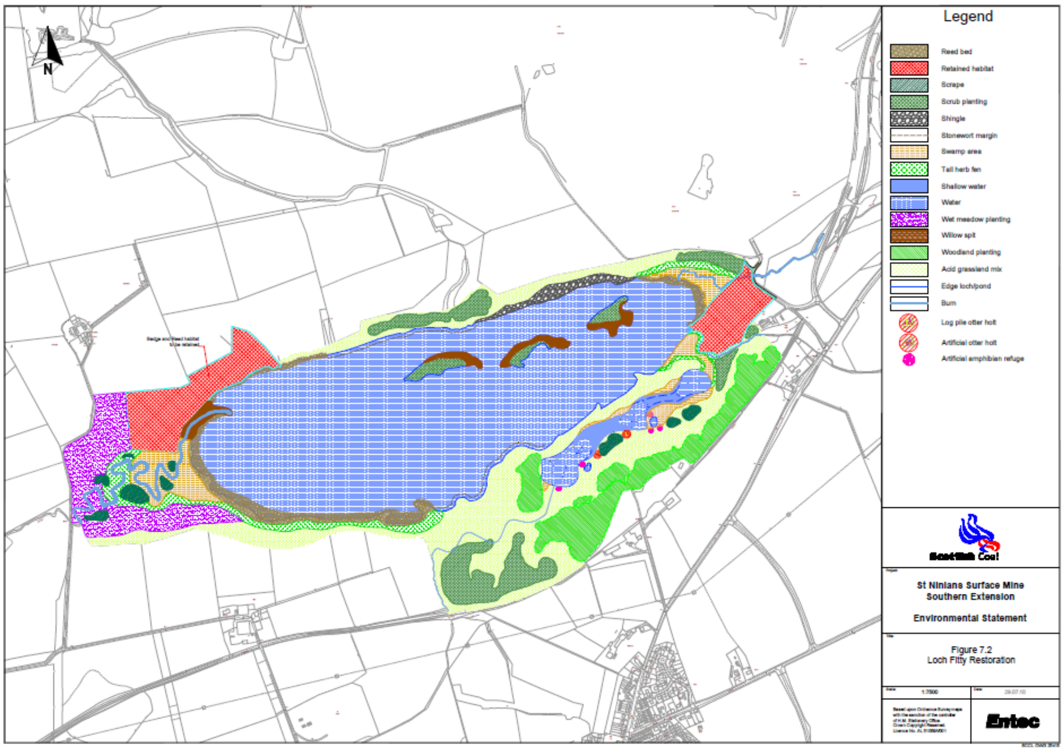

Figure 2.9 Habitat features of the restored Loch Fitty (after SCCL, 2010c).

The proposed design for the reinstated loch closely resembles that of the existing water body in terms of its geographical location, size and volume, but with a less modified and more natural shoreline. The main aim of the new design is to ensure the re-colonisation of the loch by existing macrophyte assemblages. As such, the following features will be included:

- a varied loch profile with shallow margins and a wide drawdown zone, underwater bars and shoals (undulations), and deep areas,
- a depth range of a few centimetres to 5.4m,
- substrates of various grain sizes,
- improved shoreline habitat, including the removal of concrete and rip rap shorelines and partitions,
- a gravel/pebble ‘beach’ along part of the northern loch margin to provide foraging habitat for wading wetland birds and waterfowl,
- a diverse range of habitats within the littoral and marginal zones, and
- good conditions for the translocation and recolonisation of the reinstated loch by valuable aquatic plant species that have been identified in the existing loch.

Key ecological services provided by the old Loch Fitty include:

- flood attenuation,
- nutrient management/retention, and
- sediment sink.

The new loch and has been designed to perform these services. A comparison of the key features of the old and restored Loch Fitty is shown in **Table 2.1**.

Table 2.1 Comparison of existing and restored Loch Fitty (after SCCL 2010h)

Parameter	Current status	Proposed future status
Surface area (km^2)	0.63	0.59
Maximum depth (m)	5.0	5.4
Mean depth (m)	2.3	3.6
Volume ($x\ 10^6\ m^3$)	1.54	1.69
Catchment area (km^2)		
Meldrums Mill Burn	14.29	14.29
Lassodie Burn	2.29	0.00
Unnamed Burn	0.78	0.78
Hot Water Burn	2.00	2.00
Remaining catchment	1.36	1.36
Total	20.72	18.43
Shoreline (km)	3.90	4.10 (+1.5 islands)
Hard engineered shoreline (km)	0.99	0.00
% littoral zone	14	29
Water level (m AOD)	126.10	126.10
WFD lake type	HAVS	HAS

An ecological “Clerk of Works” (CoW) will co-ordinate and manage the ecological mitigation strategy and the ecological recovery of the loch. The CoW will report to a Management Committee comprising representatives of SCCL, Fife Council, Scottish Environment Protection Agency (SEPA), Scottish Natural Heritage (SNH), and the Forth District Salmon Fishery Board (FDSFB).

3. WFD good ecological status target for re-instated Loch Fitty: phosphorus

Total phosphorus (TP) is an important "supporting" physico-chemical parameter for WFD classifications. It is a widely used measure of the nutrient enrichment status of a lake and often appears to be the main limiting nutrient of phytoplankton production in many lowland temperate systems (e.g. Dillon & Rigler, 1974; Schindler, 1977; OECD, 1982; Phillips *et al.*, 2008).

3.1 Compilation of available data and information

Data provided by SEPA indicate that annual mean TP concentrations in Loch Fitty have been steadily increasing over the past 20 years, from an annual mean value of 33 µg L^{-1} in 1988 to a peak of 111 µg L^{-1} in 2008 (Figure 2.2). Following the widely used OECD (1982) classification scheme, this represents a shift from a moderately enriched, mesotrophic state to an enriched hyper-trophic state.

Figure 3.1 comapres the seasonality of TP concentrations for the first (1988-1997) and second (2000-2009) 10 year monitoring periods. This reveals that seasonal changes in TP concentrations were more pronounced during the earlier period, showing a marked decline during spring and summer months. In contrast, this decline is less pronounced during 2000-2009, with values remaining relatively high throughout the year apart from October.

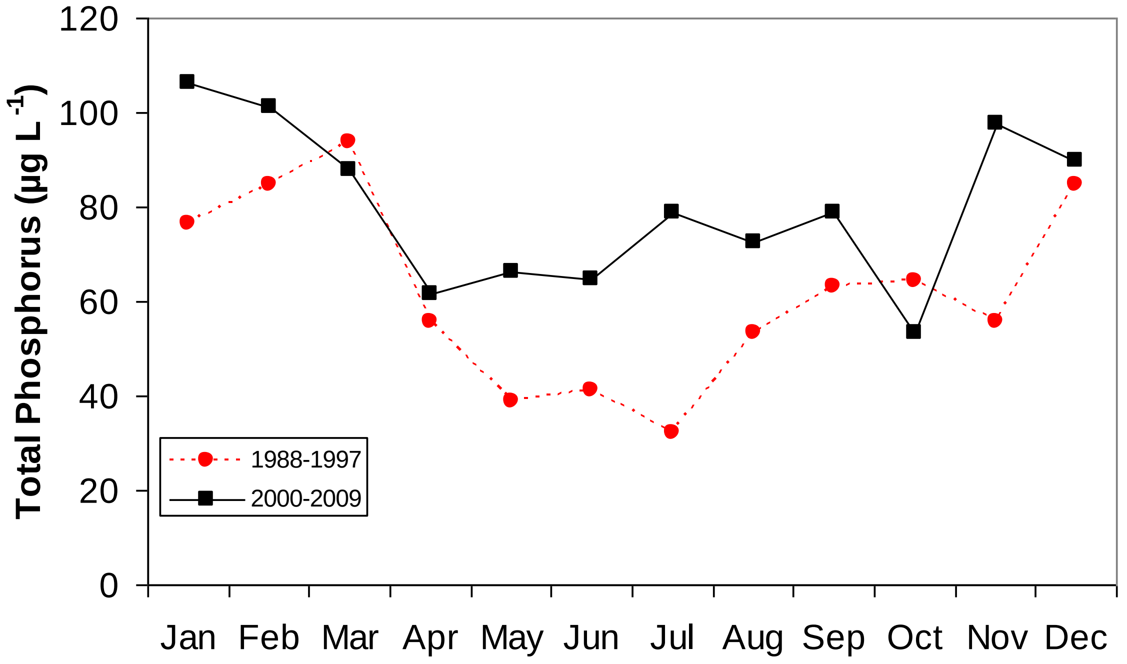

Figure 3.1 Seasonality in TP concentrations in Loch Fitty, 1988-1997 and 2000-2009

3.2 Current ecological status

The current typology of Loch Fitty is a very shallow (mean depth 2.3 m), high alkalinity loch. UK Environmental Standards (WFD UKTAG, 2008b) set an annual mean TP standard for this loch type of 32 µg TP L^{-1} in relation to the Good/Moderate boundary for lakes in northern parts of the UK. Phosphorus concentrations in Loch Fitty are consistently above this value throughout the year and the average annual mean TP concentration the site has been 90 µg L^{-1} for the past 6 years (2004-2009). These values are also in excess of the moderate/poor boundary of 64 µg TP L^{-1}. This places the loch well within the class of poor status.

WFD-UKTAG (2008b) recommends adopting site-specific TP standards for lakes, where possible, as these take account of the individual characteristics of each site, especially in relation to alkalinity and mean depth. This is based on the Morpho-Edaphic Index (MEI) modelling approach outlined by Cardoso *et al*. (2007). From this reference value, factors are used to identify TP status class boundaries representing increasing degrees of change from reference conditions. These factors are ecology-based and include the relationship between phytoplankton chlorophyll *a* and TP concentrations, the growing depth of aquatic macrophytes and evidence from palaeolimnological investigations on diatom community responses to TP (WFD-UKTAG, 2008b). Based on the equation in Cardoso *et al*. (2007), the site-specific TP standards for the current Loch Fitty are 25 µg L^{-1} for the Good/Moderate boundary, 49 µg L^{-1} for the Moderate/Poor boundary and 99 µg L^{-1} for the Poor/Bad boundary. Using these site-specific standards, Loch Fitty, with an average annual mean TP concentration of 90 µg L^{-1} for the past 6 years (2004-2009), would be classified as having Poor status.

3.3 Targets for the re-instated Loch Fitty compliant with WFD requirements

The future typology of Loch Fitty is that of a shallow (mean depth 3.6 m), high alkalinity loch. Assuming that the alkalinity class remains the same as before, the increasing depth of the loch means that it will have more challenging type-specific TP standards than the original loch. WFD-UKTAG (2008b) guidance sets a TP standard for the proposed loch type of 23 µg L^{-1} for the Good/Moderate boundary for lakes in northern parts of the UK, and 46 µg L^{-1} for the Moderate/Poor boundary.

With the increase in mean depth planned for the new loch (i.e. from 2.3 m to 3.6 m), the site-specific TP standards for the future Loch Fitty, based on the equations given by Cardoso *et al*. (2007) are 22 µg L^{-1} for the Good/Moderate boundary, 45 µg L^{-1} for the Moderate/Poor boundary and 89 µg L^{-1} for the Poor/Bad boundary.

4. WFD good ecological status target for re-instated Loch Fitty: nitrogen

Nitrogen (N) has been shown to be the primary or co-limiting nutrient for phytoplankton production in many lakes within the UK (Moss *et al.*, 1992, 1994; James *et al.*,. 2003; Maberly et al., 2003; May et al., 2010;), Europe (Sommer 1993; van der Molen *et al*., 1998) and elsewhere (Elser Marzolf & Goldman 1990). Nitrogen-limitation of phytoplankton production may be particularly significant in lochs that have a plentiful supply of phosphorus, especially during summer periods when denitrification rates may peak, converting nitrate (NO_3-N) to gaseous forms of nitrogen (N_2O and N_2) that escape to the atmosphere.

Nitrogen concentrations have also been linked to the diversity of the aquatic plant community. James *et al*. (2005), for example, established a significant correlation between declining macrophyte species richness and increasing winter concentrations of nitrate and total nitrogen in UK and Polish lakes. The authors suggested that increasing nitrogen concentrations may promote the rapid growth of a few competitive species resulting in the loss of slower-growing species. While this theory is largely speculative, it has received some support from Barker *et al*. (2008) who found a greater reduction in macrophyte species richness in mesocosm experiments where the concentration of nitrate-nitrogen exceeded about 1.5 mg L^{-1}.

4.1 Compilation of available data and information

Data provided by SEPA indicated that annual mean nitrate-nitrogen concentrations in Loch Fitty have been steadily declining over the past 5 years (2005-2009), i.e. from 0.69 mg L^{-1} in 2005 to 0.32 mg L^{-1} in 2009. Concentrations were frequently below the detection limit of the analytical technique (0.15 mg L^{-1}) from May to August. In analyzing these data, a value of half of the detection limit has been assumed when nutrient concentrations were reported to be below these levels (i.e. 0.15 mg L^{-1} for NO_3-N and 0.004 mg L^{-1} for PO_4-P).

The average (2005-2010) monthly concentrations of NO_3-N were plotted and compared with values for PO_4-P to examine potential for nutrient limitation of phytoplankton (Figure 4.1). PO_4-P availability fell to concentrations that could potentially limit phytoplankton growth from April to June and concentrations remained very low throughout the summer, rising again in November. Availability of NO_3-N could potentially limit phytoplankton growth from July to October, but in 2008 and 2009 concentrations have also been below detection limits in May and June. This analysis suggests that both PO_4-P and NO_3-N concentrations reach potentially limiting levels for phytoplankton growth during summer in the existing Loch Fitty.

Nitrate concentrations are consistently below the levels suggested to cause significant declines in aquatic macrophyte communities (James *et al*., 2005; Barker *et al*., 2008). This, alongside the removal of nutrient-rich sediments, should improve the chances of success of re-establishing less competitive plants, such as stoneworts, during restoration.

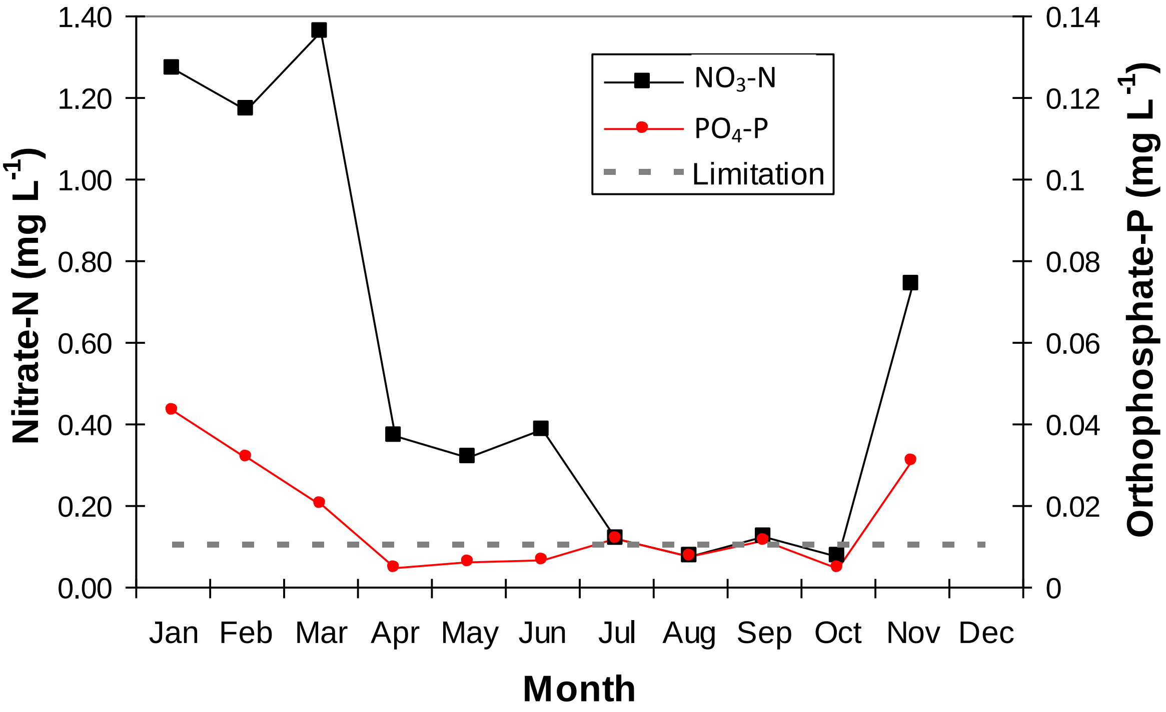

Figure 4.1 Seasonality in nitrate-N (NO_3-N) and orthophosphate-P (PO_4-P) concentrations in Loch Fitty, 2005-2010.

4.2 Current ecological status

There are currently no UK environmental standards for N for freshwaters within the UK in relation to the WFD. In a recent report to Natural England, Maberly & Carvalho (2010) have suggested N standards that could be adopted for shallow lakes as supporting standards in relation to achieving WFD chlorophyll *a* standards. In their analysis, WFD chlorophyll *a* targets were used as an ecological target for setting N targets. This was carried out by inverting European regression equations relating total nitrogen (TN) concentrations to chlorophyll *a* (Phillips *et al*., 2008) (Table 4.1).

Table 4.1: Regression equations relating chlorophyll *a* to total nitrogen (TN) concentrations (µg L^{-1}) for lake types relevant to Loch Fitty's current and future typology (Modified from Phillips *et al*., 2008).

Lake Type	Equations
HAS (future)	Log_{10} (Chl) = -2.177(±0.35) + 1.096(±0.12) Log_{10} (TN)
HAVS (current)	Log_{10} (Chl) = -2.575(±0.87) + 1.205(±0.30) Log_{10} (TN)

Currently the WFD chlorophyll *a* target for the Good/Moderate boundary for the loch is set at an equivalent annual mean of 16.5 µg L^{-1} and the future loch would have an equivalent annual mean standard of 7.5 µg L^{-1} (Carvalho *et al*., 2006: Table 7.1). On the basis of the equations given in Table 4.1, this gives indicative annual mean Good/Moderate targets of 1.4 mg L^{-1} and 0.6 mg L^{-1} TN for the current loch (HAVS) and future loch (HAS), respectively. Insufficient monitoring data were available with which existing annual mean TN concentrations could be assessed against these potential targets. However, as peak winter concentrations of NO_3-N are, on average, below the annual average target value of 1.4 mg L^{-1}, it may be expected that annual mean TN concentrations are also below this target.

4.3 Targets for the re-instated Loch Fitty compliant with WFD requirements

As described in Section 4.2, the indicative TN target for the new Loch Fitty to achieve Good Status, based on currently available information, is 0.6 mg L^{-1}. It should be stressed, however, that the proposed deepening of the loch will result in a change in the lake type category from high alkalinity, very shallow (HAVS) to high alkalinity, shallow (HAS). As such, the unconfirmed TN target for the new Loch Fitty is much lower than that proposed for the current Loch Fitty. It should also be noted that, in its current state, the available data on N suggest that the loch is comfortably within the Good Status category at the moment. However, when compared to the lower target for the re-instated loch, the current N concentrations appear to be borderline.

5. WFD good ecological status target for re-instated Loch Fitty: phytoplankton & phytobenthos

Algae (phytoplankton and phytobenthos) are key primary producers in lake ecosystems. They are a major component of the base of the food chain and represent an important source of food for zooplankton and benthic invertebrates. Consequently, they are important for sustaining higher life forms, such as fish, otters and waterfowl. Phytoplankton abundance within the water column is widely estimated using a proxy, i.e. chlorophyll *a* concentrations. However, their abundance can also be measured in terms of the volume of algae present in a water sample (biovolume).

Cyanobacteria are a particular type of algae that naturally occur in freshwaters and play a key role in nutrient cycling; however, they can also present hazards to the health of humans and other animals if large populations flourish and produce blooms. This is especially problematic when these blooms accumulate as scums on lake surfaces or along shorelines, because they frequently produce potent toxins that can result in adverse health effects ranging from mild (e.g. skin irritations and gastrointestinal upsets) to fatal (Codd *et al*., 1999; Codd *et al*., 2005).

Phytoplankton also interact with macrophytes, mainly through competition for light, and there is a tendency for very shallow lochs to exist in either a turbid phytoplankton-dominated state or a clear-water macrophyte-dominated state. For the reasons outlined above, increasing abundance of phytoplankton, and of cyanobacteria in particular, are considered to have a negative impact on the potential recreational and amenity value of lochs.

The phytobenthos community includes algae that are attached to stones and plants, and those living on the sediments and sands. Diatoms are often the dominant form of phytobenthos and species within this group of algae are widely used as indicators of the impact of nutrient pollution (Kelly *et al*., 2007; WFD-UKTAG, 2008a).

The WFD requires the ecological status of water bodies to be assessed on the condition of their biological quality elements (European Parliament, 2000: Article 8, Annex V). For lakes this includes phytoplankton and phytobenthos.

The WFD outlines three features of the phytoplankton quality element that need to be taken into account in this assessment. These are:

1. composition,
2. abundance, and its effect on water transparency
3. bloom frequency and intensity.

Classifications for phytoplankton abundance and its effect on transparency conditions have been established for this purpose (Carvalho *et al*., 2006; 2008; 2009) and UK and European standards have been adopted (see Table 7.1 in Carvalho *et al*., 2006; Poikane *et al*., 2010). A preliminary classification scheme for phytoplankton community composition has also been proposed (Carvalho *et al*., 2007) and a preliminary metric has been developed based on the percentage of cyanobacteria present. More recently, these metrics have been refined and new classification schemes for species composition and phytoplankton blooms are currently under consideration at both UK and European levels (Phillips *et al*., 2010; Mischke *et al*., 2010).

The phytobenthos community is considered to be a quality element, together with macrophytes, in Annex V of the WFD. It is intended that the composition and abundance of the community should be used to assess the status of lakes. More specifically, it is stated that the phytobenthic community should not be "adversely affected by bacterial tufts and coats present due to anthropogenic activity". A classification scheme for phytobenthos composition has been established for UK lakes based on benthic diatom species (Kelly *et al*., 2007; WFD-UKTAG, 2008).

5.1 Compilation of available data and information

Data provided by SEPA indicate that annual mean phytoplankton chlorophyll *a* concentrations have increased in Loch Fitty over the past 20 years, and especially rapidly since 2005 (Figure 2.3). Annual mean concentrations have increased from 4 µg L^{-1} in 1988 to a peak of 66 µg L^{-1}, which was recorded in 2008 (Figure 2.3).

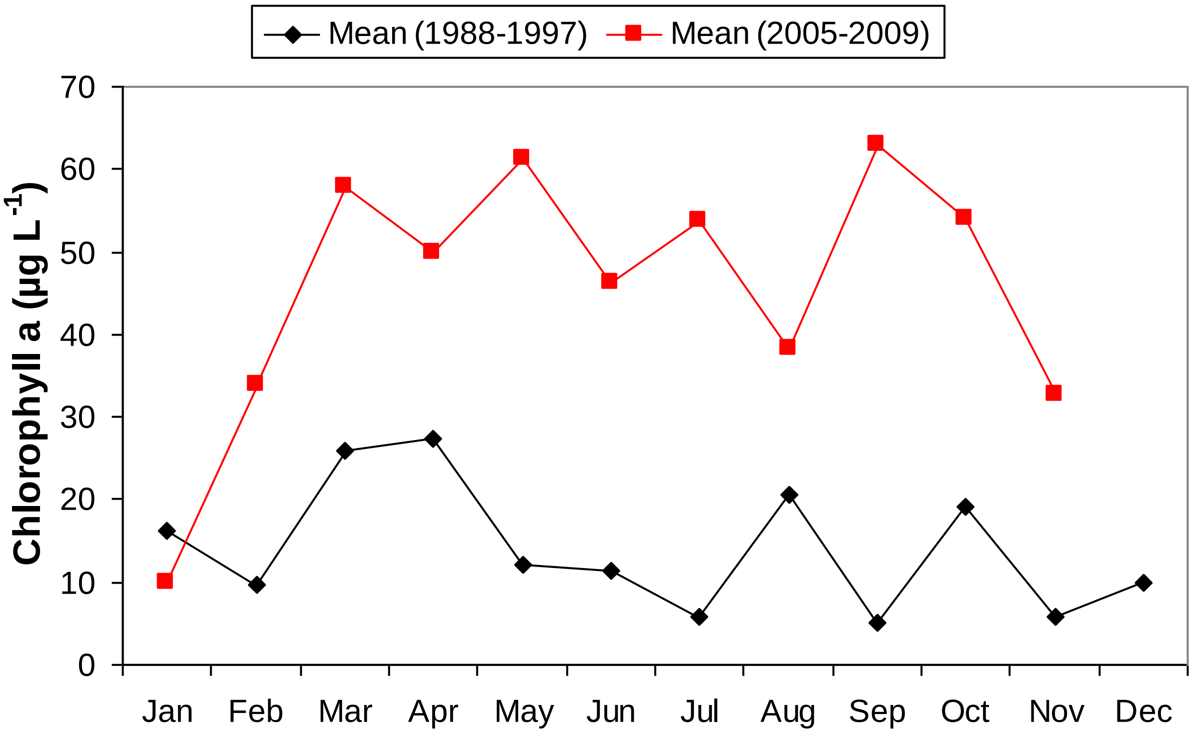

Figure 5.1 Seasonality in chlorophyll *a* concentrations in Loch Fitty, 1988-1997 and 2000-2009.

Figure 5.1 compares the seasonality of chlorophyll *a* concentrations in the loch for the first 10 years (1988-1997) and the last 5 years (2005-2009) of monitoring. This reveals that chlorophyll *a* concentrations are now much higher than before throughout most of the year. In addition, seasonality is more pronounced during 1988-1997 than 2005-2009. The earlier period is characterised by a spring maximum followed by a decrease in chlorophyll *a* concentration that are characteristic of the spring 'clear water' phase. This phenomenon is observed in many productive lakes, and is generally driven by nutrient limitation and zooplankton grazing pressures. It is unclear why the spring clear water phase is absent from during later period, but this may reflect the reduction or loss of the zooplankton community. This can occur due to a range of pressures, most notably increased predation from planktivorous fish (e.g. roach, *Rutilus rutilus*) or in response to pollution events (May & Spears, submitted).

Monthly phytoplankton composition data were available for the months July to September in 2008 (Table 5.1). These showed that, over the summer, cyanobacteria (Cyanonophyceae) were the dominant algal class, especially in July. Diatoms (Baccillariophycae) were also very abundant and these were the dominant class in September. The World Health Organisation (WHO) guidelines for safe recreational waters (WHO, 2003) outline three health risk categories for cyanobacteria: low, medium and high. High risk is assigned when surface scums are present, as cell densities and toxin concentrations tend to be very high under these conditions. Medium and low risk waters are those where cyanobacteria cells are at, or above, 100,000 and 20,000 cells ml^{-1}, respectively. These thresholds can be converted to a biovolume ($mm^3\ L^{-1}$) by multiplying by a typical cyanobacterial cell volume. Based on a typical cyanobacterium from this lake type (*Microcystis aeruginosa*) having a cell diameter of 4.5 µm, the recalculated WHO thresholds are equivalent to about 2 $mm^3\ L^{-1}$ for the low risk threshold and 10 $mm^3\ L^{-1}$ for the medium risk threshold. It is clear from these results that cyanobacteria were above the low risk threshold for 2 of the 3 months monitored and above the medium risk threshold in July. This indicates that, in 2008 at least, the loch should not have been used for recreational purposes and that warning signs of health hazards to the public should have been in place over that summer of 2008. Indeed, the loch had been closed to the public, due to concerns about water quality, since 2007.

Table 5.1 Phytoplankton data from Loch Fitty in 2008, summarised by algal class. Values are expressed as biovolume ($mm^3\ L^{-1}$) in columns 3-5 and as % relative abundance in column 6.

Algal Class	July	August	September	Average	% Average
Bacillariophyceae	0.052	0.330	10.317	3.566	28%
Chlorophyceae	5.050	0.638	1.174	2.287	18%
Chrysophyceae	0.197	0.190	0.255	0.214	2%
Cryptophyceae	0.454	0.344	0.963	0.587	5%
Cyanophyceae	10.065	0.050	6.203	5.439	43%
Dinophyceae		0.005	0.159	0.082	1%
Euglenophyceae	0.318		0.308	0.313	2%
Klebsormidiophyceae	0.020			0.020	0%
Unknown	0.506	0.057	0.021	0.195	2%
Total Biovolume	**16.661**	**1.613**	**19.401**	**12.704**	**100%**

Phytoplankton samples collected from Loch Fitty July, August and September 2010 for WFD purposes were found to be densely packed with phytoplankton taxa (Pauline Lang, *pers. comm*.). Samples were analysed according to standard SEPA procedures using an inverted microscope. At low or medium magnification (i.e. x10, x20 or x40), *Pandorina morum* (colonies), *Gymnodinium helveticum* (cells), *Pediastrum duplex* (colonies), *Snowella* sp. (colonies), *Anabaena* spp. (cells), and *Aulacoseira* spp. (cells) were relatively abundant. At high magnification (i.e. x63) the most common species were *Monoraphidium minutum* (cells), *Monoraphidium contortum* (cells), *Lagerheimia genevensis* (cells), *Tetrastrum triangulare* (cells) and *Tetrastrum staurogeniaeforme* (cells). These morphologically small taxa were a particularly abundant component of the samples, suggesting a low level of zooplankton grazing on the phytoplankton community. A number of other taxa (not listed) were also found to be present in the loch, but these were generally of lesser abundance than the taxa listed above.

Recent analysis for the development of WFD cyanobacterial bloom metrics (Mischke *et al*., 2010) show that, at a TP concentration of 19 µg L^{-1}, there is a 90% likelihood of a lake being below the WHO low risk threshold for cyanobacteria. At 30 µg L^{-1}, this risk reduces to 75% likelihood, and at 78 µg L^{-1} it reduces to only a 50%. These calculations provide some helpful TP targets that could be used for the management of cyanobacterial blooms at Loch Fitty.

No data on phytobenthos from Loch Fitty were accessible to the authors. Information may be available from Martyn Kelly at Bowburn Consultancy, who was the lead author of a report outlining the development of phytobenthos-based classifications for UK freshwaters (Kelly *et al*., 2007).

5.2 Current ecological status

The chlorophyll *a* standards for the current and future Loch Fitty are given in Table 5.2. These relate to ‘growing season’ means (i.e. April to September). Currently the loch is classified as a high alkalinity, very shallow loch, and the Good/Moderate boundary set for the growing season mean is 22.9 µg L^{-1} (Table 5.2). This is equivalent to an annual mean value of 16.5 µg L^{-1} (Carvalho *et al*., 2006: Table 7.1).

Table 5.2 WFD chlorophyll *a* standard concentrations (µg L^{-1}) for status class boundaries in high alkalinity shallow and very shallow lakes; H/G = High/Good, G/M = Good/Moderate, M/P = Moderate/Poor, P/B = Poor/Bad. Values are growing season means (i.e. for April-September).

Status class boundary	H/G	G/M	M/P	P/B
High alkalinity, shallow loch (HAS)	5.8	13.3	30.4	70
High alkalinity, very shallow loch (HAVS)	10.8	22.9	48.6	103

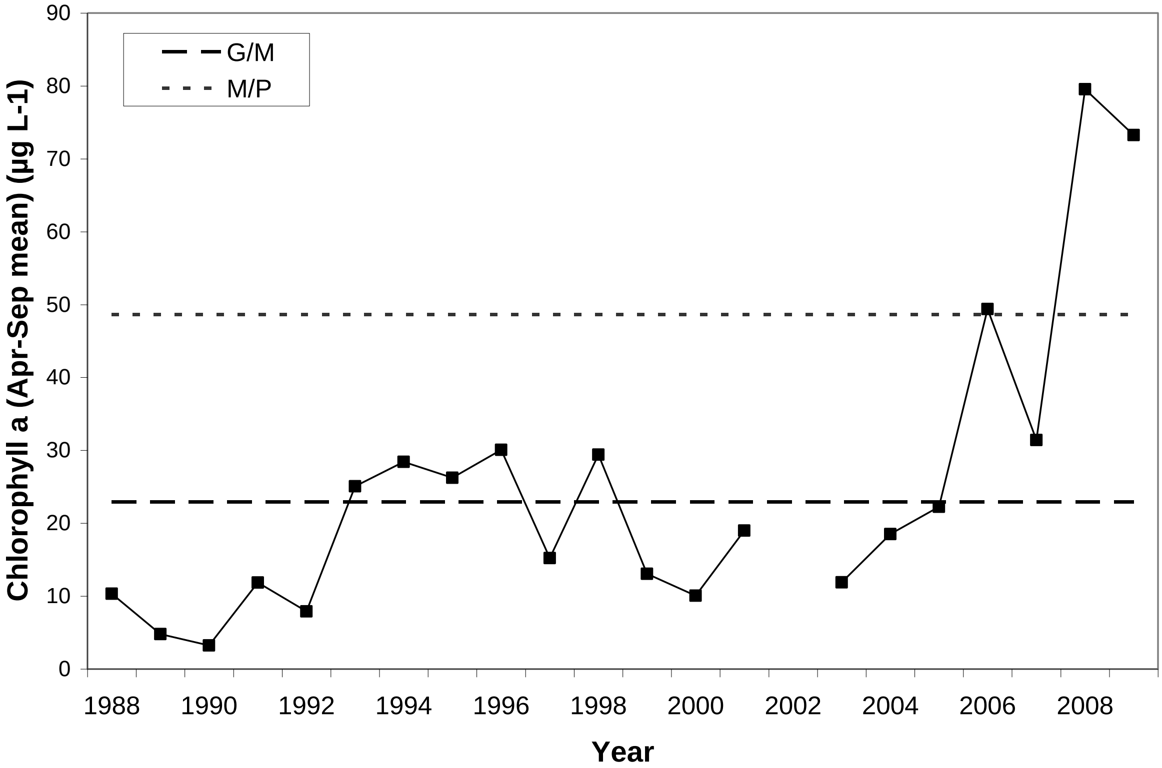

Figure 5.2 Trend in growing season (Apr-Sep) chlorophyll *a* concentrations in Loch Fitty in relation to Good/Moderate (G/M) and Moderate/Poor (M/P) status class boundaries.

The trend in the average chlorophyll *a* concentrations during the growth season (April to September) are shown in relation to the Good/Moderate and Moderate/Poor status class boundaries for the current Loch Fitty in Figure 5.2. Whereas in many previous years the loch would have been classified as Good or even High status (the growing season mean in 1990 was 4.8 µg L^{-1}), chlorophyll *a* concentrations in the loch in 2008 and 2009 were well above the Moderate/Poor boundary and the loch would have been classified as having Poor status.

WFD metrics for phytoplankton composition and blooms are still being developed. For this reason, the classifications are currently too preliminary for current status to be assessed as no status class boundary values have been agreed for the UK or the rest of Europe.

No information on the phytobenthos community was available. So, it was not possible to assess the current status of Loch Fitty in relation to this quality element.

5.3 Targets for the re-instated Loch Fitty compliant with WFD requirements

Methods for assessing the status of phytoplankton in lakes under the WFD have been formally adopted for phytoplankton chlorophyll *a*. The growing season mean chlorophyll *a* standards for the future Loch Fitty (a high alkalinity, shallow loch) are given in Table 5.2; the good/moderate boundary is set at 13.3 µg L^{-1}. This is equivalent to an annual mean of 7.5 µg L^{-1} (Carvalho *et al*., 2006).

Metrics for phytoplankton composition and blooms are still being finalised, but it is expected that the final UK method will be very different from the preliminary UK metric, which was based on percentage of cyanobacteria. Two metrics are currently under consideration. These are:

1) Phytoplankton Trophic Index (PTI). This metric is calculated as the average of the PTI scores for all phytoplankton taxa found in a sample of water from the loch during summer. PTI scores are an indication of the preference of each phytoplankton taxon in relation to a TP gradient. These scores have been calibrated from a large dataset consisting of data from more than 6000 European lakes (Phillips *et al*., 2010). Typical High/Good status taxa for high alkalinity lakes include chrysophytes such as *Bitrichia*, *Uroglena* and *Dinobryon*, and the diatoms *Tabellaria* and *Urosolenia.* Poor quality taxa, which reflect and "impacted" state, include the green algae *Actinastrum* and *Scenedesmus*, and the cyanonbacteria *Microcystis*, *Planktothrix* and *Aphanizomenon*.

2) Cyanobacterial abundance. This metric is calculated from the total biovolume of cyanobacteria in a sample of lake water, and comprises the average value from samples collected over the summer period (i.e. July to September). Details of this method are given by Carvalho *et al.* (2010). For high alkalinity shallow lakes, such as the restored Loch Fitty, it is recommended that values of this metric should be below 0.71 $mm^3\ L^{-1}$, the threshold for High status.

There is no metric available to assess the status of phytobenthos in lakes at present. However, a reference condition can be calculated from its alkalinity type. For the re-instated Loch Fitty, the reference (High status) lake diatom trophic index value would be 25.

6. WFD good ecological status target for re-instated Loch Fitty: macrophytes

The status of the macrophytes community in Loch Fitty has been reviewed in a comprehensive report by Stewart (2010). This report contains a detailed restoration plan that describes the practical measures required to maintain stocks of key macrophyte species during the coal extraction phase so that these can be reinstated into the new loch when coaling and re-instatement is complete The existing review is augmented, here, by focusing on the achievement of WFD Good Ecological Status in the new Loch Fitty. Current lake macrophyte evaluation systems in use within the UK, and the latest research from other European countries, have been used for this purpose. The findings of Stewart (2010) concerning the current ecological status and the proposed recolonisation strategy are also briefly described and critiqued, below.

6.1 Compilation of available data and information

Unlike some of the other biological elements, the aquatic macrophyte community of Loch Fitty has been well studied, especially over the last 30 years (Stewart, 2010). Loch Fitty was first surveyed between 1905 and 1909 by G. T. West (West, 1905; West 1910a; West 1910b). West's surveys form an invaluable baseline against which changes in Loch Fitty's water quality status over the intervening period can be assessed.

Loch Fitty, and other larger lochs in Fife from the original surveys by West (1905, 1910a, 1910b), were re-surveyed in 1982 (Young & Stewart, 1986), 1991 and 1995 (Bell, 2006). Further aquatic macrophyte surveys were also carried out in Loch Fitty in 1982 by the Nature Conservancy Council (Stewart, 1982), in 1997 by Scottish Natural Heritage (SNH, 1997) and by the in 2008 Scottish Environment Protection Agency (SEPA, 2008). In addition, a specific survey of the hybrid pondweed species *Potamogeton x suecicus* was undertaken between 1992 and 1994 (Preston *et al.*, 1999). Although a very detailed survey of Loch Fitty was undertaken in September 2010 by Stewart (2010), to identify colonies of key species and the location of areas of diverse aquatic macrophytes, the results from this survey were unavailable to the authors of this report.

6.2 Current ecological status

Stewart (2010) summarised all of the aquatic and wetland macrophyte species recorded from various surveys that have been carried out at Loch Fitty since 1905. During the period 1905-1909, a total of 28 submerged and floating-leaved species (including five charophytes and ten pondweeds – *Potamogeton* spp.) were recorded (West, 1905; West 1910a; West 1910b). In the most recent survey that is accessible to the authors, 20 submerged and floating-leaved species (including two charophytes and ten pondweeds) were recorded (SEPA, 2008). Stewart (2010) noted that, in addition to the species recorded in the 2008 survey, a further five species had been recorded in Loch Fitty over the last twenty years. These included additional pondweeds and charophytes. Stewart (2010) also identified a number of key aquatic and wetlands macrophyte species that, if still present, may need specific strategies to be developed for them to be conserved at this location. This was either because they were uncommon locally and/or nationally, or because of their limited ability to colonise new areas. These key species were *Callitriche hermaphroditica*, *Chara virgata*, *C. globularis*, *C. vulgaris*, *Carex aquatilis*, *C. diandra* ('Near Threatened'), *Potamogeton alpinus*, *P. filiformis* ('Nationally Scarce'), *P. gramineus*, *P. obtusifolius*, *P.* x *zizii* (hybrid between *P. gramineus* and *P. lucens*), *P. x suecicus* (hybrid between *P. filiformis* and *P. pectinatus*), and *Tolypella glomerata* ('Nationally Scarce').

Stewart (2010) concluded that a number of aquatic species had been lost since the 1900s, especially several species that are known to be sensitive to nutrient enrichment. This was demonstrated by the use of Mean Trophic Ranking Scores (TRS) and Plant Lake Ecotype Index (PLEX) scores as indicators of change under different nutrient conditions. Stewart (2010) calculated that, during the period 1905-1909, Loch Fitty had a TRS score of 7.6 and a PLEX score of 6.5, while by 2008 those scores had risen to 8.4 and 7.5, respectively. However, these analyses also showed that there had been little or no overall change in Loch Fitty's aquatic macrophyte community since at least the early 1980s. Stewart (2010) suggested that small inter-annual variations may have occurred as a response to variations in algal blooms over this period. Nevertheless, despite these apparent changes in the aquatic macrophyte community since 1905, Loch Fitty still has a relatively high diversity of species compared with other lochs in the same geographical area.

In terms of the WFD, Loch Fitty has been classified as having good ecological status (with a high degree of confidence) in relation to its macrophyte communities (SEPA, 2009). In 2008, SEPA surveyed the aquatic macrophyte community using four survey sectors and a combination of boat, wader and perimeter transects (SEPA, 2008). In 1997, Scottish Natural Heritage (SNH) surveyed the aquatic macrophyte community using a partial boat survey and shoreline sampling (SNH, 1997). Based on the results of these surveys, Loch Fitty is currently classified as a Type 10A eutrophic loch (Palmer, 1992) and as belonging to a Group I lake type (Duigan *et al*., 2006). These are mostly moderately large, base-rich lowland lakes, with *Chara* spp., *Myriophyllum spicatum* and a range of *Potamogeton* species. This classification is in contrast to the 1905-1909 period, when the list of recorded submerged and floating-leaved macrophyte species suggests that Loch Fitty was more mesotrophic in character than at present.

6.3 Targets for the re-instated Loch Fitty compliant with WFD requirements

Methods for assessing the status of macrophytes in lakes under the WFD have progressed further than for some of the other biological quality elements, but are still undergoing some revision. There has, for example, been a change in the method since Loch Fitty was assessed in 2008, as the method used by SEPA (2009) does not match the current published guidance (WFD-UKTAG, 2009). Since the WFD intercalibration exercise has not yet been completed, it is expected that the final method may be modified again before the new Loch Fitty is created.

At the time of writing, the current method for assessing the status of macrophytes is defined by the United Kingdom Technical Advisory Group (WFD-UKTAG, 2009; Nigel Willby, *pers. comm.*). This method includes assessment of five parameters (described below) to produce Ecological Quality Ratios (EQRs) for each, then subsequent combination of the five EQRs to produce a single EQR for macrophytes.

1. Lake Macrophyte Nutrient Index (LMNI). This is the average of the LMNI scores for all taxa found in the lake. LMNI scores are an indication of the preference of each plant for particular nutrient environment, such that taxa that 'prefer' less rich, oligotrophic waters have lower scores than taxa that 'prefer' richer, more eutrophic

waters. The average LMNI score is, therefore, an indicator of the nutrient environment of a lake, as experienced by the plants.

2. Number of Functional Groups (NFG). This is the number of distinct functional groups found in the assembly of macrophytes. Functional groups are groups of taxa that have similar morphology and habitat preference, so this is a measure of habitat diversity, or the number of potential types of habitat available to plants.

3. Number of Taxa (NTAXA). This is the total count of distinct species found in the lake and is a direct measure of richness.

4. Cover (COV), This is the proportion of the surface area of the lake that is covered by plants, expressed as a percentage. This is an indication of the overall suitability of the lake for macrophyte occupancy.

5. Relative cover of filamentous algae (ALG), This is the proportional cover of filamentous algal taxa in relation to the total cover of all macrophyte taxa. This parameter is included because filamentous algae are perceived to be both an indicator of nutrient enrichment and an undesirable element in their own right.

Values for the LMNI scores, membership of functional groups, and the taxa considered to be filamentous algae, are all included in the guidance document (WFD-UKTAG, 2009).

Using the UKTAG methodology, lake-specific reference conditions for each of the five parameters above can be calculated on the basis of altitude, mean depth, surface area, alkalinity, conductivity, freshwater sensitivity class and distance from the nearest coast. Ecological quality ratios (EQRs), which are values of between 0 and 1, are then calculated for each of these parameters, generally as the ratio of the observed value and the reference condition value. The five EQRs are then combined using some logical rules to create a single EQR for the macrophyte quality element of a lake. This final EQR can be used to assign a status class as outlined in Table 6.1.

A Microsoft Excel© based tool has been developed for assessing the ecological status of a loch based on its macrophytes by Nigel Willby, Stirling University, and this was used in the assessment by SEPA in 2009 (Ross Doughty, *pers. comm*.). An updated version of this tool may be available from either Nigel Willby or SEPA to calculate more accurate quality targets for future macrophyte communities in the restored Loch Fitty.

The method used to classify Loch Fitty in 2008 (SEPA, 2009) differs from the WFD-UKTAG guidance in that an additional parameter (the relative cover of invasive taxa) was also used. Now, the abundance of invasive macrophyte species is considered separately from the assessment of the macrophyte quality element (Willby, *pers. comm*.).

Table 6.1. Use of macrophyte EQRs to assign status boundaries in lakes.

Value of EQR	**Status class**
< 0.2	Bad
≥ 0.2 and < 0.4	Poor
≥ 0.4 and < 0.6	Moderate
≥ 0.6 and < 0.8	Good
≥ 0.8	High

Given the complexity of the above, it is clear that the task of setting target presence and abundance for macrophytes in the new Loch Fitty to meet WFD Good Status, is almost impossible. Indeed, the only target that can be provided with authority is that consistent with meeting 'the requirements of Good Status using the methodology defined at the time of assessment'. In the absence of specific targets, the following (relatively) simple guidance is provided:

- the macrophyte status in the lake in 2008 was defined as 'good' by SEPA (SEPA, 2009); it is, therefore, reasonable to assume that a plant community that is *as good as, or better than*, that found in 2008, will result in 'good', or better, status in the re-instated Loch Fitty.

- in this context '*as good as, or better than*' means:
    - having the same or a lower LMNI (6.71), which can be achieved by having the same or more species with lower LMNI scores, and/or the same or less species with higher scores
    - having the same (23), or a higher number, of total species
    - having the same (13), or a higher number, of functional groups
    - having the same (1.82%), or more, plant cover *per* species
    - having the same (0.03%), or less, relative cover of filamentous algae

- some caveats presented in Section 12.1.6, below, should be taken into account, especially in relation to keeping *Elodea canadensis* out of the new lake for as long as possible.

The taxa found in the 1905 and 2008 surveys are shown in Table 6.2. This includes functional group (FG) membership and LMNI score according to the WFD-UKTAG methods. PLEX and TRS scores are also provided for historical context. It is suggested that some of the species found in 1905 should be added to the target list for reinstatement, especially since it may not be possible to re-establish all of the species that were found in 2008. The species in Table 6.2 are arranged in ascending numerical order in relation to their LMNI score, so species at the top of the table should be considered for addition to the target list before those that are lower down.

Table 6.1 Submerged plant taxa found in Loch Fitty in 1905 and 2008 (Stewart, 2010), with associated scores from three assessment systems, i.e. Lake Macrophyte Nutrient Index (LMNI), Functional Groups (FG), PLEX and TRS (see text for details. NB. The TRS score for *Potamogeton* x *zizii* is the average of its parents, as this hybrid does not have its own score.

Name	1905	2008	LMNI	FG	PLEX	TRS
Potamogeton polygonifolius	+		3.5	16	3.08	3
Juncus bulbosus	+		3.72	4	3.08	3.7
Myriophyllum alterniflorum	+		4.54	7	4.23	5.5
Littorella uniflora	+		4.7	4	4.23	6.7
Potamogeton natans	+	+	5.16	16	4.23	6.7
Potamogeton gramineus		+	5.51	16	7.31	7.3
Nymphaea alba	+		5.54	12	3.08	6.7
Chara virgata	+	+	5.55	2	7.69	8.5
Nitella flexilis agg.	+		5.6	2	5.38	5.5
Potamogeton x *zizii*	+	+	5.69	16	7.69	8.6
Potamogeton praelongus	+		5.77	17	5.38	7.3
Potamogeton alpinus	+	+	5.79	16	5.38	5.5
Potamogeton perfoliatus	+	+	5.83	17	7.69	7.3
Callitriche stagnalis	+		5.98	6	7.69	7.7
Potamogeton berchtoldii		+	6.07	14	7.69	7.3
Potamogeton filiformis	+	+	6.16	15	7.69	10
Chara aspera	+		6.39	2	7.69	8.5
Hippuris vulgaris	+		6.4	7	7.88	7.7
Ranunculus peltatus	+		6.48	18	7.69	8.5
Sparganium emersum	+	+	6.59	13	7.5	10
Ranunculus hederaceus		+	6.6	11	7.69	8.5
Callitriche hermaphroditica	+	+	6.71	5	7.69	8.5
Potamogeton obtusifolius	+	+	6.72	14	6.54	7.3
Eleocharis acicularis	+		6.75	4	7.95	8.5
Nuphar lutea		+	6.92	12	6.92	8.5
Elodea canadensis	+	+	7.14	5	7.95	8.5
Chara globularis	+		7.18	2	7.69	8.5
Chara vulgaris	+	+	7.2	2	7.69	8.5
Persicaria amphibia	+	+	7.25	10	7.95	9
Potamogeton pusillus	+		7.61	14	7.95	8.5
Potamogeton crispus		+	7.64	17	7.95	8.5
Lemna trisulca		+	7.82	1	8.85	10
Myriophyllum spicatum	+	+	7.84	7	8.85	10
Potamogeton pectinatus	+	+	8.25	15	8.85	10

7. WFD good ecological status target for re-instated Loch Fitty: macroinvertebrates

As far as the authors are aware, the status of the benthic macroinvertebrate community in Loch Fitty has not been subject to any scientific study prior to the recent investigations prompted by the proposed development and restoration plans for the loch (Perfect, 2010). This study is reviewed within the context of setting site-specific targets for the macroinvertebrate community in the restored Loch Fitty to ensure compliance with WFD requirements for Good Ecological Status. Reference is made to current developments in this field across Europe and the available scientific literature has been reviewed.

7.1 Compilation of available data and information

The benthic macroinvertebrate community of Loch Fitty and its associated watercourses were surveyed in September 2010 (Perfect, 2010). Three locations on the Meldrums Mill Burn (the main inflow) and three locations on the Lochfitty Burn (the main outflow) were sampled mainly using a standard three minute kick sampling technique (ISO 7828: 1985, 1994), although in those areas of stream habitat where water flow was entirely still, timed sweep net samples were collected instead. In Loch Fitty itself, benthic macroinvertebrate samples were collected from six littoral sites using timed sweep net sampling techniques through marginal vegetation and substrate, and from three deeper water areas using an Ekman grab sampler.

7.2 Current ecological status

Perfect (2010) found a fairly uniform littoral benthic macroinvertebrate community in the loch, with a relatively restricted species richness (in comparison with the streams that were sampled). This was dominated by a few groups of animals, mainly *Asellus aquaticus*, *Gammarus pulex* and *Sigara fallen*. Only small numbers of the more sensitive EPT (Ephemeroptera, Plecoptera and Trichoptera) taxa were present. This probably partly reflects the relatively sheltered and uniform littoral habitats that were sampled, but is also indicative of a benthic macroinvertebrate community that is typical of a nutrient-enriched, shallow lake where species that can process and digest organic matter are favoured. No macroinvertebrate species of particular conservation interest, in terms of rarity value or restricted local or national distribution, were found.

Unfortunately, there are no historical macroinvertebrate data with which to assess any long-term trends in these communities within Loch Fitty. Although the loch has been classified as being of High ecological status for the benthic macroinvertebrate parameter in SEPA's current WFD classification (SEPA, 2009), neither of the two techniques listed for assessing macroinvertebrate quality elements under the WFD (i.e. Lake Acidification Macroinvertebrate Metric [LAMM] and Chironomid Pupal Exuvial Technique [CPET]) have been applied to the loch. In reality, this 'high' status reflects the fact that, as currently applied, SEPA's WFD classification scheme defaults to High status when there are no data for a particular quality element available.

It is highly unlikely that SEPA will ever apply the LAMM tool (WFD-UKTAG, 2008c) to Loch Fitty, as there is no known acidification pressure at this site (Ross Doughty, *pers comm*.). However, the CEPT assessment technique, which is based on using chironomid pupal exuviae as indicators of lake ecological quality, may be applied to the loch in the future to assess the impact of nutrient pressures (Ross Doughty, *pers comm*.). The CPET technique has been developed within the UK as a method for monitoring, assessing and classifying lakes in relation to the requirements of the WFD (WFD-UKTAG, 2008c) and its potential use at Loch Fitty is discussed in more detail, below.

7.3 Targets for the re-instated Loch Fitty compliant with WFD requirements

The UK have not yet developed a comprehensive (all groups) WFD-compliant classification tool for lake macroinvertebrates that detects the impact of nutrient enrichment on these communities. Neither have they published a formal set of relevant targets for achieving Good Ecological Status for this group. Targets that are specifically related to hydromorphological (and implicitly also to eutrophication) pressures, are currently being developed within Europe and it is hoped that these may be available within the next year or so (Martin Pusch, *pers comm.*).

In relation to the current proposal, it is not possible at present to set specific and robust targets for the benthic macroinvertebrate community, as a whole, for the re-instated Loch Fitty. However, reference values can be set using the group specific CPET methodology, which is compliant with WFD requirements and is designed to detect the impact of nutrient enrichment (WFD-UKTAG, 2008c). The advantage of using this technique is that it is based on chironomid pupal exuviae, which are ubiquitous, species rich, exhibit high ecological diversity and occur at high densities (Wilson & Ruse, 2005). They also provide an integrated sample of the whole lake, are simple and quick to sample, and their identification to genus level is relatively straightforward using the key of Wilson and Ruse (2005).

The CPET methodology is described in detail by WFD-UKTAG (2008c) and involves deriving ecological quality ratios (EQRs) by comparing the list of chironomid pupal exuviae with known nutrient sensitivities found at a lake (in four samples collected from April to October) with a site-specific reference value for the lake that is derived from spatial data. The value for the reference condition parameter applicable to the lake is calculated as follows:

$$\text{Reference value for parameter} = -1.13 - (0.357 \text{ x } \text{Log}_{10}S) - (0.455 \text{ x } \text{Log}_{10}D_{mean}) + (0.376 \text{ x } \text{Log}_{10}RT_{mean}) + (0.364 \text{ x } \log_{10}CA)$$

where:

D_{mean}	= mean depth of the lake (metres)
S	= surface area of the lake (hectares)
CA	= catchment area of the lake, excluding the surface area of the lake (hectares)
RT_{mean}	= mean length of time in that water is retained in the lake (days)

Using the environmental data in Table 7.1, which includes a designed retention time provided by SCCL (Theo Phillip, *pers comm.*) the CPET reference value for the restored loch was calculated to be -0.158.

Table 7.1 Environmental data used to calculate the reference CPET value for the restored Loch Fitty.

Variable	**Value**
Surface area (S)	59 hectares
Mean depth (D_{mean})	3.6 m
Retention time (RT_{mean})	60 days
Catchment area (CA)	1843 hectares

When Loch Fitty is restored, chironomid pupal exuviae can be collected using the methodology given by WFD-UKTAG (2008c) and the observed value of the parameter can be calculated using the sum of the nutrient sensitivity scores of the different chironomid taxa recorded and the total number of taxa found (see WFD-UKTAG, 2008c for details). The ecological quality ratio (EQR_{CPET}) for the parameter can then be calculated by comparing the observed value of the parameter with the derived reference value (i.e. -0.158), as follows:

$$EQR_{CPET} = \{[2 - (\text{observed value of parameter} + 1)] \div [2 - (\text{reference value for parameter} + 1)\} \div 1.18$$

In principal, the closer the EQR_{CPET} value is to 1, the better the ecological status of the loch. However, at this stage, it is unclear what band range of EQR_{CPET} values will correspond to WFD Good Ecological Status. It is expected, however, that the re-instated Loch Fitty, with improved water quality and better hydromorphological conditions, will support a more diverse macroinvertebrate community than at present and that this will be reflected in an increase in EQR_{CPET} values.

8. WFD good ecological status target for re-instated Loch Fitty: fish

In contrast to other biological aspects of the loch, the fish community of Loch Fitty, and its future management, have already been specifically addressed within the context of the proposed development and restoration works by Bull (2010a), Bull (2010b), Bull (2011a) and Bull (2011b). For the fish community, these studies overlap significantly with a number of the objectives of the present study. Where overlap occurs, reference has been made to these earlier findings, allowing effort here to be focused on the remaining objectives. In particular, effort has been focused on the specific objective of identifying similar projects elsewhere in Europe and, thus, recommending opportunities for appropriate further contacts and site visits.

8.1 Compilation of available data and information

As is frequently the case for European freshwaters, the earliest information available on the fish community of Loch Fitty originates from before its earliest scientific studies. Peter S. Maitland (Fish Conservation Centre, U.K., *pers. comm.*) holds fish occurrence records dating back to 1791 which note the local presence of eel (*Anguilla anguilla*), perch (*Perca fluviatilis*) and pike (*Esox lucius*). Brown trout (*Salmo trutta*) have almost certainly been present since soon after the loch's formation many thousands of years ago, although small-bodied species such as minnows (*Phoxinus phoxinus*) and stone loach (*Barbatula barbatula*) were typically overlooked in such non-scientific records. Based on a study of the loch carried out in 1969, MacKenzie (1975) reported 'excessive numbers' of perch and pike alongside more desirable brown trout. In relation to the latter, he noted that spawning gravels in the inflowing Meldurms Mill Burn were badly silted as a result of mining activities in the upstream catchment. MacKenzie (1975) went on to describe the use of rotenone, a plant-derived piscicide, to kill and remove the entire fish community of the loch in 1970, prior to it being redeveloped as a trout fishery by the subsequent successful introduction of yearling brown and rainbow trout (*Oncorhynchus mykiss*) in early 1971. Although the source of these two trout species was not specified, it is clear that it was not the loch itself for the rainbow trout, as this is native to North America, and it was unlikely to have been so for the brown trout. Consequently, if the original Loch Fitty brown trout population had any unique genetic characteristics, it is likely that these were lost at this time. The loch was subsequently operated as a trout fishery for many years until its closure in late 2007, during which time other fish species were also introduced to three ponds created to the south and east shores of the loch. Note that, with the reasonable assumption that the rainbow trout did not establish a locally reproducing population and the absence of any stocking for a number of years, this species can now be assumed to be no longer present in the system.

The only scientific survey of the fish communities of the loch and the immediately adjacent areas of its primary inflowing and outflowing streams is that of Bull (2010a), which used the combined techniques of electric fishing, gill netting, fyke netting and seine netting during August 2010. Notably, this investigation found no brown trout in the loch itself, which was dominated by perch and roach with much lower numbers of eel and pike. The main inflowing stream was found to contain eel, minnow perch, roach, stone loach and three-spined stickleback (*Gasterosteus aculeatus*), but no brown trout or pike. The main outflowing stream contained brown trout, minnow, perch, pike, roach, stone loach and three-spined stickleback, but no eel. One inflowing stream site and one inshore loch site were specifically sampled for larval lampreys (*Lampetra* sp.), but none were found. A sub-sample of fish was examined for general health, including parasites, and a relatively high prevalence of the fish louse, *Argulus foliaceus*, was noted on perch, pike and roach. Finally, a survey of potential brown trout spawning habitats in the system confirmed that the sedimentation problems noted by McKenzie (1975) still persisted.

In addition to the above study, Bull (2011a) describes plans for further gill netting to be undertaken at sites within the loch during 2011, to provide data appropriate for use by SEPA to classify the fish community for the purposes of the WFD. The aim is to use a classification tool that is apparently still in development. This 2011 sampling programme will also use gill netting and fyke netting in the ponds immediately adjacent to the loch to ascertain if additional undesirable species have been introduced to these habitats by the now closed fishery. Electric fishing and habitat assessment of the system's flowing waters will also be included in the survey. Depending on the results, netting may also be undertaken in the standing waters of the Lassodie Burn. The data to be produced by this 2011 sampling are critical for both a robust assessment of the current ecological status of the fish community and for detailed aspects of the recolonisation strategy, recovery scenarios and management options.

Further relevant information concerning the proposed scheme's Ecological Mitigation Corridor and the re-instated Loch Fitty itself are available in Bull (2010b) and Bull (2011b), respectively, but its detailed consideration here is considered to be unwarranted.

8.2 Current ecological status

As reviewed above, the surveys of Loch Fitty and its immediate watercourses reported by Bull (2010a) have, amongst other things, documented an apparent absence of native brown trout from the loch itself and from its main inflowing stream, together with an abundant population of introduced roach in the loch itself, its main inflowing stream and its outflow. Other species recorded comprised eel, minnow, perch, pike, stone loach and three-spined stickleback, with several fish species bearing considerable loads of the parasite *Argulus foliaceous*. In the absence of detailed historical surveys it is impossible to determine which of these species are truly native to the loch. However, even if the data to be produced by the further sampling described by Bull (2011a) are necessary to produce a WFD-compliant formal assessment, on the basis of the information already available it is clear that the fish community of Loch Fitty is currently of poor ecological status under any reasonable definition of the term.

8.3 Targets for the re-instated Loch Fitty compliant with WFD requirements

Within the U.K., to the best knowledge of the authors, the appropriate competent authorities have not yet developed a WFD-compliant classification tool for fish in standing waters, nor have they published a formal set of relevant targets. Consequently, specific and robust targets for the fish community of the re-instated Loch Fitty cannot be given at present. However, in this absence of authoritative guidance, Bull (2011b) has described a restoration strategy with the target of achieving a fish community comprising only brown trout, eel, stone loach and three-spined stickleback. Achievement of such a target requires the removal of minnow, perch, pike and roach from the system. Roach is clearly an introduced species and this factor, together with its potential negative impact on water quality, mean that the case for its eradication is overwhelming. The case for the removal of minnow, perch and pike is less clear, given that at least the latter two species have been present for at least 200 years. However, the new and simplified community proposed by Bull (2011b) errs on the side of caution in terms of appropriate composition in a WFD context and so it can be readily supported. In practice, the successful eradication of minnow, perch, pike and roach from the upstream system using only netting and electric fishing is likely to prove difficult, as discussed elsewhere in the present report.

9. WFD good ecological status target for re-instated Loch Fitty: zooplankton

Although not presently covered by the WFD, it is also recommended that zooplankton community composition is taken into account, too, because it has a very strong influence on algal species composition and abundance. Predation by fish such as roach can reduce zooplankton numbers to a level that results in increases in algal abundance, chlorophyll *a* concentrations and nutrient levels. If chlorophyll *a* levels increase, the water becomes turbid and this has a negative impact on macrophytes, which may also cause failure to meet WFD targets.

10. Recolonisation strategy

10.1 Phosphorus and nitrogen

Achieving the WFD TP standards, and any N standards that are set in the future, will be aided by encouraging macrophyte recolonisation through re-planting of submerged and floating aquatic plants. This is because aquatic plants play an important role in reducing nutrient release from sediments and establishing a stable clear-water state in shallow lakes (Scheffer 1998). Non-invasive native macrophytes species, such as white and yellow water lilies, could be actively planted, although care should be taken to prevent accidental introduction of invasive, non-native species that could degrade the ecological status of the loch.

10.2 Phytoplankton and phytobenthos

Algae, both phytoplankton and phytobenthos, will naturally recolonise from inflowing streams and from dispersal through the air and attached to water birds. Re-planting of macrophytes will also introduce an inoculum for re-colonisation. No specific strategy, therefore, needs to be developed for the recolonisation of these groups.

10.3 Macrophytes

Stewart (2010) suggests a recolonisation strategy that involves taking aquatic plants and sediment cores from Loch Fitty, keeping the material in ponds and then using it to recolonise the loch after reconstruction. This strategy is entirely appropriate. It makes use of locally sourced plants, thereby maintaining the genetic integrity of the Loch Fitty populations. While nearby Loch Leven provides a good source of suitable plants for passive colonisation, in general there seem to be relatively few other good sources within suitable distance (O'Hare *et al.*, submitted). Some areas of the plan require further detail, especially the methods by which the plant and sediment will be maintained over the seven year period between removal and reinstatement. A detailed plan for the recolonisation of the loch should be developed, in due course.

The conditions in the holding ponds should be specified in terms of water chemistry, sediment chemistry, substrate type, depth and shape. Water and sediment should be low in P and N, and a suitable depth of water needs to be provided for the different species, ranging from *circa* 0.3 to 3m. This is necessary to ensure that the plants have the correct growing conditions and the ponds are not vulnerable to extremes of weather, either from freezing or water level fluctuations. Sourcing suitable water, which is low in nutrients, for maintaining the water level in the ponds may be difficult. Rainwater alone is likely to be too low in pH to suit plants adapted to the high alkalinity waters of Loch Fitty. It is possible to adjust the pH of the rainwater to compensate for this and this may be the most practical solution. A routine monitoring programme should be put in place to make sure that the plants remain in good condition, with basic observations made on the health of the plants at weekly intervals during the growing season.

10.4 Macroinvertebrates

The proposed recolonisation strategy for macroinvertebrates, outlined here, focuses on creating the right habitat conditions for these animals in the new Loch Fitty. More specifically, this involves creating an environment that is naturally low in nutrients, has a suitable mix of benthic substrate types, and has a native in-lake flora around its margins. This is an entirely appropriate strategy, and will create conditions in which a macroinvertebrate fauna of high ecological status can thrive.

Early records from 1910 indicate that parts of the shoreline were already artificial by that time and that the shoreline was physically diverse. The northern shoreline of the loch was described as stony, in the south it was sandy and more vegetated, while to the west the shoreline merged into bog (Murray & Pullar, 1910). Reed and *Chara* beds were also present. All of these meso-habitats can support different assemblages of benthic macroinvertebrates. As the intention is for the new lake to contain the same, or similar, natural shoreline components, a diverse macroinvertebrate fauna can be expected. The placement of the new meso-habitats reflects the original positions of these habitats, e.g. the single bank along the north shore and reedbeds to the west. As each shoreline has a different magnitude of exposure to wind and wave disturbance, which can influence both substrate type and macroinvertebrate community, the placement of these meso-habitats is important to the success of the restoration project.

Unlike the recolonisation strategy for aquatic macrophytes there is no formal plan to reinstate macroinvertebrates collected from the loch. Instead, the strategy is dependent on natural immigration. This, again, is an entirely appropriate approach. There is substantial overlap between in-stream and in-lake macroinvertebrate communities and the upstream feeder streams will provide colonists for the lake, especially to the more exposed, gravelly shores. For this reason, it is essential that the quality of these feeder streams is assessed and, where necessary, improved before they are reconnected to the re-filled Loch.

The majority of benthic macroinvertebrates within a lake, including the chironomids used to calculate CPET scores, are the juvenile stages of flying insects that are capable of colonizing new habitats quickly through egg deposition. It is only these relatively long-lived juvenile stages that spend their life in water (Williams & Feltmate, 1994; Armitage *et al*., 1995). The remaining groups of macroinvertebrates, such as crustaceans, are carried by vectors such as birds. Experience from reservoir construction suggests that colonisation of new standing waters by these animals is rapid and that these new communities stabilise within a few years (Armitage, 1983).

10.5 Fish

For the fish community, the recolonisation strategy has already been considered effectively by Bull (2010b) in relation to the Ecological Mitigation Corridor and by Bull (2011b) for the re-instated Loch Fitty itself. Consequently, the strategy will not be revisited here except to note that it is based on the deliberate introduction of desirable fish species and numbers by stocking, although it also addresses the potential recolonisation of the loch by undesirable fish species, particularly roach, from the connected watercourses.

11. Monitoring strategy for ongoing assessment of restored loch meeting WFD targets

11.1 Catchment derived nutrient inputs

While the lake is empty, it is important that sources of high nutrient loads within the catchment are identified and addressed. Although it has been suggested that most of the nutrients entering the loch at present are associated with soil and river bank erosion problems (Jarvis & Quin, 2010), closer examination of the available data for the main inflow suggests that this may not be the case. Runoff from agricultural land generally has an OP:TP ratio of 0.4 or less if the main source is soil erosion, whereas values higher than this tend to suggest important sources of soluble phosphorus (OP) such as sewage works or septic tank discharges and/or seepage from agricultural storage areas such as slurry tanks. Figure 11.1 shows that the main inflow to Loch Fitty is often high in soluble P so it is recommended that a catchment survey of in-stream P concentrations (a "streamwalk") is undertaken under both high and low flow conditions to identify the likely sources of high OP inputs.

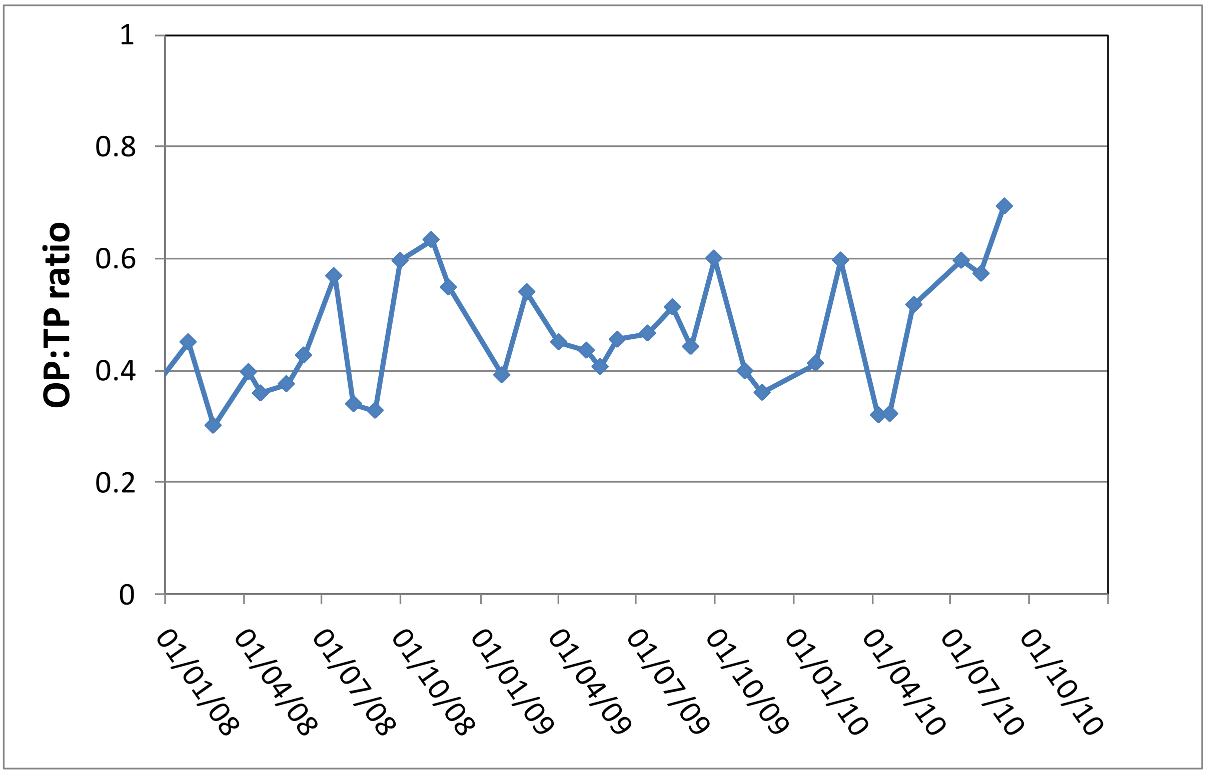

Figure 11.1 Ratio of orthosphosphate (OP) to total phosphorus (TP) concentrations in the main inflow to Loch Fitty, 2008-2010.

After re-instatement, the main external sources of nutrient inputs to Loch Fitty will continue to be associated with runoff from land, domestic waste, rainfall, roosting birds, *etc*. Although land use based modelling methods can give a rough estimate of the inputs from these sources to receiving waters, the method is not accurate enough to support site specific management activities because most are based on annual runoff values derived from very broadly applicable, land use specific, nutrient export coefficients. For site specific management purposes, detailed measurements of nutrient (N, P and Si) delivery from the catchment to the receiving waterbody are essential. These measurements should be made on a fortnightly basis, with additional, close interval, sampling during high runoff events. Regular sampling throughout the year is important because the nature, size and seasonality of the inputs (loads) can have a significant impact on lake response. Detailed sampling during high runoff events is also important because such events, though infrequent, can deliver 80 per cent of the annual nutrient input to the lake over just a few days (Defew, 2008).

Measurements of the water quality and flow in the inflows to the loch should be examined for evidence of especially high levels of pollution from particular parts of the catchment. This should then be followed up with site visits, wherever possible, to determine the cause(s) of these high discharges with a view to advising on how these can be mitigated. Landowner interviews should be conducted to address issues of identified point sources using SEPA's Diffuse Pollution Audit Form to record any responses, if appropriate.

Recommendation: Water quality problems in the inflow streams should be addressed while the loch is empty through targeted catchment monitoring and management activities. Once the loch is re-instated, fortnightly monitoring of inflows supplemented with occasional intensive monitoring during high runoff events is recommended. This should be undertaken to support the recovery process by highlighting any degradation in inflow quality before it causes lasting damage to the loch.

11.2 Phosphorus release from sediments

An initial assessment of the available in-lake nutrient monitoring data suggests that in-lake sediment nutrient processes are not the main driver of water quality in the loch at present. However, it should be noted that any catchment management activity aimed at improving water quality within the existing lake could result in a switch from catchment derived to sediment derived sources of P during the subsequent recovery period. This process can cause significant delay in lake recovery, which may last for many years (Sas, 1989). Although the current P laden sediments will not be transferred to the new Loch Fitty, it is important to monitor the accumulation and recycling of P within the sediments of the new loch as re-instatement takes place.

Sediment P release is commonly estimated using a combination of the approaches, as follows: (1) a mass balance approach, if summer internal loading peaks are evident, (2) sequential fractionation analyses of sediment samples collected from the lake bed to quantify the "release sensitive" sediment P pool, and (3) sediment core studies to assess the P flux per unit area of lake bed.

The mass balance approach subtracts the lowest TP concentration in spring from the highest concentration in autumn and assumes that the difference in concentration results from sediment P release, alone. However, over this period, variable amounts of TP will be entering the loch from the catchment and leaving the loch from the outflow, so it is important that detailed catchment loading data are available so that this can be corrected for.

Estimating the whole lake stock of release sensitive P is best conducted seasonally to assess the different processes of sediment uptake (i.e. winter and spring) and release (i.e. summer and autumn). Intact sediment cores should be collected from sites spanning the depth gradient of organic sediment in the restored loch. The sediment cores should be sectioned at 2cm intervals to a depth of 20 cm and the untreated sediment should be subjected to the following P extraction scheme, which follows that of Psenner *et al*. (1988),:

(1) extraction in 1M NH_4Cl for 30 min and quantification of total soluble P (TSP) [‘labile-P’];

(2) repeated (5 min) extraction with 0.11 M $NaHCO_3$ / 0.11 M $Na_2S_2O_4$ for 1 h and quantification of TSP [‘reductant-soluble P’];

(3) repeated (5 min) extraction in 1 M NaOH for 16 h followed by quantification of soluble reactive P (SRP) [‘metal oxide adsorbed P’] and TSP;

(4) repeated (5 min) extraction with 0.5 M HCl for 16 h followed by quantification of TSP [‘apatite bound P’];

(5) digestion with 30% (v/v) H_2SO_4 and 8% $K_2S_2O_4$ at 121 ºC for 30 min followed by SRP quantification [‘residual P’]. Organic P [‘organic P’] was quantified by subtracting NaOH-SRP from NaOH-TSP in step (3), above.

The sum of the ‘labile P’, ‘reductant-soluble P’ and ‘organic P’ fractions represents the “release sensitive” P pool in the sediment (Boström *et al*., 1982). The mass of sediment TP and release sensitive P can then be estimated at the whole loch scale if accurate knowledge of the sediment distribution across the bed is available. This spatial and seasonal data must be collected to underpin any follow up management required to control sediment P release during the recovery process.

Recommendation: Assess internal P release from the sediments and the accumulation of release sensitive P in the sediments over the recovery period to underpin any follow up management required to control sediment P release during the recovery process.

11.3 Routine monitoring of in-lake water chemistry, chlorophyll *a* concentrations, phytoplankton and zooplankton species composition and abundance

The sampling regime required for assessing water quality monitoring data against WFD water quality targets for TP and chlorophyll *a* was recently quantified by Carvalho *et al*. (2006). In summary, these authors found that single annual spot sampling produced errors of ±109% and ±48% in TP and chlorophyll *a* annual means, respectively, while quarterly sampling produced corresponding errors of ± 32% and ±14%, bi-monthly sampling produced errors of ±21% and ±10 %, monthly sampling produced errors of ±11% and ±5%, and fortnightly sampling produced errors of ±5% and ±3 %. To achieve mean annual estimates that are within acceptable levels of uncertainty, i.e. ±10%, a monthly sampling strategy is required for both chlorophyll *a* and TP concentrations.

It is recommended that samples for monitoring species composition and abundance of phytoplankton and zooplankton communities are collected in parallel to those for chlorophyll *a* and TP concentrations.

Recommendation: Chlorophyll *a* and TP concentrations, and species composition and abundance of phytoplankton and zooplankton communities should be measured at monthly intervals to assess compliance with WFD water quality targets, where appropriate, and provide supporting data for on-going lake management activities.

11.3.1 Routine monitoring of phytobenthos

Samples of benthic diatoms should be collected annually by brushing or scraping the upper surface of cobbles or small boulders obtained from the littoral zone of the loch, or from submerged stems of emergent macrophytes in areas dominated by fine sediments.

11.3.2 Routine monitoring of macrophytes

Annual macrophyte surveys should be undertaken over the recovery period, following WFD recommended methods.

11.3.3 Routine monitoring of macroinvertebrates

Surveys of the macroinvertebrate communities should be undertaken four times a year, between April until October. The surface of the lake should be skimmed with a hand net (nominal mesh size: 250 µm) with an extendable handle. The samples should be analysed following the recommendations of WFD-UKTAG (2008d).

11.3.4 Routine monitoring of fish

Annual fish surveys should be undertaken over the recovery period, following WFD recommended methods when these are available from the competent authority.

11.4 Assessment of in-lake processes

An understanding of site-specific lake processes (e.g. the seasonality of P or N limitation, chlorophyll:TP ratio; TN:TP ratio; denitrification rates; stratification; redox conditions; internal loading etc) is essential when developing future management programmes (Jeppesen *et al*., 2005; Spears *et al*., 2007; May *et al*., 2010). Lakes which appear to have similar water quality issues can require very different management approaches and can respond to management very differently depending on site-specific traits (Jeppesen *et al*., 2005). Additional information can also be gained with respect to the timing of management practices. For example, many shallow lakes are predominantly P limited; however in-lake processes such as internal P loading and denitrification, which run concurrently in summer, can result in sporadic periods of N limitation. Additionally, sediment P is predominantly released in lakes in the summer (Spears *et al*., 2007). An understanding of these processes can be inferred from an assessment of seasonal and spatial (with depth) variations in physical, chemical and biological determinands as discussed below. Additionally, these processes vary at a seasonal timescale and a combined sampling approach of outflows and open water and sediments is required to roughly estimate these processes.

Recommendation: Monthly sampling of outflows, seasonal sampling of open water depth profiles and annual sampling of sediment nutrient content are recommended to provide information on in-lake processes to inform future management decisions.

12. Likely timescale for attainment of good status

12.1 Factors affecting lake recovery

A recent review of over 700 'lake-equivalent' case studies (LECs) indicated that a wide range of pressures are expected to influence the recovery of lakes following management to solve eutrophication problems. Many of these pressures are classified as primary eutrophication pressures that will result in an increase or decrease in nutrient inputs to a lake (Table 12.1). However, a range of secondary (i.e. indirect) pressures can also impact upon, and delay, the recovery process. It is essential to understand the combined impacts of these multiple pressures when designing restoration solutions. Also, planning contingency management to address these pressures is required to minimise the risk of undesirable restoration outcomes when lake restoration is attempted (Hilderbrand *et al*., 2005).

In the case of Loch Fitty, it is important to identify any potentially confounding pressures and mitigate their impacts, prior to re-instating the loch. It is also important that the complexity of restoring heavily impacted lakes is acknowledged within a realistic long-term management plan that allows for continual assessment of ecological succession in line with existing and emerging environmental pressures. In the following sections we review the success/failure of a range of case studies in which waterbodies have been either created or have undergone sediment removal and/or dewatering. The results of these studies have been used to identify potentially confounding pressures and response periods that should be considered in relation to Loch Fitty, given the proposed restoration work.

Table 12.1. Primary and secondary drivers and pressures checklist for eutrophication in freshwater lakes used in the analysis of pressure-impact relationships. Drivers are underlined and pressures are in italics.

Eutrophication drivers and pressures		
Primary		**Secondary**
<u>Agriculture</u>	*Boat disturbance of sediments*	*Sediment dredging*
Fertiliser application	*Gardening practices*	*Boat disturbance*
Animal waste	*Waterfowl feeding*	*Mine pollution*
Soil errosion	<u>Population growth</u>	*Invasive species spread*
<u>Industry</u>	*Waste water treatment plants*	*Pesticide application*
Textiles	*Septic tank discharges*	*Climate change*
Food manufacturing	*Waste disposal*	*Fish stocking*
Paper mill discharges	Construction	*Fishery exploitation*
Mining discharges	*Transport run-off*	*Acidification*
Distillery discharges	*Detergent and soap use*	*Macrophyte removal*
Aquaculture discharges	<u>Other pressures</u>	*Industrial thermal-regulation*
Sediment dredging	*Lake-bed sediments*	*Water level management*
<u>Tourism and recreation</u>	*Waterfowl feaces*	*Waterfowl introduction*
Food waste disposal	*Atmospheric deposition*	*Extreme weather events*
Fish stocking	*N_2 – fixation*	

12.1.1 Nutrient inputs from the catchment

There is a lack of understanding of the main nutrient sources and scales of input to the current Loch Fitty from its inflows. This lack of knowledge means that any forecasting of TP concentrations based on nutrient inputs will have a high degree of uncertainty. It is welcomed that monitoring of the surface waters to improve load estimation, and the implementation of targeted measures to reduce catchment loading, are to be initiated at the commencement of the development (i.e. 7.5 years prior to re-filling) to provide baseline nutrient loading data for the restored loch. During refilling, water with high nutrient content should be avoided and only sources of water with the low nutrient concentrations should be used to re-fill the loch. Therefore, ongoing monitoring of the groundwater abstraction wells proposed to be used to refill the restored loch is required to ensure water quality is within acceptable limits. Any inflow streams that are found to have high TP or N concentrations should be investigated and management action taken to reduce these inputs before they are reconnected. As stated above, this problem will be addressed while the loch is empty. A suitable regime for monitoring nutrient input to the restored loch is outlined in Section 11.1.

12.1.2 Internal phosphorus loading

As the proposed restoration works include an initial dewatering step, the potential direct ecological impacts of the removal of sediment without dewatering can be ignored in the case of Loch Fitty. In addition, the proposed restoration works are designed to enhance the recovery of Loch Fitty, by reducing both sediment P release and nutrient inputs from the catchment and removing heavy metal laden sediment.

A range of factors regulate the recovery of lakes following dewatering and sediment removal. Chemically, the reduction of sediment P release is expected to be a key outcome for Loch Fitty. However, care needs to be taken to ensure that P-laden sediments are not re-introduced to the loch during restoration as this could limit the recovery process. In particular, where recolonisaton of macrophytes is an important step, it is important to ensure that large amounts of P-laden sediments are not re-introduced to the site as plant substrate during macrophyte translocation.

In general, sediment P concentrations increase towards the sediment surface. This is because (1) organic matter is deposited at the sediment surface and remineralised in the upper layers by bacteria and (2) P is bound within iron and manganese oxyhydroxides under aerobic conditions (i.e. towards the sediment surface) and released under anaerobic conditions (i.e. in deeper sediment layers) and so focuses towards the aerobic surface layers. This store of P is released to the water column when bottom water and surface sediment conditions are anaerobic, most commonly in warm summer months when biological oxygen demand outstrips supply (Spears *et al*., 2007). Ideally, the new Loch Fitty bed sediments will have a high P binding capacity with a low sensitivity to anaerobic release. However, if not, amendment of the bed sediments may be necessary to achieve this.

Amendment to enhance the P sorbing capacity of the lake sediments may be conducted prior to the sediments being introduced to the loch or retrospectively if internal loading occurs after the loch has been re-instated. A range of products that can be used for this purpose are commercially available (Table 12.2) and the selection of the appropriate product should be based on the properties of the proposed bed material.

Table 12.2. Summary of commercially available materials to control sediment P release at the whole lake scale, and references to documented examples of their application.

Method	Mechanism	Reference
Physical		
Discing	RIPLOX, manual injection of chemicals (FeCl, $Ca(NO_3)_2$, $CaCO_3$ into bed sediments to manipulate P sorbing capacity and/or denitrification rate through pH.	Ripl, 1976
Capping – Passive		
Sand/gravel	Adding sand/gravel/clay (> 5 cm layer) to existing bed sediments to reduce diffusion of nutrients from sediment to water column.	Cooke *et al*,. 2005
Capping-active		
Alum	Buffered (>6.5 pH) alum addition to sorb phosphate in water column and settle out to lake bed, also acts as a flocculent but potentially toxic Al^{3+} release if low pH conditions are not correctly buffered.	Cooke *et al*., 2005
Modified Zeolite	Al-zeolite clay with high binding affinity for NH_4-N and moderate binding affinity for PO_4-P. Applied as slurry to the water column where it binds dissolved nutrients before settling to the lake bed where it caps sediment nutrient release when applied at the correct dose.	Hickey & Gibbs, 2010
Phoslock®	Lanthanum-bentonite clay with high binding capacity for PO_4-P. Effective over a wide range of pH (4.5-8.5) and solubility product with PO4-P is very low, so it forms a very stable La-P complex in the sediment. Concerns about La toxicity are still being investigated.	Hickey & Gibbs, 2010

Van der Does *et al*. (1992) suggest that lake morphometry and hydraulic residence time be taken into account when deciding on the likely success of sediment removal for enhancing the recovery process. Additionally, Sas (1989) demonstrates that sediment TP concentration can be used as an indicator of the likely transient recovery time, as a result of sediment P release, following external load reduction.

Sas (1989) classified the sediment TP concentration in the upper 15 cm of lake sediments in relation to the estimated recovery time following reduction of external P loading. At concentrations of less than 1 mg TP g^{-1}d.w., internal loading is expected to be negligible with moderate summer sediment-P release events. At concentrations between 1 mg TP g^{-1}d.w. and 2.5 mg TP g^{-1}d.w., net annual sediment-P release will be high, initially, with recovery expected within a 5 year period; a high summer release event would be expected to occur that will be affected by pH, dissolved oxygen and microbial activity. At concentrations in excess of 2.5 mg TP g^{-1}d.w., net annual sediment-P release will occur for more than 5 years; in this situation, sediment-P release is expected all year round and will be greatly influenced by pH, dissolved oxygen and microbial activity.

Rowan (2011) conducted a comprehensive spatial survey of TP concentration in 38 bulk sediment samples collected from Loch Fitty in November 2010. The bulk samples averaged 19.6 cm in sediment depth. TP concentrations ranged from 0.11 to 2.45 mg TP g^{-1} d.w. Significant spatial variability was reported, with deeper water sediments having the highest TP concentrations. The mean TP concentration from the bulk sediment samples was 0.91 mg g^{-1} dw ± 0.18 mg TP g^{-1} d.w. At the whole loch scale, Rowan (2011) estimated a store of about 65 tonnes in the upper 20 cm of sediment. Significant positive correlations were reported between total Fe and TP, and total Mn and TP, indicating that a major component of the TP pool in the upper 20 cm is associated with redox-sensitive Fe-P and Mn-P complexes. These complexes represent the pool of sediment TP that is likely to be released to the water column under reducing conditions (Spears *et al*., 2007).

Dated depth profiles of TP, in the upper 29 cm of sediment, were assessed in two cores by Rowan (2011). As in most eutrophic lakes, the TP concentration increased significantly towards the sediment water interface, especially over the upper 5 cm ("active layer"). This increase is generally attributed to the migration of redox-sensitive phosphorus pools towards the aerobic sediment surface and to the fact that organic P and inorganic particulate P pools, originating from the water column, are deposited at the surface. These high TP surface sediment layers are those that are responsible for sediment-water interactions, and for prolonging the recovery of lakes from eutrophication following reductions in catchment loading (Farmer et al., 1994; Spears et al., 2007). The reported (Rowan, 2011) estimates of bulk sediment TP concentrations will, therefore, underestimate TP concentrations in the active layer as a result of dilution of surface sediments with deeper sediment layers (depth range of bulk sediment cores – 4 cm to 36 cm) of significantly lower TP concentrations.

Based on the available data, TP trends below the active layer (i.e. about 5 cm in Loch Fitty) may be used loosely to infer changes in TP concentrations in the loch over time. It should be noted that a limitation of using sediment TP depth profiles is that the redox-sensitive, labile and organic P pools are not well represented for reasons outlined above. Instead, the sediment P pools most likely to dominate in deeper, anaerobic, sediment layers are apatite-P, metal adsorbed-P and residual-P, i.e. those P pools that are likely to be driven by changes in the mineral signature of catchment inputs commonly driven by land-use changes (Boström *et al*., 1982). The results suggest that loch TP concentrations increased gradually between about 1950 and 1970 followed by a rapid increase to a peak in the mid-1970s. Following this, TP concentrations have gradually declined through to the 1990s. The values reported for the 1990s were more than double those reported for pre-1940s, indicating sustained high catchment loading.

At present, Loch Fitty has a relatively short residence time of 57 days and a mean depth of 2.2 m (Rowan, 2011). According to Sas's (1989) estimates of recovery time based on sediment TP concentrations, the upper confidence level mean sediment TP concentration for Loch Fitty of 1.19 mg TP g^{-1} d.w. (Rowan, 2011) corresponds to an estimated recovery period of less than 5 years as a result of internal loading alone. It should be noted that these estimated recovery times only apply once the external loading has been significantly reduced and will be enhanced at higher flushing rates. It should also be noted that these recovery time categories are based on meta-analyses and that specific site attributes (e.g. depth, effective fetch and residence time) are also important regulators. For example, the delay in recovery in Loch Leven (mean depth – 3.9 m, residence time ~ 146 days and sediment TP in 1995 and 2005 of about 2.5 mg TP g^{-1} d.w.; Farmer et al., 1995; Spears et al, 2007) was around 15 years. A comparison of the mean bulk sediment TP concentrations for Loch Fitty reported by Rowan (2011) together with a range of other lakes is shown in Figure 12.1. It should be noted that variations in sample design and procedure across the studies included in Figure 12.1 allows only rough comparison between lakes.

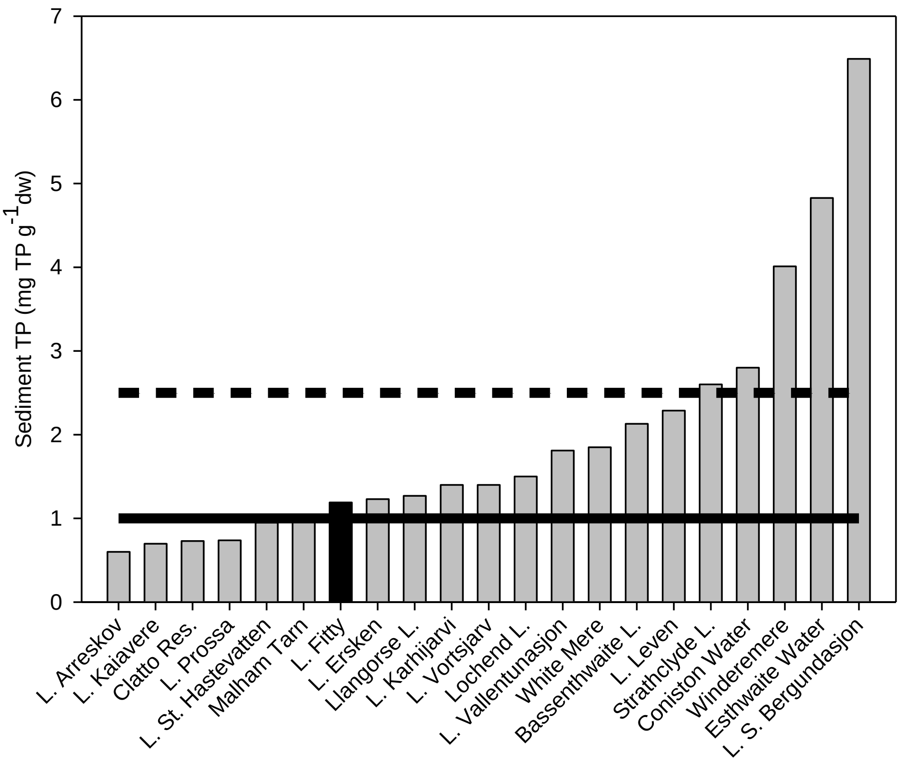

Figure 12.1 Summary of TP concentrations for surface sediments in lakes across the world, with the exception of Loch Fitty (black bar) where estimates are based on bulk sediment cores over a range of depths (4-36 cm). Following significant reductions in catchment nutrient load, lakes with concentrations that lie below the solid black line are expected to recover quickly, those with concentrations between the full line and the dashed line represent a recovery period of less than 5 years, and those with concentrations above the dashed line are expected to take more than 5 years to recover as a result of internal loading (Sas, 1989).

12.1.3 Fish and fishery management

Fish communities have been managed throughout Europe for hundreds of years. This has resulted in a huge scientific literature on the subject, the review of which is far beyond the remit of this report. However, a relatively recent review of the particular field of fish stocking and introduction can be found in the edited proceedings of a symposium on this subject edited by Cowx (1998). This contains a number of papers that are specifically concerned with brown trout and eel, both of which are key species in the present context. In terms of UK-specific projects in this area, Roger A. Sweeting (Honorary Research Fellow, Freshwater Biological Association, U.K. E-mail RSweeting@fba.org.uk, *pers. comm.*) has extensive personal experience of introducing and re-introducing fish to gravel pit lakes in the south of England, including observations of the subsequent rapid expansion of some species before the community settled down to some form of balance. Also relevant to this area, Keith J. Wesley (Bedwell Fish Farms and Bedwell Fisheries Services. Tel. 07802 783233) is a commercial provider of services with over 25 years of experience, including the restocking of de-silted or otherwise modified water bodies.

With a more recent history from the 1980s to the present, and a more restricted scope in terms of the species involved, fish community management has also been undertaken for biomanipulation purposes in which fish numbers and species composition are manipulated to produce cascading changes in the zooplankton, phytoplankton and, ultimately, in the water quality of a standing water body. An authoritative review of this field on a global basis is provided by Hansson *et al.* (1998). Such studies have relevance to the present project because biomanipulation in Europe frequently involves the removal, or at least great reduction, of roach populations due to this species frequently being a major predator of zooplankton populations, especially in eutrophic waters. Such biomanipulated fish populations have been removed or reduced by manual removal through netting or electric fishing, by the addition of predatory fish, or by the use of chemicals such as rotenone. The U.K. experience with biomanipulation, where fish removal/reduction methods are tightly restricted, is reviewed by Moss *et al.* (1996), with Perrow *et al.* (1997) providing a more fish-focused account and Phillips *et al.* (1999) concentrating on its practical aspects. In the U.K., such work has been focused almost entirely on the Norfolk Broads of south-east England, with the manual removal of roach and, to some extent, other species, by repeated electric fishing, some netting and some egg removal being reviewed by Perrow *et al.* (1997) and Phillips *et al.* (1999). One particularly clear conclusion which can be drawn from these and other studies is that the complete removal of roach and similar species from a standing water body is extremely difficult to achieve when it is attempted only by physical capture. Even if only a few individuals evade capture, the population biology of such small cyprinids is such that they will rapidly recover in population abundance within a few years. Andrea Kelly (Broads Authority, U.K. E-mail andrea.kelly@broads-authority.gov.uk, *pers. comm.*) has managed much of this work and has confirmed that, although fish are not being actively removed at present following successful earlier exercises, their eggs continue to be removed and their populations continue to be monitored by annual surveys.

12.1.4 Ingress of non-native invasive species

Non-native invasive species are defined by the GB non-native species secretariat as "any non-native animal or plant that has the ability to spread causing damage to the environment, the economy, our health and the way we live." The recovery of Loch Fitty would be adversely affected by ingress of any of a range of non-native invasive species. Loch Fitty is currently infested with the non-native invasive aquatic plant species *Elodea canadensis* (Canadian pondweed) and the non-native fish species roach). Although many non-native invasive species are introduced by humans for a specific purpose (i.e. stocking of non-native invasive coarse fish), by definition, they also have the ability to spread throughout the environment and infest lakes without human intervention. A summary of the types of non-native invasive species, and their vectors, that may affect the recovery of Loch Fitty by out-competing native species and resulting in an undesirable recovery end point, including the failure of the WFD quality targets, are outlined in Table 12.3.

Table 12.3. Summary of organism groups, their mode of introduction, and possible environmental impacts for non-native invasive species considered to pose a high risk to the recovery of Loch Fitty (after, Manchester and Bullock, 2000)**.**

Type of organism	Purpose	Impacts	Examples
Fish and shellfish	Angling, accidental introduction	Competition, predaition, habitat disturbance, disease vector, increased sediment P release, reduction of macrophyte cover	Grass carp, common carp, crayfish, rainbow trout
Invertebrates	Accidental introduction	Competition with native species, grazing of phytoplankton	Zebra mussel, alien gammarids
Plants	Accidental introduction	Prolific vegetative growth, forms dense matts leading to deoxygenation, outcompetes native flora.	*Crassula helmsii, Elodea Canadensis* and *E. Nutalii*

As it is extremely difficult to eradicate non-native invasive species once they have infested a lake, it is important to minimise the risk of infestation to Loch Fitty. This is especially important as this risk is likely to be heightened by the translocation and re-planting of the native plant species that currently inhabit the loch.

The proposed restoration work opens up two conflicting opportunities in relation to the control of non-native invasive species in Loch Fitty. These are:

- the opportunity to remove Canadian pondweed (*Elodea canadensi)* and roach populations, and

- the opportunity for new non-native species to colonise during the loch recovery phase.

Measures to reduce the risk of Loch Fitty being colonised of by non-native invasive species may include appropriately controlling public access during the re-colonisation phase and improving public awareness in line with the GB Non Native Species Secretariat "Be Plant Wise" campaign (https://secure.fera.defra.gov.uk/nonnativespecies/home/index.cfm). Care needs to be taken to ensure that holding ponds do not become infested with non-native invasive species. It should also be noted, however, the fact that some non-native invasive species can be introduced by waterfowl may make them difficult to control (e.g. *Crassula helmsii*, *Elodea Canadensis* and *Eloea nutalii*). An invasive species action plan should, therefore, be drawn up in collaboration with SEPA and the local community to identify high risk species and to monitor their presence/absence in the loch. These measures will reduce the risk of infestation and increase control over recovery to the desired end point.

12.1.5 Control of non-native invasive phytoplankton and phytobenthos

The phytoplankton and phytobenthos communities are generally considered to be ubiquitous due to their ease of distribution. As such, no practical management measures are considered necessary for these communities.

12.1.6 Control of non-native invasive macrophytes

In terms of recovery, there is a risk that invasive, macrophyte species could outcompete desirable species during the establishment phase. As part of the process of mitigating those risks, and to inform management of the expected rate of recovery, it is recommended that pilot work is undertaken to confirm the recruitment rates and growth rates of the native species. The results of this work can be used to determine sensible stocking densities. Similar pilot work has proven very effective in the restoration of rivers in Denmark (Riis *et al*., 2009; Riis *pers comm*.).

Equally, the species present in Loch Fitty are from groups that are known to exhibit different growth rates and there is anecdotal evidence that species that are closely related to those present in Loch Fitty may reproduce at different rates. This raises the possibility that the most competitive species may dominate at an early stage, leading to the exclusion of rarer, less competitive species (Grime *et al.,* 1988). A recolonisation plan informed by a pilot study, as described above, will mitigate this risk.

There are a number of further prophylactic steps that can be taken to prevent undesirable recovery scenarios. A critical first step is to reduce nutrient input to the lake, because enriched conditions favour undesirable species. On occasion, this has been sufficient to provide the conditions whereby native species can overcome invasive species (Pot & ter Heerdt, 2009).

Consideration should also be given to restricting access to propagule vectors, including people and birds, within the first few years of reinstatement. Any sources of propagules of invasive macrophyte species within the upstream area of the catchment should also be identified and controlled, especially during the recolonisation period.

Like freshly turned garden soil, a new lake without an established native plant community, is vulnerable to weedy growth. In the USA, where the US Army Corp of Engineers is responsible for building new reservoirs, this scenario is commonly observed in the years immediately following construction (Dr M. Grodowitz, US Army *pers. comm.*). Their experience is that the conditions found in new water bodies can encourage either nuisance algal blooms or aggressively invasive macrophytes, making it difficult for native species to establish. In either case, reversing the process is expensive or practically difficult, so prevention is the better option.

Other potentially more aggressive invasive macrophytes are known to occur locally and there is a risk of these becoming established in the new Loch Fitty. For example, Canadian pondweed (*Elodea canadensis*) is found in the nearby Loch Leven, Nuttall's pondweed (*Elodea nutalii*) is found in Linlithgow Loch and New Zealand pygmy weed (*Crassula helmsii*) is present in the wider area. Other species, such as Himalayan balsam and Japanese knotweed, also occur locally and can infest riparian zones. However, it is recognised that this risk exists for the waterbody currently and, indeed, the existing Loch Fitty suffers from invasive plant species. So, the above mitigation measures are proposed to mitigate this risk in the early years of establishment of the new loch.

The impermeable nature of the clay liner alone may slow colonisation, therefore, the provision of an additional substrate layer above the clay liner is welcomed and consideration should be given to using materials such as sands and low nutrient silts as rooting media that favour native species over non-native species. The seedbank of these materials should be assessed to ensure that non-native invasive species are not inadvertently introduced.

The prophylactic measures suggested above will help prevent problems with aggressive invasive species. The GB non-native species secretariat has recently published a range of guidance documents designed to provide bio-security advice to stop the spread of non-native invasive macrophyte species.

12.1.7 Control of non-native invasive macroinvertebrates

As a calcareous hardwater site, Loch Fitty is especially suitable for crustacea and mollusca, as calcium will be available for the shells of these animals. For this reason the loch could be susceptible to invasion from, for example, zebra mussel (*Dreissena polymorpha*), North American signal crayfish (*Pacifasticus leniusculus*) and alien gammarids (e.g. *Dikerogammarus villosus*). These pest species are not very prevalent in Scotland at present, either because they have not migrated to this area yet or because many Scottish waters are relatively low in calcium and not suitable for them. However, if they did successfully invade the new Loch Fitty, the WFD ecological status of the loch, under any measurement scheme, would probably be reduced. It is therefore important to prevent their ingress. Careful control on the sourcing of materials used in the construction of the loch, and the decontamination of any work/leisure craft or sampling equipment used on the loch should prevent the spread of these invasive species into the new loch.

The GB non-native species secretariat has recently published a range of guidance documents designed to provide bio-security advice to stop the spread of non-native invasive macroinvertebrate species.

12.1.8 Control of non-native invasive fish

In recent years, a number of fish communities have been managed with the aim of addressing the problems associated with invasive species. In Europe, the invasive (or 'undesirable') fish species in this context are usually cyprinids closely related to roach. Examples of such operations in mainland Europe include the work of Carlos Fernandez-Delgado (Departamento de Zoología, Campus Universitario de Rabanales, Universidad de Córdoba, Spain. E-mail carlos.fdelgado@uco.es, *pers. comm.*) who has used rotenone to remove introduced common carp (*Cyprinus carpio*) and mosquitofish (*Gambusia affinis*) from water bodies in southern Spain (Fernandez-Delgado, 2009). In one such waterbody, the Laguna de Zóñar with a surface area 37 ha and a maximum depth 14 m, both invasive species were successfully removed while a desirable native species (*Atherina boyeri*) was temporarily held elsewhere and then successfully reintroduced. In a second water body with a surface area 87 ha and a maximum depth 2 m, only the common carp was successfully eradicated. Elsewhere in Spain, similar work is currently underway under a LIFE Nature project (LIFE08 NAT/E/000078) at the Lake Area of the Estany de Banyoles, in the north-east of the country, under the leadership of Miguel Campos (Consorci De l'Estany, Banyoles, Spain. E-mail mcampos@consorcidelestany.org).

Within the U.K., work on the control of invasive fish species has concentrated on the small cyprinid topmouth gudgeon (*Pseudorasbora parva*) and has been developed by Rob Britton (Centre for Conservation Ecology and Environmental Change, School of Conservation Sciences, Bournemouth University, U.K. E-mail rbritton@bournemouth.ac.uk) and Matt Brazier (Environment Agency, U.K. E-mail matt.brazier@environment-agency.gov.uk), with the latter individual subsequently leading the national application of the developed technique for the Environment Agency. This work involves the temporary removal by netting of desirable native fish for safe-keeping elsewhere during the eradication process, which is then itself accomplished using the piscicide rotenone. Such use of rotenone remains potentially controversial in the U.K. and so is understandably subject to detailed restrictions. In the present context, significant research was required to develop the technique into an acceptable protocol. This research has included the determination of appropriate treatment procedures (Allen *et al.*, 2006), the detailed documentation of their field application at a number of sites (Britton & Brazier, 2006; Britton *et al.*, 2007; Britton *et al.*, 2008), and a more strategic consideration of this approach to the issue (Britton *et al.*, 2010). In addition to a remarkable record of successful eradication, this rotenone-based approach has also been shown to lead to no long-term problems at the application sites as evidenced by subsequently increased growth and production of native fish (Britton *et al.*, 2009).

12.2 Evidence of chemical and ecological recovery from similar case studies

Information on similar projects within the UK and across the rest of Europe was sought by reviewing scientific literature and consulting colleagues within the international research community. This information was used to evaluate possible management issues, likely recovery times and potential contingency management options at Loch Fitty. The survey returned the following general results:

- sediment removal has been widely used in restoration projects across Europe,
- a response in P, chlorophyll *a* and, to a lesser extent, nitrogen concentrations were most commonly reported,
- no studies have reported on European lake restoration that directly addressed WFD-compliance, or any other recovery targets for BQEs of restored lakes
- recolonisation by macrophytes, phytoplankton and zooplankton after sediment removal has been reported for a small number of UK and other European lakes,
- ecological recovery is currently being comprehensively studied in a range of created lakes across Europe from which practical management lessons may be learned,

- a small number of lakes within the Czech Republic have been created on spent brown coal mining sites; the recolonisation of the fish community has been managed and monitored in these lakes, and
- there a number of sites to which visits could be made that would provide useful information on the recovery process.

Stewart (2010) has documented many examples of lake creation projects within the UK, but with little or no indication of their success or failure. The locations of these projects include Cotswold Water Park (Wiltshire and Gloucestershire), Windrush Valley (Oxfordshire), Cresswell (Northumberland), and the Mountcastle/Wilderness sand quarries (Fife). Stewart's report also includes guidance for mineral extraction companies that has been produced by the Ponds Conservation Trust, and which is considered to be applicable to the proposed work on Loch Fitty. However, this guidance is only aimed at strategies for the promotion of stoneworts (charophytes) in new lakes. Although this will be important in the restoration of Loch Fitty, the needs of other species must also be taken into account. Stewart (2010) addresses these needs, to some extent.

There are many examples of lake restoration projects throughout Europe, although most of these have been attempted without draining the lakes and completely removing the sediments. Many lessons that are pertinent to the proposed work at Loch Fitty can be learned from these examples. Many of these case studies have been reviewed by Gulati *et al.* (2008) and Søndergaard *et al.* (2007), who highlight several factors that are important for successful restoration:

1. the reduction of inputs of P and N from external sources such as inflowing streams and groundwater, and from internal sources such as sediment release

2. the removal or control of planktivorous fish, which allows the proliferation of zooplankton grazers; these, in turn, remove many of the phytoplankton that would otherwise compete with submerged aquatic plants for light

3. the reduction of sediment resuspension by removing bottom-feeding fish or reducing wind-induced mixing (e.g. by reducing fetch)

Although few examples of lake creation *per se* are documented in the scientific literature, it is likely, that this has been attempted many times, given the examples found by Stewart (2010) within the UK. Given the relative complexity of the Loch Fitty macrophyte recolonisation strategy, it would be helpful (during the operational period of the development) to explore which of these projects have been considered to be successes or failures and what the main causes of these successes/failures were, if this information is available. Such information may provide some important lessons that have relevance to the Loch Fitty restoration project.

12.2.1 Pheonix See, Germany - a lake creation site in progress

Pheonix See is a lake in Dortmund, Germany, that has been created on an old blast furnace and steel plant site. The created lake is going through the early phases of management and ecological colonisation. This includes macrophyte and fish introduction work and a strict creation plan that includes initial control of internal loading by laying sand. The lake is 1.2 km long and 320 metres wide, with a total surface area of 24 ha and a maximum depth of 3-4 m. The lake is fed, primarily, from groundwater inputs. The development has brought together a range of experts from the fields of hydrogeology, ecology and lake system remediation who are currently working together towards the creation of a healthy functioning ecosystem. This management process is extremely dynamic and the project group for Loch Fitty would benefit from visiting the site to discuss the approach taken there, the results achieved and the lessons learned.

Figure 12.2 Aerial photograph of Pheonix See, Dortmund, Germany (image from http://www.phoenixseedortmund.de).

The Loch Fitty project group has been invited to discuss the Pheonix See creation project with Heinz Hueppe, the Chief Executive of the Pheonix See development project. His contact details are as follows:

Heinz Hueppe
PHOENIX See Entwicklungsgesellschaft mbH Barcelonaweg 14
44269 Dortmund
http://translate.google.com/translate?hl=de&sl=de&tl=en&u=http://www.phoenixseedortmund.de/2.html&rurl=translate.google.de
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Fax: +49(0)231 / 22 22 77 - 19
eMail:hueppe@phoenixseedortmund.de
http://www.phoenixseedortmund.de

12.2.2 Lake Kraenepoel, Belgium – restoration by de-watering, dredging and biomanipulation

Lake Kraenepooel (mean depth 1.0 m, maximum depth 1.5 m) has a surface area of 22 ha and was split into two separate basins in 1957 by the creation of a dyke. The north basin was drained every 10 years to harvest its fish population, whereas the south basin was not. The south basin was fed by nutrient rich inflow waters, whereas the north basin was not. Both of the basins have suffered from eutrophication, although the north basin retained a healthy *Littorelettea* vegetation, whereas the south basin became turbid and phytoplankton dominated. In order to reduce nutrient concentrations in the lake, the nutrient rich inflow was diverted from the southern basin, both basins were drained and their fish populations were removed, and the north basin was dredged to remove nutrient rich sediments whereas the south basin was not. Both basins were stocked with juvenile Pike. Van Wichelen et al. (2007) documented the chemical and ecological recovery of both basins for 1 year prior to and 2 years post restoration. The key results of this study in relation to the proposed work on Loch Fitty are outlined below:

- Ammonium and phosphate concentrations and phytoplankton biomass all decreased in the north basin (with sediment removal) within 1 year of restoration, mainly because in-lake nutrient cycling shut down.

- This improved water quality in the north basin was associated with a shift from abundant *Potamogeton* spp. to a more “benthic” *Littorelletea* vegetation; of particular note was the emergence of *Hypercium elodes* and an increase in the number of pre- and post-restoration desmid species from 3 to 30.

- In the southern basin (without sediment removal), no reduction in N and P concentrations was observed, despite the removal of external nutrient inputs; it is likely that the source of these nutrients was the sediments given the high remineralisation rate that is expected to be associated with dewatered surface lake sediments.

- The south basin became acidified (to about pH 4) a few months after refilling, probably as a result of oxidation of sulphides during the drying period and subsequent release of sulphides to the water column on refilling; refilling with rainwater probably reduced the buffering capacity of the lake to changes in pH.

- Acidification resulted in the immobilisation of phosphates at low pH and favoured nitrification; direct pH effects on the ecology of the lake were also expected and a shift from *Potamogeton* spp. to *Littorelletea* vegetation was not reported in the south basin.

12.2.3 Mine lakes in the Czech Republic- emerging evidence on fish population management

Information was sought on similar fish community projects elsewhere in the U.K. and across the rest of Europe by combining of a search of the scientific literature with enquiries to over 50 personal contacts in appropriate research and management fields. Unsurprisingly, no published scientific accounts of similar projects were found, but published and unpublished investigations of varying relevance were discovered Furthermore, a number of personal contacts (see below) confirmed that activity in most of these areas continues to the present day. These contacts also revealed limited, but current, activity in the area of fish community establishment following reservoir dewatering and the creation of new lakes when extraction activities, including brown coal mining, had ceased. The information summarised below is, largely, from unpublished sources.

Pavel Jurajda (Department of Fish Ecology, Institute of Vertebrate Biology, Czech Republic. E-mail jurajda@brno.cas.cz, *pers. comm.*) has some experience of dewatering and the subsequent rebuilding of fish communities in reservoirs in the Czech Republic. Specifically, the 220 ha Brno Reservoir was dewatered to 70 ha for a year during which time sediment and cyprinids were removed with the ultimate aim of water quality improvement, i.e. a form of biomanipulation. At present, two smaller reservoirs (Plumlov Reservoir and Luhacovice Reservoir) are completely dewatered and their sediment is being removed prior to refilling next year.

In terms of the construction of fish communities in new water bodies within the U.K., this is primarily restricted to the development of flooded gravel pits for angling, and to some extent for nature conservation, purposes. The experience of Roger A. Sweeting noted above in a fisheries context is also relevant here, as is a review by Giles *et al.* (1992) relating to 20 years of practical work on the development of such habitats including their fish communities.

Finally, Jan Kubecka (Biology Centre of Czech Academy of Sciences, Institute of Hydrobiology, Czech Republic. E-mail kubecka@hbu.cas.cz, *pers. comm.*) is currently leading the development and monitoring of fish communities following lake creation after the completion of open-cast brown coal mining activities in the Czech Republic. Specifically, he is currently running three relevant projects in relation to the creation of new lakes and the construction of appropriate fish communities on the basis of the species that would be expected to be present if the lakes were completely natural. Although this work has not yet reached full publication, several years of post-construction monitoring data for the specific site of Chabařovice Lake are already held and this work is briefly summarised in Peterka *et al.* (2010). Additionally, Martin Neruda has been working with Jan Kubecka to assess responses in plankton, flora and fauna in Lake Most (Martin Neruda, Jan Evangelista Purkyně University, Czech Republic, Email: Martin.Neruda@ujep.cz; *pers. comm.*). Martin Neruda will be visiting Scotland in June 2011 and is willing to discuss this work with the project group.

The above review of similar projects and individuals involved allows a number of specific recommendations to be made concerning appropriate further contacts and site visits. In the U.K., to develop understanding of the likely dynamics of the introduced fish populations, further contacts are recommended with Roger A. Sweeting (Freshwater Biological Association), potentially with Keith J. Wesley (Bedwell Fish Farms and Bedwell Fisheries Services) if a commercial operator is required, and, in particular, with Andrea Kelly (Broads Authority) and Matt Brazier (Environment Agency). Site visits are also recommended to locations where projects managed by the latter two individuals have been undertaken. At the same time, it is recommended that contact with Matt Brazier also considers the potential value and practical considerations, including public acceptability, of the use of rotenone in the present project. Should further consideration of such use be considered appropriate, further international contacts are recommended with Carlos Fernandez-Delgado (Universidad de Córdoba, Spain) and Miguel Campos (Consorci De l'Estany, Banyoles, Spain). Also elsewhere in Europe, in the context of rebuilding fish communities further contact is recommended with Pavel Jurajda (Institute of Vertebrate Biology, Czech Republic), potentially with an associated site visit to the Brno, Plumlov and Luhacovice Reservoirs in the Czech Republic. Finally, and most importantly in the context of establishing new fish communities, further contact is highly recommended with Jan Kubecka (Institute of Hydrobiology, Czech Republic) together with a site visit to his current project locations in the Czech Republic, particularly Chabařovice Lake.

12.2.4 The Wetland Centre, London - water level, waterfowl and colonisation trajectories

An example of a well documented lake creation project is given by Yallop and O'Connell (2000), who followed the developmental progress of three artificial lakes at the Wetlands Centre, London. The project was designed to enhance habitat for waterfowl by introducing macrophytes as a food source and as a refugia habitat for macroinvertebrates and fish in these lakes. The lakes were created in 1995 and inundated between 1996 and 1997. In this scheme, early planting of aquatic plants was abandoned due to extensive growths of filamentous macroalgae and, for the first two years, all three lakes were characterised by such growths together with frequent phytoplankton blooms, high concentrations of total phosphorus (1996-1999; 200-5000 µg TP L^{-1}) and high turbidity. Phosphorus concentrations decreased in all of the lakes in 1999, although they remained relatively high (>100 µg TP L^{-1}).

In 1999, one of the lakes, Sheltered Lagoon, underwent a switch from a phytoplankton dominated state to a clear water macrophyte dominated state. This coincided with the development of extensive beds of the macrophyte *Myriophyllum spicatum* (spiked water-milfoil) and a reduction in the cover of filamentous algae. This switch was not caused by the lowering of nutrient concentrations. Instead, it has been attributed to the shallow and sheltered nature of the lake and the fact that a drop in water level preceding the switch resulted in a "window of opportunity" where light levels at the sediment surface were temporarily increased.

Between 1999 and 2001, two of the lakes, Sheltered Lagoon and Main Lake, oscillated between a turbid, phytoplankton dominated state and a macrophyte dominated state, whereas Reservoir Lagoon, remained turbid and dominated by phytoplankton and benthic filamentous green algae (dominated by *Enteromorpha sp.*), throughout. Waterfowl grazing was identified as the main driver of the undesirable phytoplankton dominated state in Reservoir Lagoon (high grazing) and the relatively high diversity of macrophytes in Main Lake (low grazing). Both water level fluctuations (increasing competitive advantage) and elevated pH levels (pH> 10 decreasing propagule survival) affected macrophyte communities in Sheltered Reservoir (Yallop *et al*., 2004).

The various state shifts experienced by the Wetland Centre lakes are summarised in Table 12.4.

Table 12.4 Summary of state shifts in The Wetland Centre lakes, after Yallop *et al*. (2004). RL: Reservoir Lake; ML: Main Lake; SL: Sheltered Lagoon. Clear: clear water macrophyte dominated state (with dominant macrophyte species indicated in brackets); Turbid: phytoplankton or benthic filamentous algal dominated turbid water state; TP and phytoplankton crop measured as summer means (May - September). Macrophytes consumed expressed as percentage of macrophyte community estimated to be removed by calculated grazing rates for waterfowl community.

Lake name	Year	State	TP ($mg\ L^{-1}$)	Macrophyte max. biomass ($kg\ dw^{-1}\ ha^{-1}$)	Phytoplankton crop ($\mu g\ Chl_a\ L^{-1}$)	Macrophytes consumed by waterfowl (%)
SL	1998	Turbid	0.2 – 5.0	ND		
	1999	Clear (M)	0.38	2450	13	
	2000	Clear (E^1,M)	>0.10	1983	11	7.7
	2001	Turbid (E^1)		Low	150	7.3
RL	1998	Turbid	0.2-5.0	ND		
	1999	Turbid	0.39	620	45	
	2000	Turbid	>0.10	1650	45	22.0
	2001	Turbid			58	15.0
ML	1998	Turbid	0.2-5.0			
	1999	Turbid (Z)	0.19	230	105	
	2000	Clear (P^1,P^2,E^2)	>0.10	1717	8	32.9
	2001	Clear-Turbid ($P^{1,2}$)			46	8.5

Species key – *Myriophyllum spicatum* (M); *Elodea nutalii* (E^1); *Zannichellia palustris* (Z), Potamogeton pectinatus (P^1); *Potamogeton pusilis* (P^2); *Elodea Canadensis* (E^2).

This case study highlights the importance of the stable states hypothesis when considering the climax ecological structure in newly created shallow lakes (Scheffer, 1998). In the stages of early lake succession, the dominant autotrophic community will be fast growing phytoplankton and filamentous benthic algae where TP concentrations are high (i.e. summer TP concentrations of 200-5000 µg TP L^{-1}, in this case). Once this community has established, it will shade out macrophytes, thereby reducing the likelihood of a switch from phytoplankton to macrophyte dominated state. The high biomass of phytoplankton will die back during the winter months returning P to the sediment where it will be remineralised and released into the water column the following year, thus sustaining the feedback loop and resilience to change. Further resilience to change was evident in the above case study in the form of high waterfowl grazing of macrophytes and high pH reducing propagule success. The steady state hypothesis proposes that shallow lakes may switch between phytoplankton and macrophyte dominated states, with no change in P load, as a result of a disturbance (Scheffer, 1998). In the case of Sheltered Lagoon, it was initially hypothesised that this switch took place as a result of a temporary increase in the euphotic depth associated with water level reduction (Yallop & O'Connell, 2000). The fact that this phenomenon was not observed in Main Lake or Reservoir Lagoon was attributed to this lake being particularly sheltered from wind induced sediment disturbance. It is, therefore, evident that lakes may follow alternative colonisation trajectories based on lake specific pressures.

It should be noted that a range of natural disturbances (excluding increased P loading, which has been well studied) have been reported in the literature, including extreme weather events, fish stocking practices and waterfowl grazing of macrophytes and that resilience of the macrophyte dominated state to change increases at low TP concentrations (Scheffer, 1998). A small number of studies have also investigated forcing a desirable state change to macrophyte dominance through controlled ecosystem disturbances. These include the control of sediment P release using P-capping agents (e.g. Loch Flemington, B.M. Spears, unpublished data, http://www.ceh.ac.uk/sci_programmes/water/LochFlemington.html), biomanipulation studies (Hosper, 1998) and sediment removal programmes (Søndergaard et al., 2007).

12.2.5 Olentangy River Wetland Research Park – managing colonisation trajectories

The Olentangy River Wetland Research Park (ORWR) is a research site on which two wetlands (1 ha in area; < 1 m in depth) were created to have similar hydrological signatures (Mitsch et al., 2005). These wetland systems included both large shallow areas to favour marginal macrophyte species and open water "deeper" (up to 80-100 cm) areas to support submerged macrophyte species. The project addressed a range of hypotheses designed to compare the effects of macrophyte introduction by planting to natural macrophyte ingress. This hypothesis that was relevant to the proposed work at Loch Fitty was that "planted and unplanted basins will be similar in function in the beginning, diverge in function during the middle years and ultimately converge in structure and function", as conceptualised in Figure 12.3.

The theoretical recovery trajectories presented in Figure 12.3 were comprehensively tested in the two ORWR constructed wetlands using macrophyte community composition and productivity estimates over a 10 year period following wetland creation (Mitsch *et al*., 2005). The key results from this study included:

- Survival of planted macrophytes was higher for the emergent community (~ 5 – 80% survival of individual plants) than the submerged community (<10 % survival; *Potamogeton pectinatus*) within 1 month of planting.

- Macrophyte cover increased annually in each wetland for the first 5 years, with an apparent stable community being established at this time.

- Macrophyte colonisation rate was higher in the planted wetland for the first 3 years.

- After 3 years, the unplanted wetland was dominated by a monoculture of *Typha* spp., whereas the planted wetland had a more diverse community (*Sparganium eurycarpum*, *S. tabernaemontania*, *Typha* spp. and *Scirpus fluviatilis*).

- Herbivorous grazing by muskrats reduced macrophyte cover in both wetlands (to 27.6% and 17.4% cover in the planted and unplanted wetland, respectively) in Year 6. This peak in muskrat grazing was attributed to an association between high macrophyte cover in Year 5 leading to an increase in muskrat productivity as a result of improved habitat. Muskrat numbers dropped in Year 7 following the low macrophyte cover in Year 6.

- An experimental drawdown was conducted in the spring of Year 7 in an attempt to "re-set" the seed bank. This resulted in the highest recorded macrophyte cover (73% and 74% cover for the "planted" and "non-planted" wetland, respectively) and a decrease in the dominance of *Typha* spp. in both wetlands. The community structure of the wetlands had generally converged by this point.

- Macrophyte productivity (expressed as dry weight) increased in both wetlands throughout the study. Productivity was consistently higher in the unplanted wetland with the onset of an apparent plateau reported in the unplanted wetland, only, in Year 10.

- Macrophyte community diversity (as community diversity index) was consistently higher in the planted wetland than in the unplanted wetland.

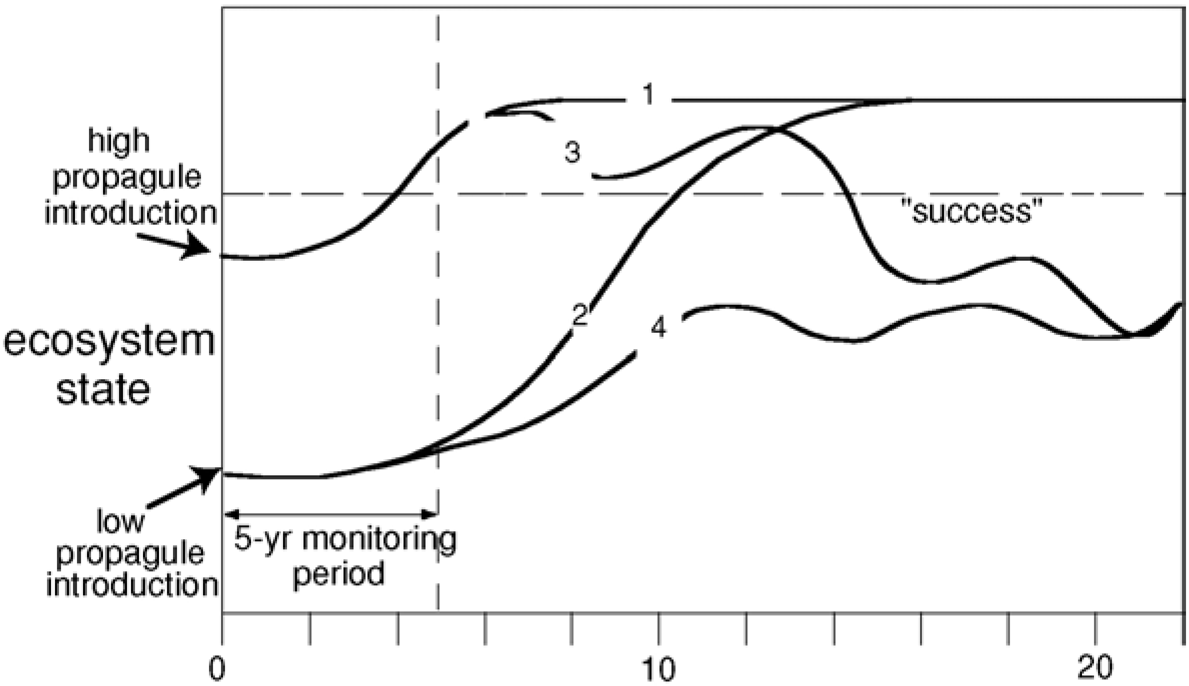

Figure 12.3 Hypothetical recovery scenarios based on macrophyte colonisation for created wetlands (after Mitsch *et al.*, 2005). Scenarios 1 and 2 represent recovery trajectories with no secondary pressure, whereas Scenarios 3 and 4 represent recovery trajectories with secondary pressures leading to deterioration in the recovery end point in terms of ecosystem state.

Based on the results outlined above, it would appear to be sensible to wait until the macrophyte community has established and reached a steady state before introducing the fish. This would reduce the likelihood of negative fish impacts in the important early years of macrophyte colonisation and allow targeted contingency management to be put in place where necessary. Additionally, planting of desirable species will increase the resilience of the system to undesirable monocultures, although the end point community structure may not be dependent on the stocked plant community structure. Finally, as outlined already in this document, it is essential that non-native invasive species (especially macrophytes and fish), and perhaps also non-native waterfowl species, be excluded from the site during the early macrophyte colonisation phase where possible.

13. Conclusions

The proposed surface mining project provides a unique opportunity to restore Loch Fitty. It will address some key improvements that are essential for helping the loch meet WFD targets, i.e. good water quality and good hydromorphological conditions.

Good water quality can only be attained by addressing sources of nutrient pollution within the loch and its catchment, and by removing heavy metal laden loch bed sediments. This will ensure that previous water quality problems at the loch do not re-occur and that the improvement in status of the waterbody is sustainable over time.

Ensuring that invasive non-native species cannot return to the loch requires the ecological quality of the inflows and outflow to be improved. This is an additional benefit of the proposed loch restoration project.

Overall, the proposal of SCCL for restoring Loch Fitty has a better chance of delivering the required end result of Good Ecological Status than the River Basin Management Plan proposal, because failure to meet WFD ecological quality targets at this site cannot be resolved by catchment management, alone. Pressures due to hydromorphological modification, non-native species and the accumulation of heavy metals in the sediments also need to be addressed.

14. Acknowledgements

We thank our colleagues from across Europe, too numerous to name here individually, for their swift and informative responses to our requests for information on current projects on fish community management. We also thank Peter Maitland for access to his fish occurrence records.

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